ISSN 1831-9424

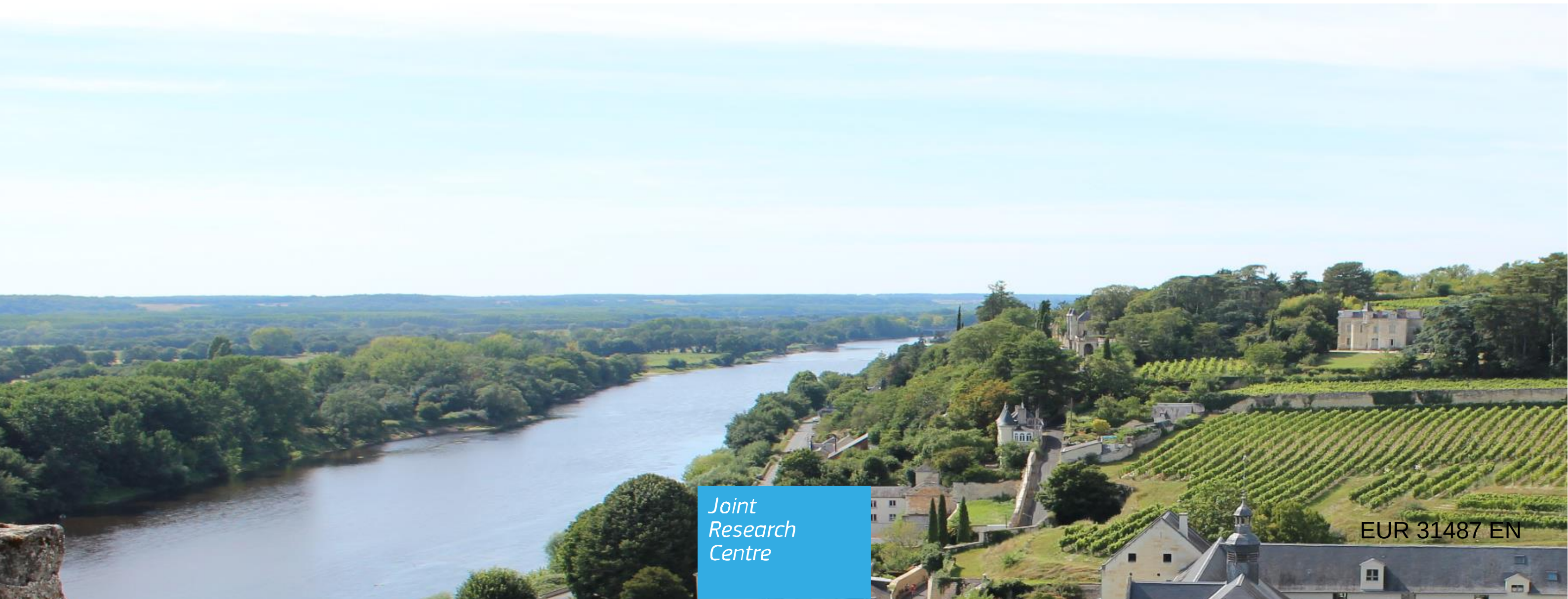
European Commission

JRC TECHNICAL REPORT

Knowledge for Integrated Nutrient Management Action Plan (INMAP)

Grizzetti, B, Vigiak, O, Aguilera, E, Aloe, A, Biganzoli, F, Billen, G, Caldeira, C, de Meij, A, Egle, L, Einarsson, R, Garnier, J, Gingrich, S, Hristov, J, Huygens, D, Koeble, R, Lassaletta, L, Le Noë, J, Liakos, L, Lugato, E, Panagos, P, Pisoni, E, Pistocchi, A, Sanz Cobeña, A, Udias, A, Weiss, F, Wilson, J, Zanni, M.

2023



Joint Research Centre

EUR 31487 EN

Contact information
Name: Bruna Grizzetti
Address:
Email: bruna.grizzetti@ec.europa.eu
Tel.:

EU Science Hub
https://joint-research-centre.ec.europa.eu

JRC129059

EUR 31487 EN

PDF ISBN 978-92-68-02654-0 ISSN 1831-9424 doi:10.2760/692320 KJ-NA-31-487-EN-N

Luxembourg: Publications Office of the European Union, 2023

How to cite this report: Grizzetti, B, Vigiak, O, Aguilera, E, Aloe, A, Biganzoli, F, Billen, G, Caldeira, C, de Meij, A, Egle, L, Einarsson, R, Garnier, J, Gingrich, S, Hristov, J, Huygens, D, Koeble, R, Lassaletta, L, Le Noë, J, Liakos, L, Lugato, E, Panagos, P, Pisoni, E, Pistocchi, A, Sanz Cobeña, A, Udias, A, Weiss, F, Wilson, J, Zanni, M, *Knowledge for Integrated Nutrient Management Action Plan (INMAP)*, Publications Office of the European Union, Luxembourg, 2023, doi:10.2760/692320, JRC129059.

Contents

Abstract

In the Biodiversity Strategy to 2030, the Farm to Fork Strategy and the Zero Pollution Action Plan the European Union has set an ambitious and ground-breaking goal to reduce by 50% nutrient losses to the environment (air, water, soil) by 2030, while preserving soil fertility. To this end, the Commission will work with Member States to develop an Integrated Nutrient Management Action Plan (INMAP). The 'Knowledge for INMAP' project, developed by the JRC during the year 2021, aimed to gather scientific knowledge and data available in the EU to support the discussion and preparation of the Integrated Nutrient Management Action Plan. In particular, the work focused on three major tasks: 1) the description of the current flows of nitrogen and phosphorus in the EU considering all sources and sectors involved (agriculture, industries, urban, energy and transport) and all environmental losses in air, water, and soils; 2) the evaluation of the distance to environmental targets, considering the EU legislation and strategies; 3) the analysis of measures to reduce nutrient pollution at different intervention points in the nutrient cycle. Online map viewers and dashboards were also developed to facilitate the visualisation of the collected data and the analysis of regional pollution hotspots and major polluting sources.

Authors

Grizzetti B.[1], Vigiak O.[1], Aguilera E.[2], Aloe A.[3], Biganzoli F.[4], Billen G.[5], Caldeira C.[1], de Meij A.[6], Egle L.[1], Einarsson R.[2,7], Garnier J.[5], Gingrich S.[8], Hristov J.[1], Huygens D.[1], Koeble R.[3], Lassaletta L.[2], Le Noë J.[9], Liakos L.[4], Lugato E.[1], Panagos P.[1], Pisoni E. [1], Pistocchi A. [1], Sanz Cobeña A.[2], Udias A. [1], Weiss F. [1], Wilson J. [1], Zanni M.[3]

Affiliations:

1 European Commission, Joint Research Centre (JRC) - (Unit B5) Dries Huygens, Lukas Egle, (Unit C5) Julian Wilson, Enrico Pisoni, (Unit D2) Bruna Grizzetti, Olga Vigiak, Angel Udias, (Unit D3) Carla Caldeira, Panos Panagos, (Unit D4) Jordan Hristov, (Unit D5) Emanuele Lugato, Franz Weiss, (Unit I2) Alberto Pistocchi

2 CEIGRAM, Universidad Politécnica de Madrid, Spain

3 Arhs Developments - (Milano, Italy) Michela Zanni, Renate Koeble, (Luxembourg) Alberto Aloe

4 Unisystems - (Milan, Italy) Fabrizio Biganzoli, (Luxembourg Sàrl, Luxembourg) Leonidas Liakos

5 Sorbonne Université and CNRS, Paris France

6 MetClim, Varese, Italy

7 Chalmers University of Technology, Gothenburg, Sweden

8 Boku University, Vienna, Austria

9 Geology Laboratory of the École Normale Supérieure, PSL University, France

Authors' contribution (CRediT):

Grizzetti B.: Conceptualization, Supervision, Project administration, Funding acquisition; Methodology, Formal analysis, Writing - original draft (Sections 1; 2.1; 2.3; 3.2, 3.5, 3.6; 4.6, 5); all sections: Review and editing. **Vigiak O.**: Sections 2.1; 2.3, 4.6: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Aguilera E.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Aloe A.**: Software, Data Curation, Visualization. **Biganzoli F.**: Section 2.2: Methodology, Formal analysis, Investigation, Data Curation; all sections: Review and editing. **Billen G.**: Section 4.7: Methodology, Formal analysis, Investigation, Writing – original draft; all sections: Review and editing. **Caldeira C.**: Section 2.2: Methodology, Formal analysis, Investigation, Data Curation, Writing – original draft; all sections: Review and editing. **de Meij A.**: Sections 3.1, 4.3: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Egle L.**: Sections 3.4, 4.1: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Einarsson R.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Garnier J.**: Section 4.7: Methodology, Formal analysis, Investigation, Writing – original draft; all sections: Review and editing. **Gingrich S.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Hristov J.**: Section 4.6: Formal analysis, Investigation; all sections: Review and editing. **Huygens D.**: Sections 3.4, 4.1: Methodology, Formal analysis, Investigation, Writing – original draft; all sections: Review and editing. **Koeble R.**: Sections 3.3, 4.2: Methodology, Formal analysis, Investigation, Writing – original draft; all sections: Review and editing. **Lassaletta L.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Le Noë J.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Liakos L.**: Section 4.5: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Lugato E.**: Section 4.4: Methodology, Formal analysis, Investigation, Data Curation, Writing – original draft; all sections: Review and editing. **Panagos P.**: Section 4.5: Methodology, Formal analysis, Investigation, Data Curation, Writing – original draft; all sections: Review and editing. **Pisoni E.**: Sections 3.1, 4.3: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Pistocchi A.**: Sections 3.2, 4.6: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Sanz Cobeña A.**: Section 4.7: Methodology, Formal analysis, Investigation; all sections: Review and editing. **Udias A.**: Sections 2.1; 4.6: Methodology, Software, Formal analysis, Investigation; all sections: Review and editing. **Weiss F.**: Section 4.6: Formal analysis, Investigation; all sections: Review and editing. **Wilson J.**: Sections 3.1, 4.3: Methodology, Formal analysis, Investigation, Writing – original draft; all sections: Review and editing. **Zanni M.**: Software, Data Curation, Visualization.

1 Introduction

In the Biodiversity Strategy to 2030[1], the Farm to Fork Strategy[2] and the Zero Pollution Action Plan[3] the European Union has set an ambitious and ground-breaking goal to reduce by 50% nutrient losses to the environment (air, water, soil) by 2030, while preserving soil fertility. “This will be achieved by implementing and enforcing the relevant environmental and climate legislation in full, identifying with Member States the nutrient load reductions needed to achieve these goals, applying balanced fertilisation and sustainable nutrient management, and by managing nitrogen and phosphorus better throughout their lifecycle. To this end, the Commission will work with Member States to develop an Integrated Nutrient Management Action Plan in 2022”[4] (INMAP). The plan will cover all sectors and environmental compartments involved in the nitrogen (N) and phosphorus (P) cycles.

The ‘Knowledge for INMAP’ project, developed by the JRC during the year 2021, aimed to gather scientific knowledge and data available in the EU to support the discussion and preparation of the INMAP. In particular, the work focused on three major tasks: 1) the description of the current flows of N and P in the EU considering all sources and sectors involved (agriculture, industries, urban, energy and transport) and all environmental losses in air, water, and soils; 2) the evaluation of the distance to environmental targets, considering the EU legislation and strategies; 3) the analysis of possible measures to reduce nutrient pollution at different intervention points in the nutrient cycle.

This report presents the results of the project. The report is organised in three parts, following to the project’s tasks: Chapter 2 presents nutrient flows for Europe, according to available data sources; Chapter 3 describes the EU environmental legislation related to nutrients and the distance to policy targets; and Chapter 4 discusses measures to reduce nutrient losses to the environment making reference to recent scientific studies and new modelling assessments. Additional information on the methodological approach and references to other international projects and initiatives relevant for the preparation of the INMAP are provided in the Annexes.

In the ‘Knowledge for INMAP’ project, online map viewers and dashboards were developed to facilitate the visualisation of the collected data and the analysis of regional pollution hotspots and major polluting sources. They are available at https://water.jrc.ec.europa.eu (Integrated nutrient management page).

In the present work many data sources were used (and combined) for the analysis N and P stocks and flows in the different sectors and environmental compartments. Similarly, for the scenarios analysis different modelling tools were considered that are based on specific datasets and assumptions. Therefore, the study cannot ensure a complete coherence of all the datasets adopted. It focused on gathering relevant scientific knowledge available in Europe for the preparation of the INMAP, acknowledging that uncertainty in flows estimations is part of the complexity of the N and P cycle analysis.

[1] COM(2020) 380 final. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS. EU Biodiversity Strategy for 2030. Bringing nature back into our lives.

[2] COM(2020) 381 final. 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 for a fair, healthy and environmentally-friendly food system.

[3] COM(2021) 400 final. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS. Pathway to a Healthy Planet for All. EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil'.

[4] COM(2020) 380 final.

2 Nitrogen and phosphorus flow

To support the preparation of the INMAP, available data for Europe (by June 2021) were collected and analysed to describe the current N and P cycles in Europe, with the aim to understand the magnitude of nutrient fluxes and the role of all sources and sectors involved. The main sectors of emissions included agriculture, industries, urban settlements, energy and transport. Nutrient losses to air, soil and water were considered in the study. In order to cover the majority of fluxes, both reported data by countries and estimations by modelling were adopted. The assessment of nutrient flows was carried out using data of most recent years (centred on 2015 or the most recent period available). Major flows of N and P cycles were quantified for the entire EU27 and for each EU country (Section 2.1). Similarly a detailed material flow analysis on nutrients in the food system was developed, including food waste (Section 2.2). Information on N and P losses to the environment were also calculated at different spatial resolution, depending on data availability, including administrative (country, NUTS2) and hydrological units (river basin), with the intention to show regional impacts and pollution hotspots, and support several planning levels (Section 2.3). Web maps and dashboard applications were developed to browse interactively information on nutrient fluxes and sources contribution (https://water.jrc.ec.europa.eu).

2.1 Assessment of major nutrients flows

Human activities have altered the natural nitrogen and phosphorus cycles, increasing the amount of nutrients losses to air, water and soils, with impacts for human health, ecosystem functioning, biodiversity and climate change. In 2011 the first European Nitrogen Assessment (ENA) was published, gathering scientific knowledge on the alteration of the nitrogen cycle in Europe and its consequences on air and water quality, terrestrial biodiversity and climate, and drawing attention on possible actions to curb and reduce nitrogen pollution acting at different points of the cycle. ENA provided a first assessment of the magnitude of nutrient fluxes from/to all sectors and environmental compartments at the continental scale (ENA, Sutton et al. 2011).

Knowledge on the nutrient fluxes originated from different sources and flowing into different environmental compartments is key for understanding the level of disruption of the natural N and P cycles and for planning measures to reduce nutrient pollution while preserving soil fertility. In this study data on nutrient fluxes were collected from different sources, including both reported data and modelled data, with the aim of estimating all major fluxes in the N and P cycles. The list of data and respective references are shown in Table 1.

Based on these data, we provide a quantification of the major fluxes in the N and P cycles at the EU27 level for a period centred in year 2015. Values were also computed for each EU country (Annex A1). For an analysis of the agro-food system in Europe see also Section 4.7.

Table 1. List of datasets used in this study.

Data Source	Data type	Format of data	Unit of measure	Reference year	Spatial coverage	Ref Sys (EPSG)	Description / Source / Citation
EDGAR Database	Map of N_2O emissions	netcdf, 0.1 x 0.1 deg	kg/sqm/s	2015	global	4326	The Emissions Database for Global Atmospheric Research (EDGAR) is an independent global emission inventory of greenhouse gases (GHG) and air pollutants developed by the Joint Research Centre of the European Commission. The non-CO_2 components in EDGARv5.0 cover a long time series of emissions for the period 1970-2015; emissions are estimated for all anthropogenic emission sectors with the exception of Land Use, Land Use Change and Forestry (LULUCF) at country and annual level in a consistent and comparable way for all world countries. Regarding Nitrogen, EDGAR provides N_2O, NO_x and NH_3 emissions (unit: kt). Crippa, M., Oreggioni, G., Guizzardi, D., Muntean, M., Schaaf, E., Lo Vullo, E., Solazzo, E., Monforti-Ferrario, F., Olivier, J. and Vignati, E., Fossil CO_2 and GHG emissions of all world countries, EUR 29849 EN, Publications Office of the European Union, Luxembourg, 2019, ISBN 978-92-76-11100-9 (online),978-92-76-11025-5 (print), doi:10.2760/687800 https://data.europa.eu/doi/10.2904/JRC_DATASET_EDGAR https://edgar.jrc.ec.europa.eu/overview.php?v=50_GHG https://edgar.jrc.ec.europa.eu/overview.php?v=50_AP
	Map of NO_x emissions						
	Map of NH_3 emissions						
	IPCC2006 total N_2O, NO_x, NH_3 emissions by sector and Country	table	Gg	2015	EU28 (by country)		
EMEP Model	Total N deposition	Netcdf, 0.1 x 0.1 deg	mg/sqm/y	2015	Physical Europe	4326	The co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe (unofficially 'European Monitoring and Evaluation Programme' = EMEP) is a scientifically based and policy driven programme under the Convention on Long-range Transboundary Air Pollution (CLRTAP) for international co-operation to solve transboundary air pollution problems. Inside this programme, the ‘EMEP open-source' model has been developed, to simulate air quality and deposition due to emission reduction policies. The EMEP model, for this project, has been run at the Joint Research Centre.

DayCent Model	Sinthetic and organic fertilization, biological N fixation (plant+soil), N deposition (from EMEP), N export from grain, N export from above ground biomass, N in NO3 leaching, N in DOC (organic N leaching), N2O, NO, N2, N loss by erosion	tiff (Raster dataset with 12 bands), 1000 x 1000 m	kg/ha/y	Average value 2010-2019	Physical Europe	3035	JRC-D.3 full N budget in agricultural soils by DayCent model Quemada, M., Lassaletta, L., Leip, A., Jones, A., and Lugato, E. Integrated management for sustainable cropping systems: Looking beyond the greenhouse balance at the field scale. Global Change Biology, 2020a, 14989, https://doi.org/10.1111/gcb.14989 Lugato, E., Leip, A., and Jones, A. Mitigation potential of soil carbon management overestimated by neglecting N2O emissions. Nature Clim Change, 8, 2018a, 219–223, https://doi.org/10.1038/s41558-018-0087-z Lugato, E., Paniagua, L., Jones, A., de Vries, W., and Leip, A., Complementing the topsoil information of the Land Use/Land Cover Area Frame Survey (LUCAS) with modelled N2O emissions. PLoS ONE, 12, 2017, e0176111. https://doi.org/10.1371/journal.pone.0176111
Domestic emissions to water	TN and TP	Vector data, catchment level	t/y	2016	Physical Europe	3035	N and P domestic waste emissions to waters (tN/y; tP/y) from UWWTP, Sewage discharges, Individual Appropriate Systems (IAS), scattered dwellings Vigiak, O., Grizzetti, B., Zanni, M., Aloe, A., Dorati, C., Bouraoui, F., Pistocchi, A., 2020. Domestic waste emissions to European waters in the 2010s. Sci. Data 7. https://doi.org/10.1038/s41597-020-0367-0 (updated to 2016 data) https://data.jrc.ec.europa.eu/dataset/0ae64ac2-64da-4c5e-8bab-ce928897c1fb
GREEN Model	TN and TP diffuse emissions		kg/ha	2014-2018			Vigiak, O., Udías, A., Grizzetti, B., Zanni, M., Aloe, A., Weiss, F., Hristov, J., Bisselink, B., de Roo, A., Pistocchi, A., Recent regional changes in nutrient fluxes of European surface waters, Science of the Total Environment, 858, 2023, 160063, http://dx.doi.org/10.1016/j.scitotenv.2022.160063 Grizzetti, B., Vigiak, O., Udias, A., Aloe, A., Zanni, M., Bouraoui, F., Pistocchi, A., Dorati, C., Friedland, R., De Roo, A., Benitez Sanz, C., Leip, A., and Bielza, M., How EU policies could reduce nutrient pollution in European inland and coastal waters, Global Environmental Change, 69, 2021, 102281, https://doi.org/10.1016/j.gloenvcha.2021.102281
	TN and TP loads to the seas		t/y				

Global P losses due to soil erosion		tiff, 0.5 x 0.5 deg	kg/ha/y	2015	Global	4326	Alewell, C., Ringeval, B., Ballabio, C., Robinson, D.A., Panagos, P., Borrelli, P. 2020. Global phosphorus shortage will be aggravated by soil erosion. Nat Commun 11, 4546. https://doi.org/10.1038/s41467-020-18326-7 https://esdac.jrc.ec.europa.eu/content/global-phosphorus-losses-due-soil-erosion
Mean phosphorus stock (LUCAS points)		tiff, 500 x 500m	mg/kg and kg/ha	2009	EU28	3035	Ballabio, C., Lugato, E., Fernández-Ugalde, O., Orgiazzi, A., Jones, A., Borrelli, P., Montanarella, L. and Panagos, P., 2019. Mapping LUCAS topsoil chemical properties at European scale using Gaussian process regression. Geoderma, 355: 113912. https://esdac.jrc.ec.europa.eu/content/chemical-properties-european-scale-based-lucas-topsoil-data
Phosphorus Plant removal		NUTS2	kg/ha	2016	EU28	3035	Panagos, P., Muntwyler, A., Liakos, L., Borrelli, P., Biavetti, I., Bogonos, M. and Lugato, E., 2022. Phosphorus plant removal from European agricultural land. Journal of Consumer Protection and Food Safety, DOI: 10.1007/s00003-022-01363- https://esdac.jrc.ec.europa.eu/content/phosphorus-plant-removal
CAPRI Model	N and P Mineral fertiliser, Manure applied to soil, N Crop fixation, Crop uptake, Surplus		kg/ha	2014	EU27		Barreiro-Hurle, J., Bogonos, M., Himics, M., Hristov, J., Pérez-Domiguez, I., Sahoo, A., Salputra, G., Weiss, F., Baldoni, E., Elleby, C. Modelling environmental and climate ambition in the agricultural sector with the CAPRI model. Exploring the potential effects of selected Farm to Fork and Biodiversity strategies targets in the framework of the 2030 Climate targets and the post 2020 Common Agricultural Policy, EUR 30317 EN, Publications Office of the European Union, Luxembourg, 2021, ISBN 978-92-76-20889-1, doi:10.2760/98160, JRC121368.
Nutrients (N, P) in sewage sludge applied to land		table	kg/ha		EU27		Based on EUROSTAT statistics (sludge volumes, aggregated at MS level), spatial locations of waste water treatment plants and estimates of nutrient contents in sewage sludge, we will be estimating sewage sludge applications to agricultural land at a small spatial scale (to be defined, but likely around 5 x 5 km) (work done in cooperation with Alberto Pistocchi, and D.3) (units: kg N/P ha-1 yr-1)

Food waste	N and P amounts in food waste at processing, distribution, household, food services, wastewater collection, treatment and waste management	table	kt N/y and kt P/y	2015	EU27		Corrado S., Caldeira C., Carmona-Garcia G., Körner I., Leip A., Sala S.. Unveiling the potential for an efficient use of nitrogen along the food supply and consumption chain. Global Food Security 25, 100368 https://doi.org/10.1016/j.gfs.2020.100368; Caldeira, C., De Laurentiis, V., Ghoose, A., Corrado, S., Sala, S. 2021. Grown and thrown: Exploring approaches to estimate food waste in EU countries. Resources, Conservation and Recycling, 168, p. 105426 https://doi.org/10.1016/j.resconrec.2021.105426 De Laurentiis, V., Caldeira, C., Sala, S., Building a balancing system for food waste accounting at National Level, EUR 30685 EN, Publications Office of the European Union, Luxembourg, 2021, ISBN 978-92-76-37275-2, doi:10.2760/316306, JRC124446

2.1.1 Major fluxes in the N cycle

Based on models and data listed in Table 1, the main flows of reactive nitrogen between land, air, and water are summarized in the Sankey Diagram of Figure 1. The dimension of fluxes represented in the diagram are proportional to flux volumes (nitrogen quantities are reported in Table 2). The conceptualization of fluxes reflects as close as possible the INMAP pollution compartments, i.e. air, water and soil, but with some exceptions. Indeed, in the diagram primary production (including cropping systems, pastures, livestock, and forests) is merged with soil into a single node called LAND, which was so named to remark the inclusion of primary activities directly sustained by soils. Hence, in this representation, manure is a product of LAND (from livestock) that returns to LAND as soil fertilization. Future work will aim at disentangling primary production from soils. The FOOD node includes all activities related to FOOD processing, distribution and consumption, i.e. from the farm gate to the households, from and to territories OUTSIDE the EU27, and to WASTE (The estimation of these fluxes in presented in Section 2.2).

Nitrogen fluxes were quantified with models that differ for spatial extent, i.e. some covering the whole territory others only agricultural land, and temporal coverage (Table 1). Thus, stocks in each compartments in terms of net accumulation or depletion are not fully quantified. Nevertheless, the Sankey diagram allows identifying the major fluxes, and thus helps focusing measures to reduce pollution.

Figure 1. Nitrogen flows in EU27 around 2015-2020. Data sources reported in the text.

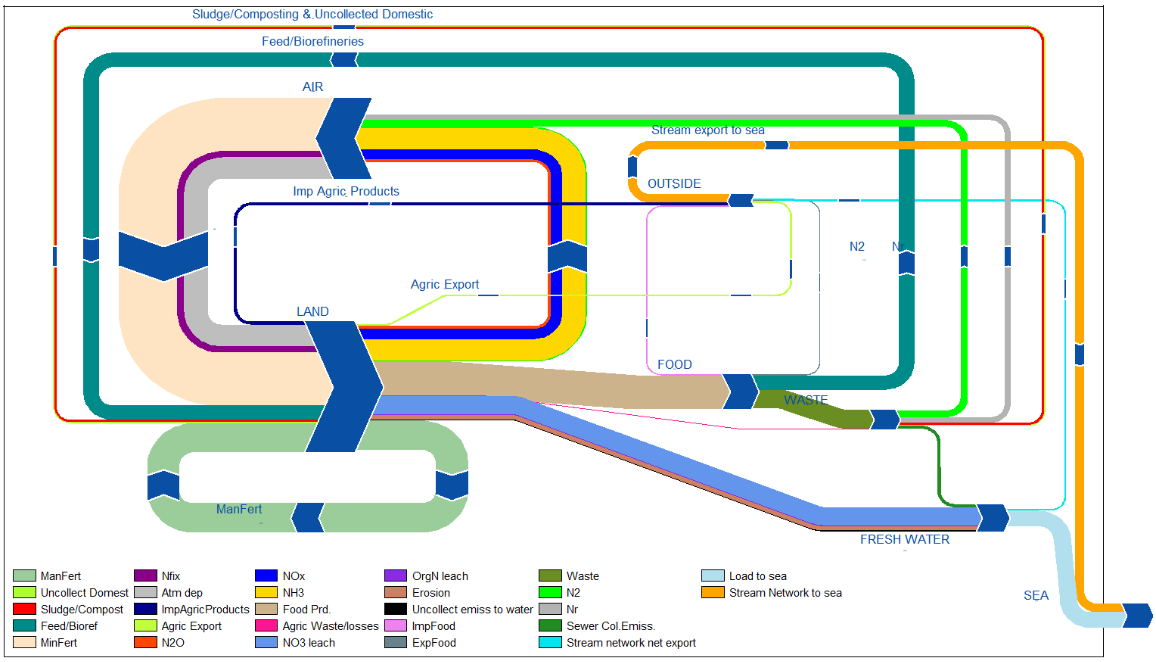


Table 2. Assessed contemporary nitrogen fluxes (ktN/y) in EU27.

FROM Node	TO Node	Nitrogen (ktN/y)	Flux	Data source and spatial extent
LAND	AIR	521	N2O (1)	EDGAR, EU27
LAND	AIR	1996	NOx (1)	
LAND	AIR	4179	NH3 (1)	
LAND	FRESHWATER	3543	NO3 leaching	DAYCENT; agricultural land
LAND	FRESHWATER	70	OrgN leaching	
LAND	FRESHWATER	822	Losses in soil erosion	
AIR	LAND	10344	Mineral fertilization	CAPRI; agricultural land
LAND	LAND	5707	Manure fertilization	
AIR	LAND	1256	Plant N fixation	
AIR	LAND	4210	Atmospheric deposition	EMEP; EU27
FOOD	LAND	2789	Feed and biorefineries	Food/waste Cycle; EU27
FOOD	OUTSIDE	148	Food export	
FOOD	WASTE	3691	Waste	
LAND	FOOD	6571	Agricultural food production	
LAND	OUTSIDE	376	Export of agricultural products	
LAND	WASTE	203	Agriculture waste/losses	
OUTSIDE	FOOD	56	Food import	
OUTSIDE	LAND	604	Import of agricultural products	
WASTE	AIR	1335	Emissions of N2(2)	
WASTE	AIR	1077	Emissions of N-reactive(2)	
WASTE	LAND	445	Sludge and composting(2)	
WASTE	LAND	131	Uncollected domestic emissions	Domestic emissions
WASTE	FRESHWATER	605	Collected domestic emissions to water	
FRESHWATER	OUTSIDE	334	Stream network net export	GREEN; Europe
FRESHWATER	SEA	3053	Load to sea	
OUTSIDE	SEA	1627	Stream network load to sea	
LAND	FRESHWATER	87	Quota of uncollected emissions reaching stream network	

(1) These fluxes exclude emissions from WASTE management.
(2) These fluxes consider emissions from WASTE management of food cycle, less emissions to freshwater (estimated in Domestic emissions as in Vigiak et al., 2020).

2.1.2 Major fluxes in the P cycle

EU27 phosphorus fluxes are represented in Figure 2. The diagram is somehow simpler as the compartment of AIR is absent due to the phosphorus cycle characteristics. Instead, a node ROCK has been added to represent depletion of geological resources of phosphorus for production of mineral fertilizers. Phosphorus fluxes are reported in Table 3.

Figure 2. Phosphorus flows in EU27 around 2015-2020. Data sources reported in the text.

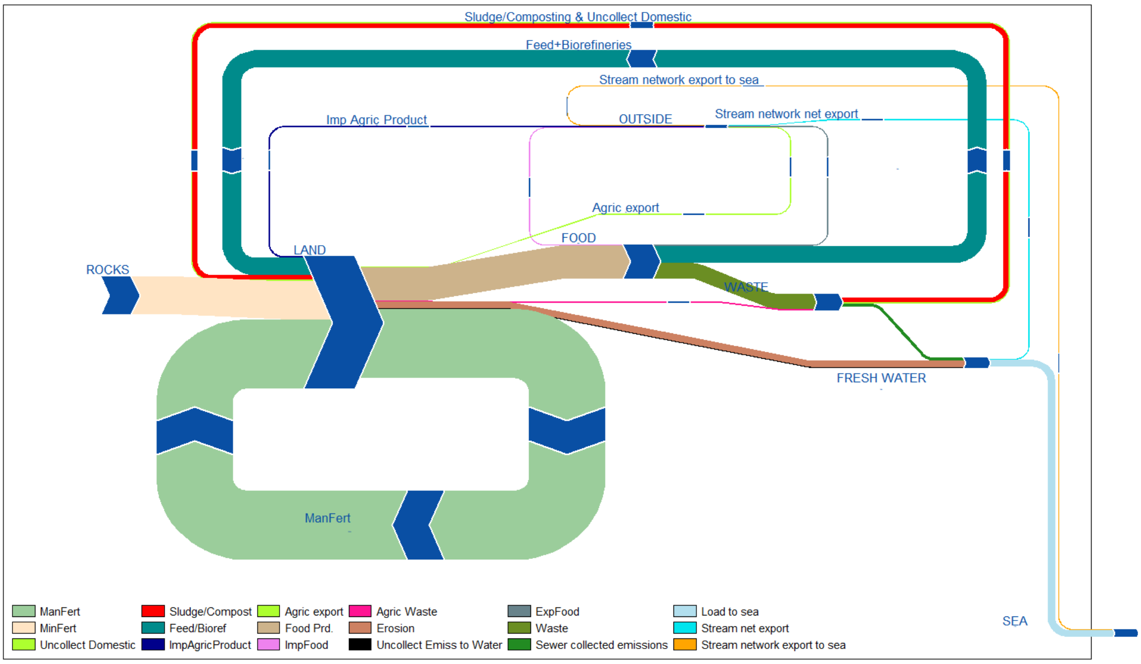


Table 3. Assessed contemporary phosphorus fluxes (ktP/y) in EU27.

FROM Node	TO Node	Phosphorus (ktP/y)	Flux	Data source and spatial extent
ROCKS	LAND	1120	Mineral fertilization	CAPRI; agricultural land
LAND	LAND	2012	Manure fertilization	
LAND	FRESHWATER	183	Losses in soil erosion	Global P losses; global
LAND	FOOD	962	Agricultural food production	Food/waste Cycle; EU27
LAND	OUTSIDE	27	Export of agricultural products	
LAND	WASTE	31	Agricultural waste/losses	
OUTSIDE	FOOD	7	Food import	
OUTSIDE	LAND	43	Import of agricultural products	
FOOD	OUTSIDE	21	Food export	
FOOD	LAND	501	Feed and biorefineries	
FOOD	WASTE	448	Waste	
WASTE	LAND	138	Sludge and composting	
WASTE	FRESHWATER	60	Collected domestic emissions to water	Domestic emissions
WASTE	LAND	18	Uncollected domestic emissions	
LAND	FRESHWATER	13	Quota of uncollected emissions reaching stream network	GREEN; Europe
FRESHWATER	SEA	200	Load to sea	
FRESHWATER	OUTSIDE	7	Net export through stream network	
OUTSIDE	SEA	143	Export through stream network	

2.1.3 Data limitations and knowledge gaps

This study provides a not exclusive compilation of nutrient fluxes based on recent reported or modelled data from various sources (Table 1) to describe the current N and P cycles in EU27 (Figure 1, 2, Tables 2, 3). Major knowledge gaps in the quantification of N and P cycles concern the legacy and buildup of N in groundwater and of P in soil. These stocks are not represented. The quantification of nutrient fluxes is limited by the data availability, their spatial and temporal resolution, and the consistency in datasets and assumptions of the different modelling assessments. To provide a qualitative understanding of the data variability and uncertainty we compared the nutrient fluxes (Sankey's diagrams, Figures 1 and 2) with values reported in the ENA (Sutton et al. 2011), and with the independent analysis performed by the GRAFS model (Billen et al. in preparation, Section 4.7). In addition, we compared nutrient fluxes in agriculture from several data sources/assessments adopted in the study. Quantifying nutrient cycles and comparing different reported and modelled fluxes provide insight on data variability and knowledge gaps.

The comparison between nitrogen flows estimated in this study, ENA and GRAFS model (Table 4) is only qualitative, as the three assessments differ in the approach (how compartments and processes are represented), data sources, time period of analysis (year 2000 in ENA; around year 2015 in the other approaches), and spatial extent (this study: EU27 after Brexit, as from February 2020, ~4.2 10^6 km^2; ENA: EU27 before Croatia joined, as in 2007- 2013, ~4.4 10^6 km^2; and GRAFS: EU27 plus UK, Norway, Switzerland and Balkans, ~4.9 10^6 km^2).

N emissions to air, estimated by EDGAR, are higher in this study compared to GRAFS as the latter considers only N emissions from the agricultural system, while industries and traffics are also included here. The estimation of N loads to European seas appear coherent between the assessments, while there is some variability in the estimation of nitrogen leaching to groundwater, although in the same order of magnitude (there are also differences in the agricultural areas considered). N input in the agricultural system by mineral fertilisers, manure and biofixation are almost coherent across the three assessments. Similarly, N inputs from atmospheric deposition are consistent, also considering that GRAFS provides N deposition only in agricultural land, while the in the other cases the figures refer to the whole surface. The estimation of N fluxes from domestic waste in the present study is net of N removed by wastewater treatments, therefore lower than the values in ENA and GRAFS, which report N fluxes from domestic waste before treatment. The comparison of N fluxes in the food production system is not straightforward across the three assessments, as the material flow analysis presented in this study considers only N entering the food processing system, while in ENA and GRAFS assessments report data on the whole agricultural production system (food+feed).

Table 4. Comparison of nitrogen fluxes estimated in this study (Figure 1 and Table 2), in the European Nitrogen Assessment (ENA, Sutton et al. 2011) and GRAFS model (Billen et al. Section 4.7). Explanation in the text.

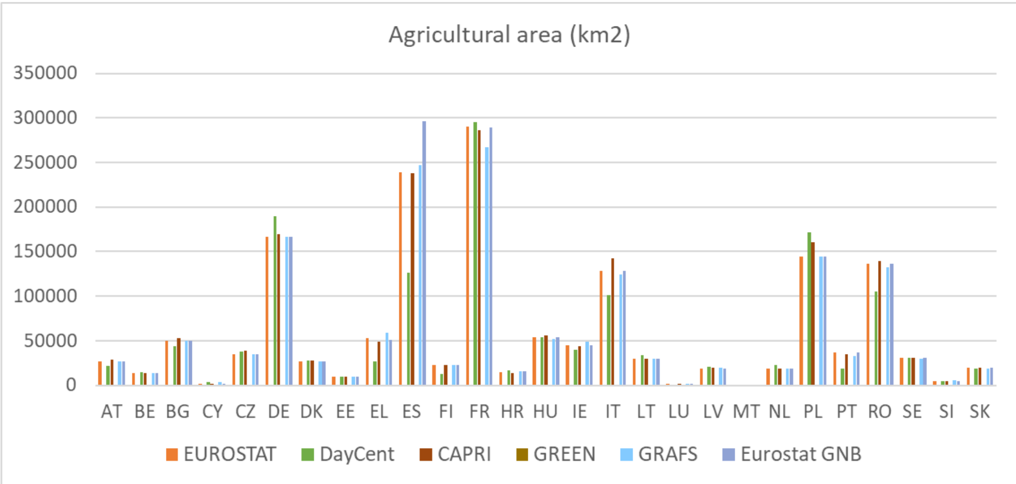
From NODE	To NODE	Flux (This study, Table 2)	This study (TgN/y)	ENA (TgN/y)	GRAFS (TgN/y)
LAND	AIR	N2O (except waste)	0.5		0.4
LAND	AIR	NOx (except waste)	2.0	3.4	
LAND	AIR	NH3 (except waste)	4.2	3.8	3.0
FRESHWATER	SEA	Load to sea	3.1	4.5	
LAND	FRESHWATER	NO3 leaching	3.5	6.2	5.9
LAND	FRESHWATER	OrgN leaching	0.1		
LAND	FRESHWATER	Losses in soil erosion	0.8		
WASTE	FRESHWATER	Collected domestic emissions to water	0.6	1.8	3.2
AIR	LAND	Mineral fertilization	10.3	11.2	12.3
LAND	LAND	Manure fertilization	5.7	8	5.6
AIR	LAND	Plant N fixation	1.3	1	4.1
AIR	LAND	Atmospheric deposition	4.2	3.8	2.9
FOOD	LAND	Feed and biorefineries	2.8		
FOOD	OUTSIDE	Food export	0.1	0.4	2.6
FOOD	WASTE	Waste	3.7	3.6	
LAND	FOOD	Agricultural food production	6.6	4.3	8.8
LAND	OUTSIDE	Export of agricultural products	0.4		18
LAND	WASTE	Agricultural waste/losses	0.2		
OUTSIDE	FOOD	Food import	0.1	0.4	
OUTSIDE	LAND	Import of agricultural products	0.6		
WASTE	AIR	Emissions of N2 (waste treatment)	1.3	0.6	
WASTE	AIR	Emissions of N-reactive (waste treatment)	1.1		
WASTE	LAND	Sludge and composting	0.4	0.1	
FRESHWATER	OUTSIDE	Stream network net export	0.3		

OUTSIDE	SEA	Stream network load to sea	1.6		
WASTE	LAND	Uncollected domestic emissions	0.1		
LAND	FRESHWATER	Quota of uncollected emissions reaching stream network	0.1		

Concerning nutrient fluxes in agriculture, several data sources were available and compared per EU27 countries, including data from EUROSTAT and FAOSTAT, which are official statistics, and datasets used in this study for the modelling assessments, which include assumptions on data spatialisation and gap filling. In specific, we considered nutrient input data from the modelling analysis of DayCent, CAPRI, GREEN, P model and GRAFS presented in Section 4 for the period 2014-2018 (data from DayCent refer to the period 2010-2019). The comparison highlights when similar datasets and assumptions have been adopted.

Nutrient fluxes depend on the extent of the agricultural area. The latter is consistent for most of the modelling assessments with Eurostat statistics. In DayCent the agricultural area is based on the spatial information of the Corine Land Cover map. Values are close to Eurostat statistics except for some countries. The Coefficient of Variation (CV) for EU values is CV=5% (Figure 3).

Figure 3. Comparison of agricultural area (km^2) per EU27 countries according to different data sources and modelling approaches. Data refer to average 2014-2018 (except DayCent 2010-2019).



Overall, the estimation of N input to the agricultural system by mineral fertilisers is very coherent across different data sources and modelling approaches in this study (CV=9%) (Figure 4), with differences mainly related to the surface of agricultural land considered. The assessment of N input by manure presents more variability (CV=18%), which could be related to the excretion coefficients (per animal types) adopted or to assumptions on NH_3 emissions (Figure 4). Similarly, the estimation of N input from biological fixation varies in the different approaches (CV=63%), although this flux is lower in terms of magnitude compared to mineral and manure inputs (Figure 4).

With regard to P fluxes, the quantification of mineral fertiliser input seems consistent in the data sources, except for few higher values in CAPRI and GREEN in some countries (Germany, France, Poland) and a striking higher value of the P model for Italy (CV=5%). As for N, also the estimation of P input in manure is quite variable (CV=29%), note that in many cases it is estimated from N manure input by a fix N:P ratio (Figure 5).

The fact that several data sources and assessment have similar values of nutrient flows suggest a certain coherence in the underpinning datasets and assumptions, but does not necessarily indicate that they are more correct than the others. (For example the values reported for the model GREEN in the present application are based on the data from the CAPRI model, plus assumption on the spatial distribution of the input in the river basin). Nevertheless, these comparisons (Figures 4 and 5) help understanding the range of variability of nutrient agricultural fluxes we are confronted with for EU27, and how this could be reflected in the modelling scenario analysis.

Figure 4. Comparison of nitrogen agricultural inputs: synthetic fertiliser (above), manure (centre) and biological fixation (below), per EU27 countries, according to different data sources and modelling approaches. Data refer to average 2014-2018 (except DayCent 2010-2019).

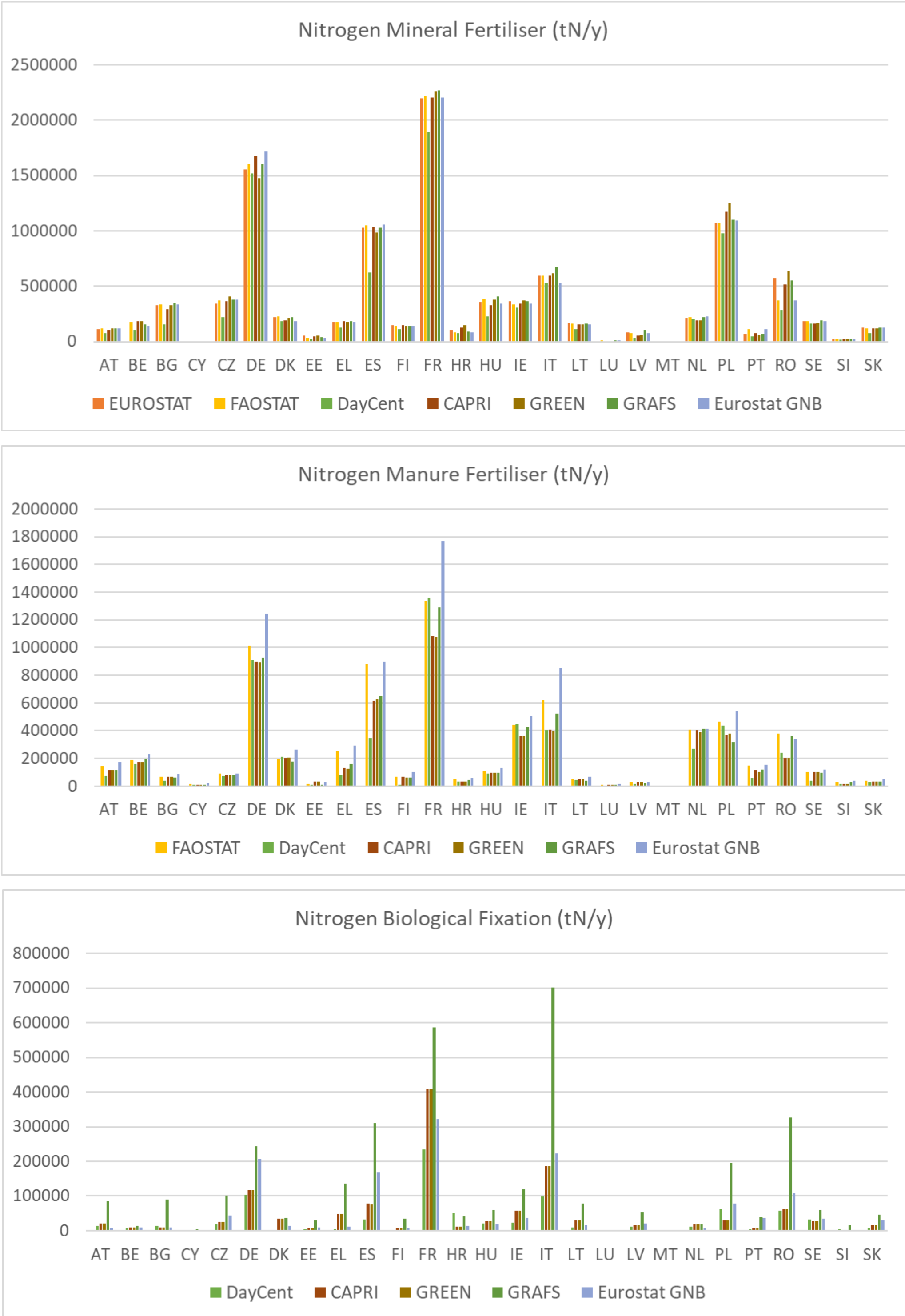
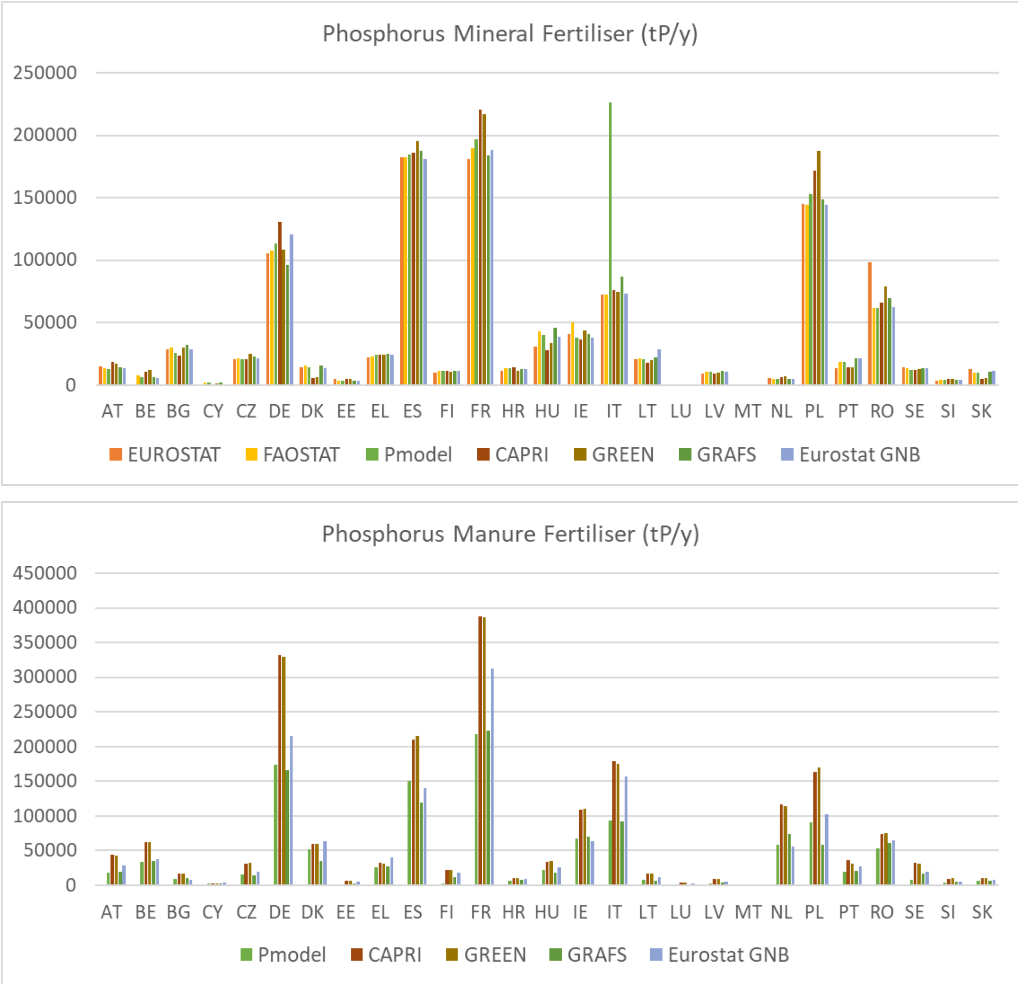


Figure 5. Comparison of phosphorus agricultural inputs, synthetic fertiliser (above) and manure (below), per EU27 countries, according to different data sources and modelling approaches. Data refer to average 2014-2018 (except DayCent 2010-2019).



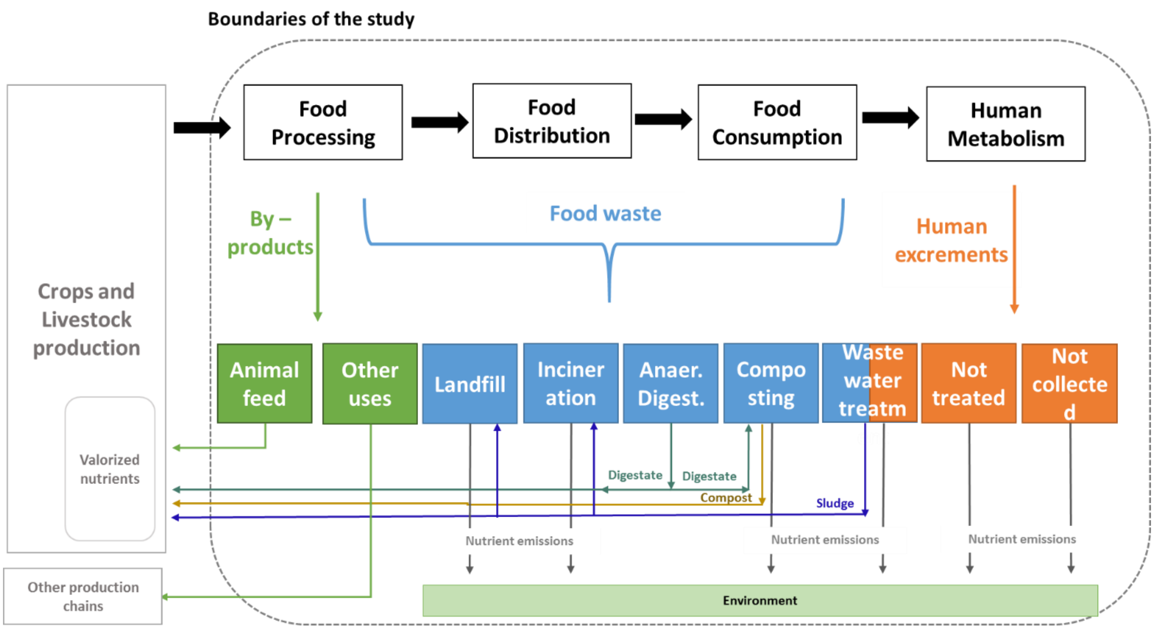
2.2 Flow material analysis of nutrient in the food system, including food waste

The model developed by Corrado et al. (2020) to quantify nitrogen flows along the European food chain from food processing up to waste treatment, was further developed to include the quantification of P flows and downscaled to quantify N and P flows in the food system at Member State (MS) level. The model has been applied for all years between 2002 and 2017 for the EU27 MSs. In this report, results are presented for 2015.

The nutrient content in food products, food waste, by-products, human excrements, and losses to the environment of these flows were quantified (Figure 6). Food waste is defined according to EU legislation as: "all food as defined in Article 2 of Regulation (EC) No 178/2002 of the European Parliament and of the Council (EC, 2002) that has become waste" (EC, 2018). The definition of 'food' laid down in Regulation (EC) No 178/2002 encompasses food as a whole, along the entire food supply chain from production until consumption, and includes inedible parts, where those were not separated from the edible parts when the food was produced, such as bones attached to meat destined for human consumption. 'Waste' means any substance or object which the holder discards or intends or is required to discard (European Parliament and Council, 2008). By-products are defined as surplus food used as animal feed and for non-food uses such as the production of biobased materials. The study system boundaries include post-farm gate stages of the food chain, i.e. processing, distribution, and consumption, and human metabolism (intended as the processes of human digestion of food and excretion of residues) as well as waste management.

In the following sections, we describe briefly how the nutrient content in food products, by-products, food waste, and human excrements (section 2.2.1) and the nutrient losses to the environment from the different destinations (section 2.2.2) were calculated. More information on the model and approach are provided in Corrado et al. (2020) and in Annex A2.

Figure 6. Boundaries of the study, flows and destinations considered. Adapted from Corrado et al. (2020).



2.2.1 Nutrient content in food products, food waste, and human excrements

Quantities of consumed food, food waste, and by-products at each stage of the EU food chain for each MS were obtained from the food waste model developed by Caldeira et al. (2021). The calculation of the nutrient content in food products and food waste is described below separately for N and P.

2.2.1.1 Nitrogen content in food products and food waste

The nitrogen content in food products was calculated according to Corrado et al. (2020) considering (i) crude protein content in food products, food waste, and by-products; (ii) conversion factors between protein content and nitrogen content (Jones' factors; Jones, 1941); and (iii) food consumption amounts.

The N content in food products and food waste calculated is presented in Table 5.

Table 5. N content in food products food waste, and by-products based on Corrado et al. (2020). PP: Primary Production; P&M: Processing and Manufacturing; R&D: Retail and Distribution; Cons: Consumption (household and food service).

N content in food waste and by-products (g N/kg waste or by-product)										
	Meat	**Fish**	**Dairy**	**Eggs**	**Cereal**	**Fruits**	**Vegetable**	**Potato**	**Sugar**	**Oilcrop**
PP	23.8	33.1	5.2	19.2	18.8	1.2	3.2	3.4	2.3	37.5
P&M	36.7	33.1	5.6	20.9	27.4	2.0	3.3	3.4	9.1	52.2
R&D	32.7	33.1	5.5	19.2	10.2	1.2	3.6	5.0	2.2	34.1
Cons	32.7	33.1	10.2	20.9	10.2	1.7	3.5	5.0	1.6	3.6
N content in food product (g N/kg product)										
PP	25.9	33.1	5.2	19.2	18.8	1.2	3.2	3.4	2.3	37.5
P&M	32.7	33.1	10.2	19.2	10.2	0.9	3.5	5.0	1.6	3.6
R&D	32.7	33.1	10.2	19.2	10.2	0.9	3.5	5.0	1.6	3.6
Cons	32.7	33.1	10.2	19.1	10.2	0.8	3.5	5.0	1.6	3.6

2.2.1.2 Phosphorus content in food products and food waste

The phosphorus content in food products, in food waste, and in by-products was calculated following a similar approach as for nitrogen. The conversion factors protein content/ phosphorous content (equivalent to Jones' Factors for Nitrogen) were obtained from USDA (2018). The P content was calculated in the same manner as for N for dairy, eggs, fruits, vegetables, potatoes, sugarbeet and oilcrops. P content in fish was derived from FAO/INFOOD (FAO, 2016) assuming the same P content in food product and in food waste across supply chain stages. For cereals, protein content of whole cereals was used for the food product and protein content in bran cereals was used for food waste. For meat, approach and data were adapted from van Dijk et al. (2016), allowing to quantify the P content in meat products (generally muscles), by-products, and waste (generally bones). Table 6 presents the P content in food products, food waste, and by-products.

Table 6. P content in food products, food waste, and by-products based on Corrado et al. (2020). PP: Primary Production; P&M: Processing and Manufacturing; R&D: Retail and Distribution; Cons: Consumption (household and food service).

P content in food waste and by-products (g P/kg waste or by-product)										
	Meat	**Fish**	**Dairy**	**Eggs**	**Cereal**	**Fruits**	**Vegetable**	**Potato**	**Sugar**	**Oilcrop**
PP	6.0	2.4	0.9	1.9	3.9	0.2	0.5	0.3	0.5	5.6
P&M	8.9	2.4	1.1	2.1	3.9	0.3	0.5	0.3	1.9	7.5
R&D	1.7	2.4	1.0	1.9	2.0	0.2	0.6	0.5	0.4	5.3
Cons	1.7	2.4	1.6	2.1	2.0	0.3	0.5	0.5	0.3	0.7
P content in food product (g P/kg product)										
PP	6.0	2.4	0.9	1.9	5.9	0.2	0.5	0.3	0.5	5.6
P&M	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7
R&D	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7
Cons	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7

2.2.2 Quantification of nutrient losses to the environment

The nutrient losses to the environment comprised those occurring in the different destinations of the food waste (i.e. landfill, incineration, anaerobic digestion, composting and wastewater treatment) and human excrements (i.e. wastewater treatment, not treated and not collected). Nutrient content in sludge from wastewater treatment that are then directed to landfill and incineration, from where additional losses occur, are also considered. The destinations for food waste and human excrements are those considered in Corrado et al. (2020). However, coefficients for waste and sludge were updated using country-specific values for each MS and for each year, based on Corrado et al. (2020), data from Eurostat (Eurostat, 2021a; Eurostat, 2021b) and from EEA (European Environmental Agency; EEA, 2021). Missing data from Eurostat and EEA were replaced with the value from the last available year. This approach was preferred in order to make use of existing data, instead of using interpolations.

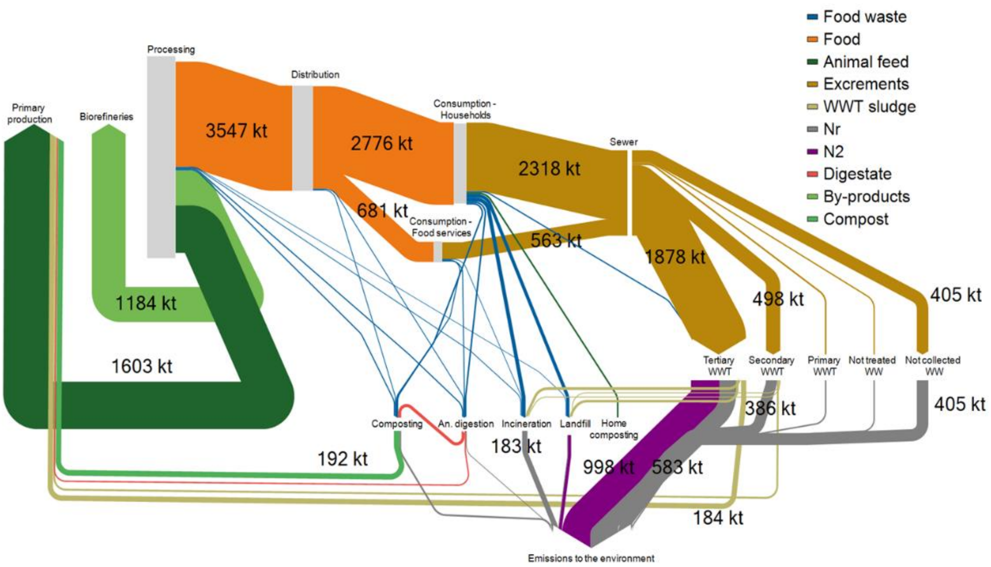
2.2.2.1 Nitrogen losses to the environment

N emissions to the environment from the different waste treatments were quantified. Emissions of N can occur as non-reactive, i.e. molecular N (N2), not harmful for the environment, or reactive N (Nr), including nitrates, ammonia, and nitrogen oxides, responsible for various pollution phenomena. The remaining N, i.e. the share that is neither emitted as Nr nor as N2, remains in the food waste or in the output of the waste treatments. This flow can be valorised as input either to the food chain or to other productive chains. The quantification of N emissions was done according to Corrado et al. (2020) as follow:

1. N emissions from composting: average reactive N factors from composting, also when preceded by anaerobic digestion, were assumed as 23.2% of initial N for ammonia and N2O (being the later a minor share of about 0.5-1%) and 0.3% for N leaching (median values reported by Körner, 2009). For home composting the shares assumed were 59.5% N2 and 5.2% Nr (Andersen et al 2011). It was assumed that N captured in ventilation air purification systems is not recycled.
2. N emissions from incineration: a share of 30% emitted as Nr and 70% emitted as N2 was based on weighted average emissions from DeNOx (Corrado et al. (2020; Doka G., 2003). A share of 2 % of N was considered to be present in the residues (i.e. ash) from the incineration and assumed to be disposed in landfills.

3. N emissions from landfills: N losses from landfills are highly variable and depend on different elements, such as the type of waste and the climatic conditions. Emissions from landfills are assessed at 47.4% of available N, thereof 39.2% as N2 and 8.2% as Nr, whereas the remaining 52.6% stays in the body of the landfill.
4. N emissions from incinerated or landfilled sewage sludge and ash: the N emission factors abovementioned for incineration and landfill were used.
5. N emissions from wastewater treatment: for primary and secondary treatment we assumed nitrogen removal efficiencies of 10% and 25%, respectively (Bonomo, 2008). For tertiary wastewater treatment technologies, it was considered 30% of N emitted as Nr, and 51% emitted as N2 (McCarty, 2018). For all the treatment, the share of N not emitted to the environment stays in the wastewater sludge.
6. A summary of coefficients for N losses to the environment in the different destinations is presented in annex A2, table 8. Figure 7 shows the nitrogen fluxes resulting from the model application for EU27 in 2015.

Figure 7. Sankey diagram for N flows in the EU27 food system for the year 2015. Small fluxes are not represented in the Sankey for presenting purposes.



In 2015, at EU level, the share of nitrogen in food waste was 12.7%. Regarding the losses to the environment, these were 49.3% of the total amount of nutrients available in the food chain. Table 7 presents these shares in each Member State for 2015.

Table 7. Shares of N in food waste and lost to the environment in the EU and for each 27MS in 2015.

Member State	Share of N in food waste (%)	Share of N lost to the environment (%)
Austria	11.6	38.6
Belgium	9.9	44.3
Bulgaria	7.3	32.1
Croatia	13.7	54.7
Cyprus	12.7	49.9
Czech Republic	11.3	44.7
Denmark	13.8	54.0
Estonia	11.6	42.8
Finland	12.4	56.1
France	11.4	46.9
Germany	10.8	47.2
Greece	11.1	44.2
Hungary	8.5	32.4
Ireland	14.7	45.1
Italy	12.0	43.1
Latvia	13.9	47.5
Lithuania	11.6	41.7
Luxembourg	11.1	52.7
Malta	10.9	47.3
Netherlands	9.9	39.5
Poland	14.2	52.0
Portugal	11.8	40.2
Romania	12.4	50.3
Slovakia	10.1	42.5
Slovenia	12.7	47.4
Spain	11.0	39.2
Sweden	11.3	49.4
European Union	**12.7**	**49.3**

2.2.2.2 Phosphorus losses to the environment

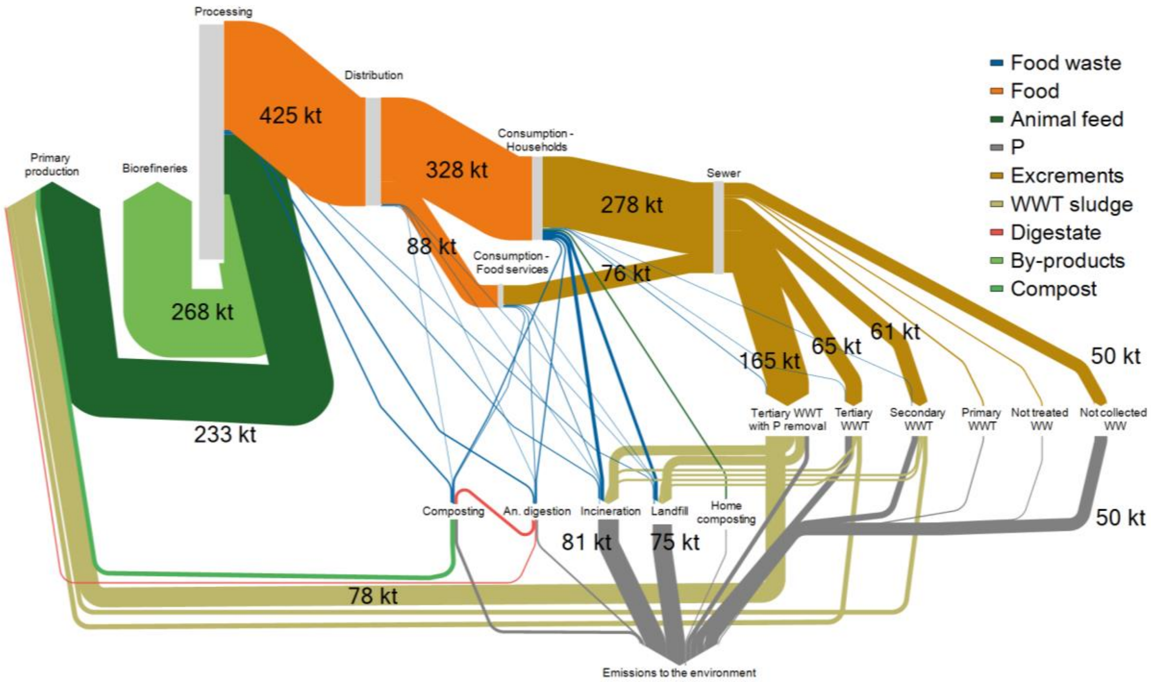
Regarding P losses from waste treatments, no significant losses to air take place. The emissions were assumed as follows:

1. *P emissions from composting*: composting is subject to nutrient losses as not all is recirculated on agricultural land. Compost is used also in other non-agricultural applications, such as backfilling and mine sites rehabilitation and landfill (as cover material). On average, 40% of the phosphorous in compost is lost (Barth, 2006; EC, 2008; van Dijk et al., 2016). During the anaerobic digestion process and when handling its residues, it was assumed a 1% P loss. It is mostly directly applied in agriculture.
2. *P emissions from incineration*: all P in waste incinerated and landfilled is assumed to be removed from the biogeochemical cycle. Ashes are mostly used as construction materials, because technologies that recover P from (sewage sludge) ashes (Tonini et al., 2019) are not yet being implemented at a relevant scale.

3. *P emissions from landfills*: Current technologies do not enable the return of landfilled P into the biogeochemical cycle. Therefore, a 100% loss is assumed for both treatment destinations.
4. *P emissions from wastewater treatment*: P losses from wastewater treatment were calculated as the complementary of the removal efficiency, as reported in Vigiak et al. (2020).

A summary of coefficients for P losses to the environment in the different destinations is presented in annex A2, Table 8. Figure 8 shows the phosphorus flow material Sankey diagram for Food waste cycle for EU27 in 2015.

Figure 8. Sankey diagram for P flows in the EU27 food system for the year 2015. Small fluxes are not represented in the Sankey for presenting purposes.



In 2015, at EU level, the share of phosphorous in food waste was 10%. Regarding the losses to the environment, these were 39.4% of the total amount of P available in the food chain. Table 8 presents these shares in each Member State for 2015.

Table 8. Shares of P in food waste and lost to the environment in the EU and for each 27MS in 2015.

Member State	Share of P in food waste (%)	Share of P lost to the environment (%)
Austria	9.2	13.5
Belgium	8.3	40.0
Bulgaria	6.2	39.1
Croatia	11.5	71.4
Cyprus	9.1	35.2
Czech Republic	9.1	27.8
Denmark	9.3	47.7
Estonia	8.8	21.1
Finland	9.7	44.0
France	8.7	46.5
Germany	8.6	37.3
Greece	8.8	45.1
Hungary	6.3	19.8
Ireland	11.4	34.4
Italy	10.0	32.3
Latvia	10.7	30.9
Lithuania	9.0	22.2
Luxembourg	8.4	25.2
Malta	8.4	58.7
Netherlands	8.3	36.2
Poland	10.5	31.0
Portugal	9.4	30.3
Romania	10.1	55.7
Slovakia	8.0	29.2
Slovenia	9.4	27.7
Spain	8.9	32.8
Sweden	8.4	23.2
European Union	**10**	**39.4**

2.3 Nutrient flows per regions and sources (web maps and dashboards)

Nutrient flows (N and P) from different data sources, presented in Session 2.1, were analysed per region and per source type, providing spatial information at different spatial level of aggregation. Table 1 provides an overview of the available data. All data can be explored in the web maps and dashboards https://water.jrc.ec.europa.eu

2.3.1 Spatial maps (web maps)

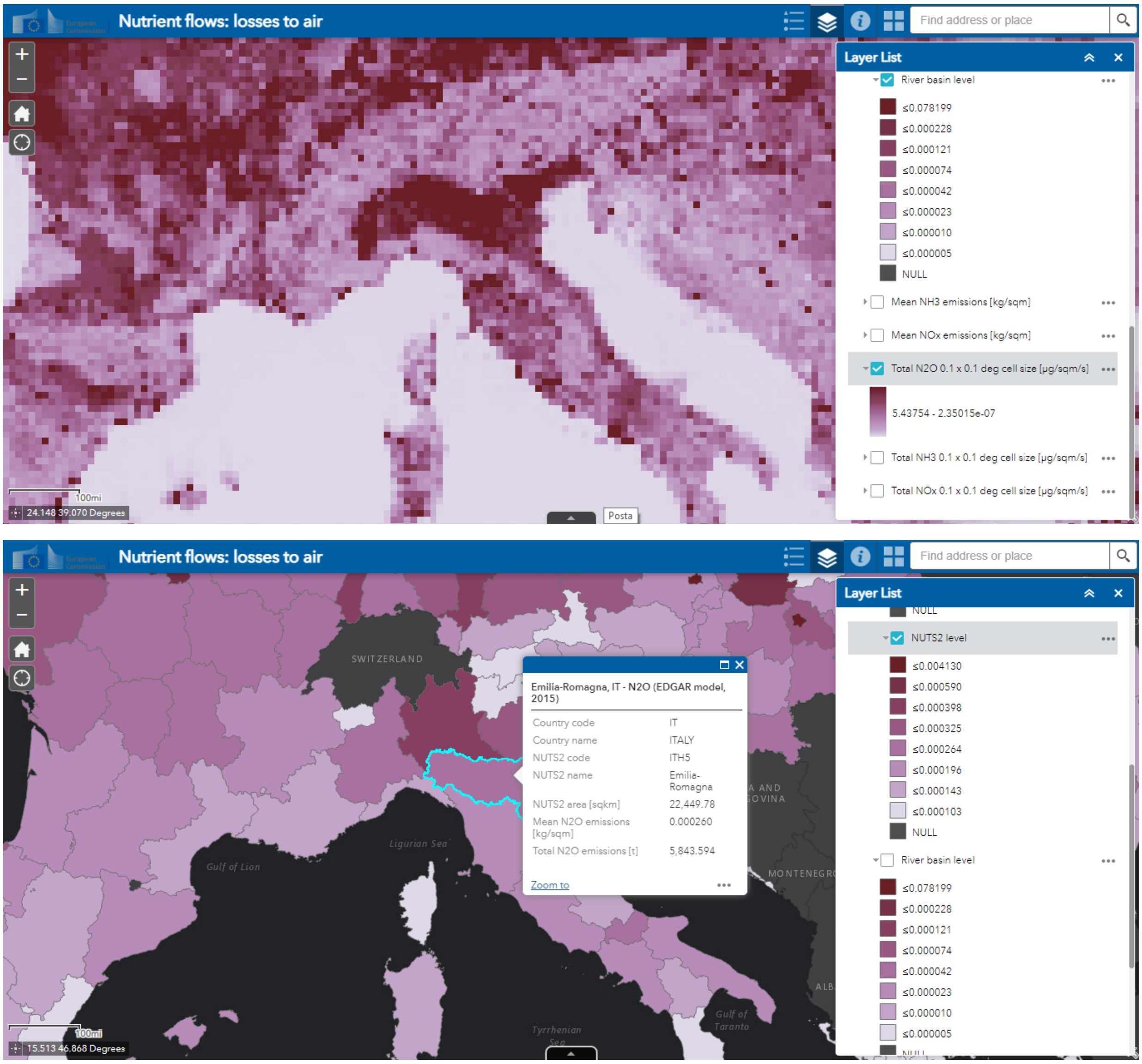
Interactive maps showing nutrient losses to air, water and soils were developed, based on different data sources available in the project. Spatial nutrient losses were aggregated and presented at different spatial units (country, nuts2, river basin, catchment). The available maps and sources are listed in Table 9. An example is shown in Figure 9. Available spatial maps:

- Losses to air
- Losses to water
- Changes in soil

Table 9. List of spatial map produced in the study, with data aggregation type and spatial resolution.

Topic	Spatial maps	Type of aggregation	by country	by NUTS2	by river basin	catchment	Unit of measure
Losses to air	EDGAR N2O	MEAN	√	√	√		[kg/sqm]
	EDGAR NOx	MEAN	√	√	√		[kg/sqm]
	EDGAR NH3	MEAN	√	√	√		[kg/sqm]
Losses to water	GREEN PsN	SUM	√	√	√	√	[t]
	GREEN SdN	SUM	√	√	√	√	[t]
	GREEN DeN					√	[kg/ha]
	GREEN N loads to the seas	SUM	√	√	√	√	[t]
	GREEN PsP	SUM	√	√	√	√	[t]
	GREEN SdP	SUM	√	√	√	√	[t]
	GREEN DeP					√	[kg/ha]
	GREEN P loads to the seas	SUM	√	√	√	√	[t]
	DayCent NO3 leaching (N leaching)	MEAN	√	√	√		[kg/ha]
	DayCent organic N leaching (N in DOC)	MEAN	√	√	√		[kg/ha]
Inputs to soil	EMEP Ndep	MEAN	√	√	√		[mg/sqm]
	DayCent MinN	MEAN	√	√	√		[kg/ha]
	DayCent ManN	MEAN	√	√	√		[kg/ha]
	DayCent N fixation	MEAN	√	√	√		[kg/ha]
Changes in soil	DayCent N surplus	MEAN	√	√	√		[kg/ha]
	DayCent N gross erosion*	MEAN	√	√	√		[kg/ha]
	Global P erosion	MEAN	√	√	√		[kg/ha]
	LUCAS point	MEAN	√	√	√		[kg/ha]

Figure 9. Example of interactive maps of nutrient emissions to the environment: N2O emissions to air (source EDGAR) original grid map (up) and aggregated values by NUTS2 (bottom).



2.3.2 Summary by country (dashboard)

A dashboard representing all N and P fluxes (for which data were available in the project) per EU27 countries was developed. The dashboard allows to compare the magnitude of different fluxes, considering different sources and environmental losses. See examples Figures 10 and 11.

Figure 10. Dashboard representing the summary of nutrient flows by country for EU27. On the top panels map and bar chart refer to nitrogen flows, in the bottom panel map and bar chart refer to phosphorus flows. Flows are expressed in tons.

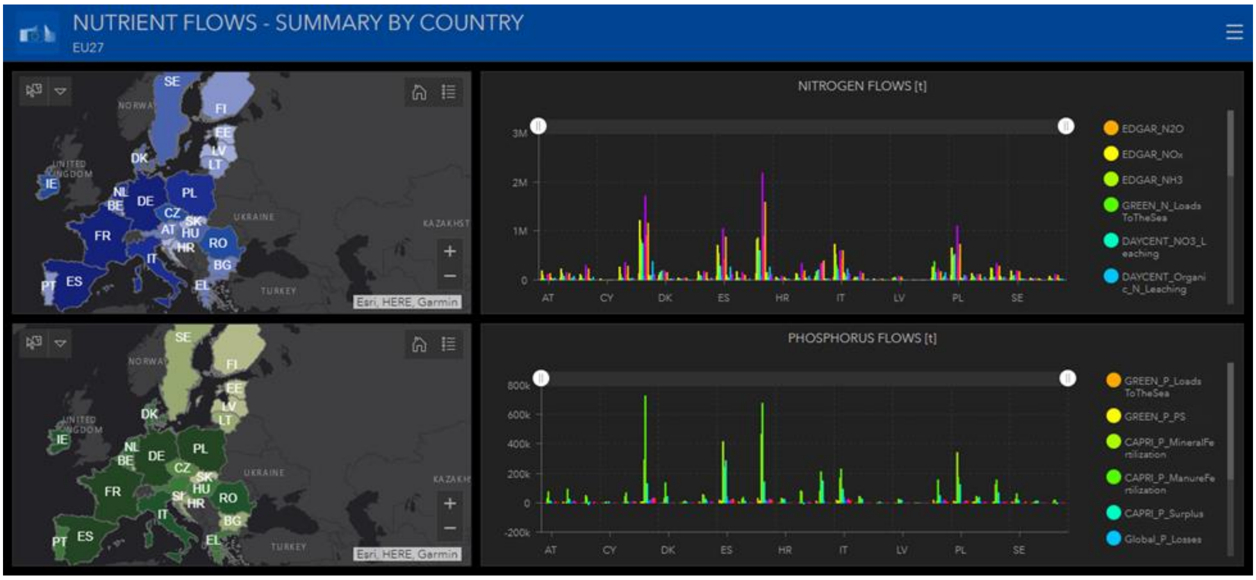
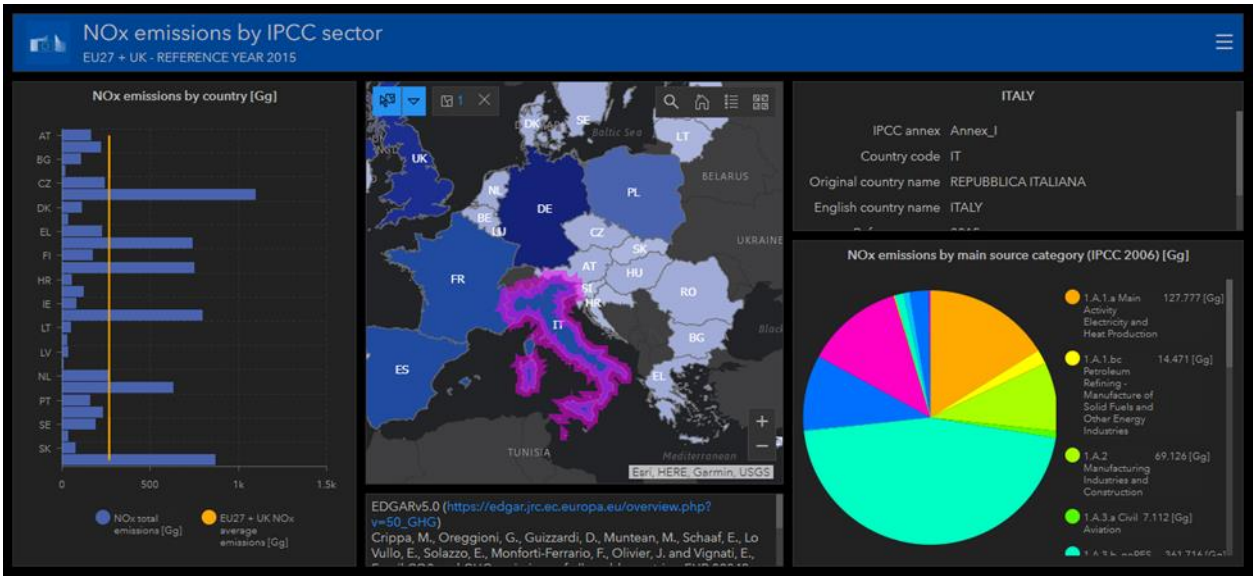


Figure 11. Dashboard representing the summary of nutrient flows by country for EU27. Clicking on a country in the map, the bar chart is automatically updated and shows data related to that country. In case of multiple selection the bar chart shows data for each country separately.

2.3.3 Sources contribution (dashboard)

Emissions to air of NOx, NH3, and N2O by IPCC sectors were represented per EU27 countries showing the contribution of different sectors. An example is shown in Figure 12.

Figure 12. Dashboard 'NOx emissions by IPCC sector' for EU27 countries and UK. Left panel shows a bar chart with total emissions by country in Gg. The orange line represents the average EU emissions. Clicking on countries in the map, the country details and the pie chart on the right side update automatically showing data referred to selected country(ies). In case of multiple selection on the map, the pie chart shows the sum of contributions for each sector.



3 Distance to targets

In the Biodiversity Strategy the Commission has set the goal of reducing nutrient losses by at least 50%. "This will be achieved by implementing and enforcing the relevant environmental and climate legislation in full, identifying with Member States the nutrient load reductions needed to achieve these goals, applying balanced fertilisation and sustainable nutrient management, and by managing nitrogen and phosphorus better throughout their lifecycle. To this end, the Commission will work with Member States to develop an Integrated Nutrient Management Action Plan in 2022" (INMAP). Furthermore, the Farm to Fork Strategy indicated that the INMAP will "address nutrient pollution at source and increase the sustainability of the livestock sector", and then the Zero Pollution Action Plan mentioned that the INMAP will address "holistically a long-standing environmental challenge, maximising synergies between policies and making best use of the green architecture of the new common agricultural policy, especially via conditionality and eco-schemes". The goal of reducing nutrient losses will be achieved by implementing the current EU legislation on nutrient and by a new holistic action plan (INMAP) for sustainable nutrient management. In this Chapter we illustrate the current[5] EU legislation dealing with nutrient emissions to the environment (air, soil, water) and nutrient management and recycling in waste, highlighting existing environmental goals and, where possible, showing data on the distance to the policy targets. We conclude the Chapter summarising the indicators adopted in the EU legislation for monitoring the progress towards policy objectives, and we discuss the EU target of halving nutrient losses in the perspective of planetary boundaries.

3.1 Nutrient emissions to air

3.1.1 The National Emission reduction Commitments Directive (NECD)

The National Emission reduction Commitments Directive -NECD- (2016/2284/EU) limits the maximum national emission of five main pollutants in the air to move towards achieving levels of air quality that do not give rise to significant negative impacts on and risks to human health and the environment (acidification, eutrophication and ground-level ozone pollution). Replacing earlier legislation (Directive 2001/81/EC), the NECD sets emission reduction commitments for the periods 2020 – 2029 as well as more ambitious ones for 2030 and beyond for five main air pollutants (NOx, NMVOCs, SO2, NH3 and PM2.5). It also ensures that the emission ceilings for 2010 set in the earlier directive remain applicable for Member States until the end of 2019.The EEA Dashboard "NECD Emission data viewer" provides access to the latest air pollutant emission inventory reported to EEA by EU Member States as well as Member States reduction commitments for 2020 and 2030.

EU Member States were required to report a national air pollution control programme (NAPCP) in 2019 for the first time. Updates of the NAPCP have to be provided at least every four years thereafter. The NAPCPs is the main governance instrument by which EU Member States must ensure that the emission reduction commitments for 2020-2029 and 2030 onwards are met. The NAPCPs include policies and measures (PaMs) that Member States selected as relevant for fulfilling their commitments to reduce emissions set for the periods 2020-2029 and from 2030 onwards. An overview on individual PaMs by sector reported by Member States is provided by the EEA dashboard "National Emission Ceiling Directive – Policies and Measures (PaMs) to reduce air pollutants emissions".

The latest assessment regarding the reduction commitments and compliance of Member States is presented in EEA (2021). The assessment indicates that in 2019 "all Member States respected their respective national ceilings for NOx", while for NH3 Croatia, Czechia, Ireland and Spain exceeded their 2010 national emission ceilings. To achieve the 2030 emission reduction commitments, the EU27 will need to further reduce NH3 emissions by 12% and NOx emissions by 36%, (Figure 13, data source: EEA, 2021).

[5] At the time of report preparation March 2022.

Figure 13. Percentage emission reductions compared with 2019 levels required by EU Member States to meet their emission reduction commitments for 2030 onwards. Source: EEA, 2021 (https://www.eea.europa.eu/publications/national-emission-reduction-commitments-directive-2021, data accessed August 2022).

3.1.2 Industrial Emissions Directive (IED)

The Industrial Emissions Directive -IED- (2010/75/EU) is the main EU instrument regulating pollutant emissions to air, water and land (including NH3, NOX, N2O, total N, total P) from industrial installations. The IED aims to achieve a high level of protection of human health and the environment taken as a whole by reducing harmful industrial emissions across the EU, in particular through better application of Best Available Techniques (BAT). Installations undertaking the industrial activities listed in Annex I of the IED are required to operate in accordance with a permit (granted by the authorities in the Member States). This permit should contain conditions set in accordance with the principles and provisions of the IED.

The IED regulates the emissions through the establishment of sector-specific BAT REFerence documents - BREFs - containing information about the sector and the latest emission control techniques used. The key chapter of a BREF, the BAT Conclusions, are then passed as secondary legislation (implementing decision). BATs cover both the technology used and the way in which the installation is designed, built, maintained, operated and decommissioned. It aims to achieve a high level of environmental protection under economically and technically viable conditions[6].

Through the European Pollutant Release and Transfer Register (E-PRTR), emission data at facility level reported by Member States are made accessible in a public register, which is intended to provide environmental information on major industrial activities.

([6]) https://www.fuelseurope.eu/policy-priorities/environment-air-quality/industrial-emissions-directive-ied/

3.2 Nutrient emissions to water

3.2.1 Water Framework Directive (WFD)

The Water Framework Directive (WFD, 2000/60/EC) establishes a framework for protecting and enhancing aquatic ecosystems and ensuring the sustainable use of water resources. The Directive sets the environmental objective of achieving good status for all water bodies: rivers, lakes, groundwater, transitional and coastal waters (by 2027). Groundwater bodies achieve good status when their quantitative status and chemical status are at least good. Among other parameters, for being in good chemical status nitrates concentration in groundwater should not exceed 50 mg/L (Groundwater Directive 2006/118/EC, Annex 1). Surface water bodies achieve good status when both their ecological status and chemical status are at least good. The ecological status is an evaluation of the condition of water bodies as high, good, moderate, poor or bad, based on assessment methods that consider biological quality elements (BQEs, that are phytoplankton, flora, invertebrate fauna and fish fauna), and information on physico-chemical and hydromorphological conditions of the water body. 'Nutrient conditions' contributes to the evaluation of the ecological status (i.e. 'Nutrient conditions' is a key component of the 'Chemical and physico-chemical elements supporting the biological elements', WFD, Annex V).

As per the WFD, Member States analyse the environmental impact of human activities on waters and develop River Basin Management Plans (RBMPs) every 6 years, including a Programme of Measures to achieve the environmental objective of good status. The measures include among others the implementation of the EU legislation for the protection of water from nutrient pollution from point sources (Directive 91/271/EEC concerning urban waste-water treatment; Industrial Emissions Directive 2010/75/EU) and diffuse agricultural sources (Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources). Measures include also the protection of water bodies for abstraction of drinking water, to avoid their deterioration and reduce the level of purification treatment. The Drinking Water Directive (Directive EU 2020/2184) prescribes a maximum concentration of 50 mg/L of nitrate and 0.50 mg/L of nitrite for water intended for human consumption.

Nutrient pollution affects the condition of water ecosystems. According to the second RBMPs, diffuse pollution, atmospheric deposition and point sources were indicated among the major pressures impairing surface waters. Specifically, 26% of surface water bodies reported impact of nutrient pollution and 17% of ground water bodies area reported impact of nutrient pollution([7]). Regarding the distance to the WFD environmental targets: 74% of the groundwater bodies are in good chemical status; 38% of surface water bodies have achieved good chemical status and 40% good ecological status or potential (COM(2019) 95 final([8])). Data on the groundwater bodies chemical status and surface waters ecological status per EU27 countries are shown if Figures 14 and 15, respectively. An estimation of the probability of rivers of being affected by nutrient pollution impact is also shown in Figure 16 (assessment based on modelling not on reported data).

Poikane et al. (2019) analyzed nutrient criteria adopted by EU Member States to support good ecological status. They highlighted that different threshold nutrient concentrations are used to define the boundary between "good" and "moderate" ecological status across Europe. For example good-moderate threshold concentrations in the range of 0.25–4.00 mgN/l (total N) and 5–500 µgP/l (total P) were reported per lakes, and good-moderate threshold concentrations in the range of 0.25–35 mgN/l (total N) and 8–660 µgP/l (total P) were reported per rivers.

([7]) Data sources EEA, 2018. WISE Water Framework Directive (data viewer) https://www.eea.europa.eu/data-and-maps/dashboards/wise-wfd accessed in December 2021.

([8]) COM(2019) 95 final, Report of the Commission on the implementation of the Water Framework Directive (2000/60/EC) and the Floods Directive (2007/60/EC) Second River Basin Management Plans, First Flood Risk Management Plans.

Figure 14. Chemical status of groundwater bodies according to the 2nd River Basin Management Plans (data up to 2015). Data source: EEA, 2018. WISE Water Framework Directive (data viewer) https://www.eea.europa.eu/data-and-maps/dashboards/wise-wfd (access December 2021)

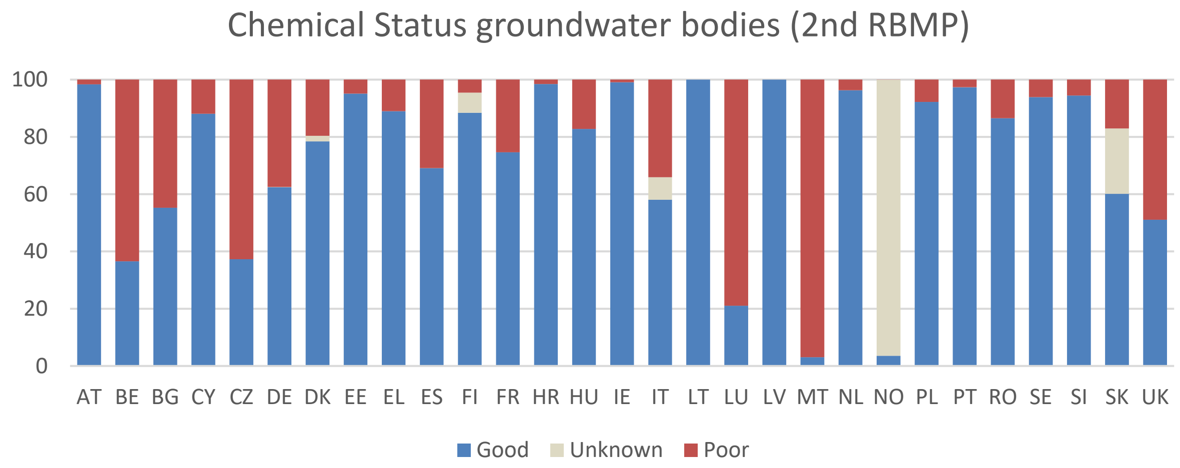


Figure 15. Ecological status of surface water bodies according to the 2nd River Basin Management Plans (data up to 2015). Data source: EEA, 2018. WISE Water Framework Directive (data viewer) https://www.eea.europa.eu/data-and-maps/dashboards/wise-wfd (access December 2021)

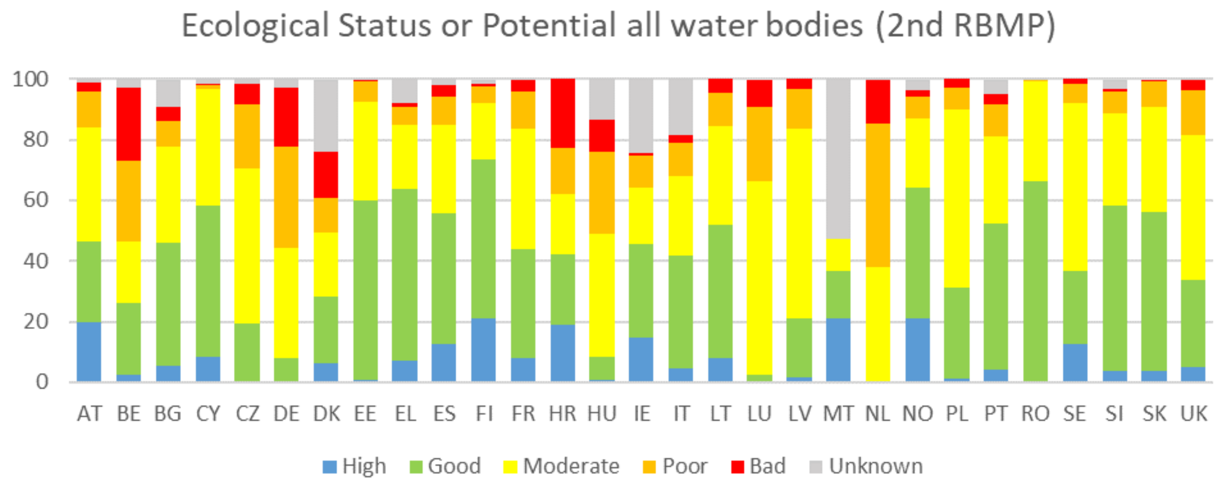
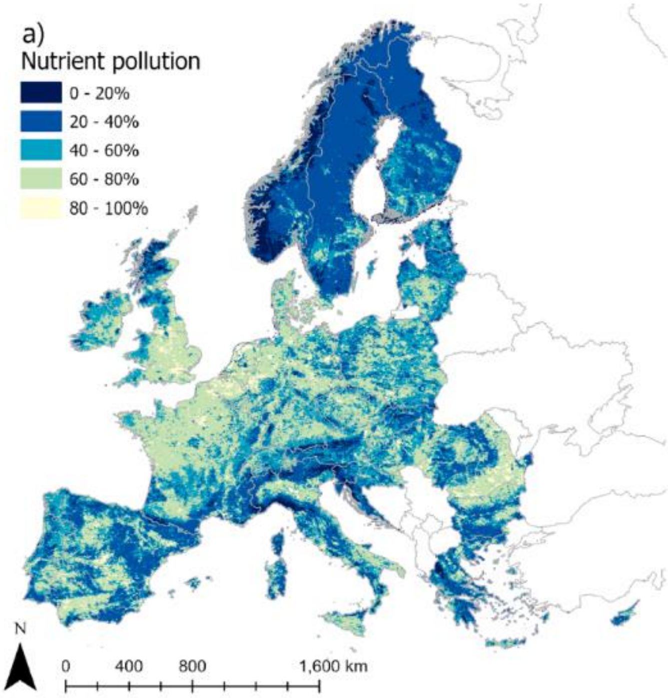


Figure 16. Modelled probability of occurrence of nutrient pollution in European rivers. Source: Vigiak et al. 2021.



3.2.2 Urban Waste Water Directive (UWWTD)

The Urban Waste Water Treatment Directive (UWWTD, 91/271/EEC) aims at protecting waters from the adverse effect of waste water discharges from domestic and certain industrial sources. The Directive establishes the size of agglomerations that require waste water collection and treatment, the necessary level of treatment, and the deadlines to achieve the progressive implementation of the legislation. More stringent treatments are required for waste waters discharging in sensitive areas, which are areas where freshwater bodies (lakes, estuaries and coastal waters) are eutrophic or may become eutrophic in the near future in absence of protective actions (Annex II)([9]). Nitrogen and phosphorus in waste water are reduced according to the level of treatment (i.e. primary, secondary, tertiary treatment, advance P removal).

The UWWTD entered into force 30 years ago and has led to a progressive improvement of the collection and treatment of urban waste water, with a reduction of nutrient pollution discharged into surface waters. The compliance rates of the Directive are 95% for collection of waste waters, 88% for secondary treatment, and 86% for more stringent removal of phosphorus and nitrogen([10]). However, the full compliance with UWWTD has not being attained yet and the distance to target remains significant in some Member States([11]).

Recent estimates of N and P emissions from wastewaters to surface waters in the current situation (year 2016) and in the scenario of full implementation of the UWWTD indicate that overall for EU27 a reduction of 7% of N load and 13% of P load is needed to achieve the objective of the Directive (Pistocchi et al. in preparation) (Figure 17).

At present, the UWWTD is under revision. The Commission has launched the impact assessment that will analyse areas of improvements of the Directive and the effects of possible measures. The effects of

---

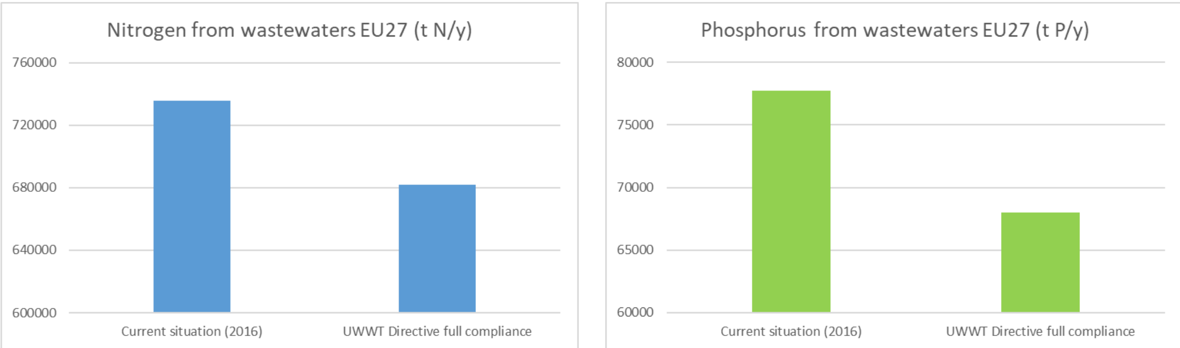
([9]) In the UWWTD 'Eutrophication' means the enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned" (Art.2).

([10]) COM(2020) 492 final. Tenth report on the implementation status and programmes for implementation (as required by Article 17 of Council Directive 91/271/EEC, concerning urban waste water treatment).

([11]) COM(2020) 492 final. Tenth report on the implementation status and programmes for implementation (as required by Article 17 of Council Directive 91/271/EEC, concerning urban waste water treatment).

scenarios of measures to further reduce N and P pollution from waste waters under the UWWTD are presented in Section 4.6 and refer to the work of Pistocchi et al. (in preparation).

Figure 17. Nitrogen (left) and phosphorus (right) emissions to surface waters from wastewaters (including urban waste water treatment plants, individual appropriate systems (IAS) and scattered dwellings) in the current situation (year 2016) and in case of full implementation of the UWWT Directive in EU27. Data source: Pistocchi et al. (in preparation).



3.2.3 Nitrates Directive (ND)

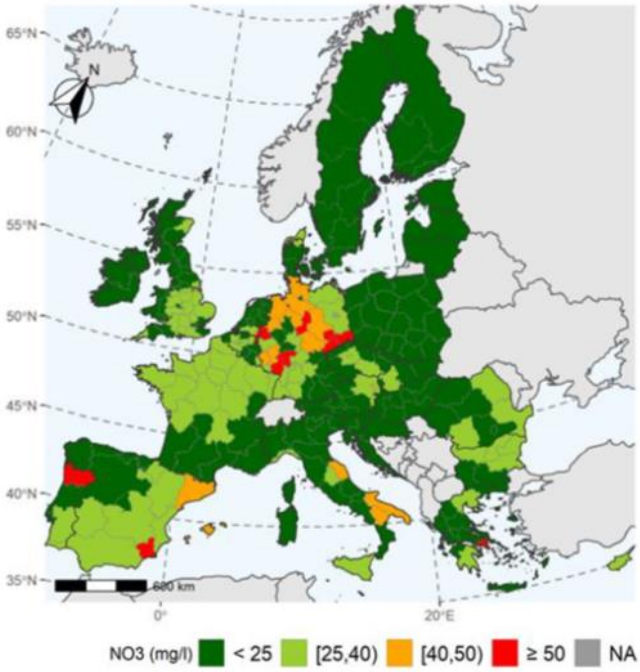
The objective of the Nitrates Directive (91/676/EEC) is to protect waters against pollution caused by nitrates from agricultural sources. According to the Directive, Member States designate nitrates vulnerable zones (NVZ) that are areas of land in their territories draining into the waters affected by pollution or that could be affected by pollution if action is not taken. Criteria for identifying such waters include, inter alia, whether surface waters (especially those intended for the abstraction of drinking water([12])) and groundwaters contain or could contain more than 50 mg/l nitrates if action is not taken, and whether natural freshwater bodies, estuaries, coastal waters, and marine waters are eutrophic or may become eutrophic in the near future if action is not taken (Annex I). Member States establish codes of Good Agricultural Practice (implemented by farmers on voluntary basis). These codes become mandatory in NVZs, where Action Programmes including measures to prevent and reduce nitrates pollution of water, such as a maximum annual application of livestock manure set at 170 kg N/ha([13]) also apply. Member States may also decide to apply their Action Programme on their whole territory, without having to designate NVZ. (Member States must also revise, at least every four years, the designation of vulnerable zones).

Within the frame of the Nitrates Directive, Member States shall monitor and report (every four years) to the Commission the nitrates concentration in surface and ground waters and the eutrophication of surface waters. In addition, they evaluate and revise the Action Programmes. An online viewer to access and explore data reported under the Nitrates Directive is available at this link https://water.jrc.ec.europa.eu (Nitrates Directive page).

The last implementation report of the Nitrates Directive (reporting period 2016-2019)([14]) indicates that 14.1% of groundwater stations exceeded the environmental target of annual average 50 mg nitrates/L (Figure 18), and 81% of marine waters and at least one third of rivers, lakes, transitional and coastal waters are reported as eutrophic. The report concludes that, in spite of some progress, the level of implementation is still insufficient to reach the objectives of the Directive.

([12]) 50 mg/l nitrates is also the limit of nitrates in drinking waters, Directive (EU) 2020/2184.
([13]) Member States can ask for “derogations” to the limit of 170 kg N/ha of livestock manure for areas where scientific evidence show that higher amount of nitrogen from manure does not cause water pollution.
([14]) COM(2021) 1000 final. Report on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2016–2019.

Figure 18. Annual average nitrate concentrations in groundwater at the NUTS2 level (reporting period 2016-2019). Source: European Commission (2021) ([15]).



3.2.4 Marine Strategy Framework Directive (MSFD)

The Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC) establishes a framework for achieving good environmental status (GES) in the marine environment. GES is defined as "*the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive*" (Article 3). The GES of marine waters is characterised by 11 qualitative descriptors (Annex I). Most of them are influenced by nutrient pollution, including biodiversity, presence of non-indigenous species, fish population, reproduction, eutrophication and sea floor integrity (Descriptors 1-6).

The MSFD applies an ecosystem-based approach for the management of human activities, with the aim to ensure sustainable use of marine goods and services. For each marine region or subregion in their territory Member States develop a marine strategy where they assess the environmental status of national marine waters, establish environmental targets with associated indicators, and set a programme of measures and a monitoring programme to achieve the GES. The marine strategies are reviewed every 6 years. To achieve the objectives of the marine strategies Member States also cooperate in Regional Sea Conventions: the Helsinki Convention on the Baltic Sea (HELCOM), the OSPAR Convention on the North-East Atlantic (OSPAR), the Barcelona Convention on the Mediterranean (UNEP) and the Bucharest Convention on the Black Sea (Black Sea Commission).

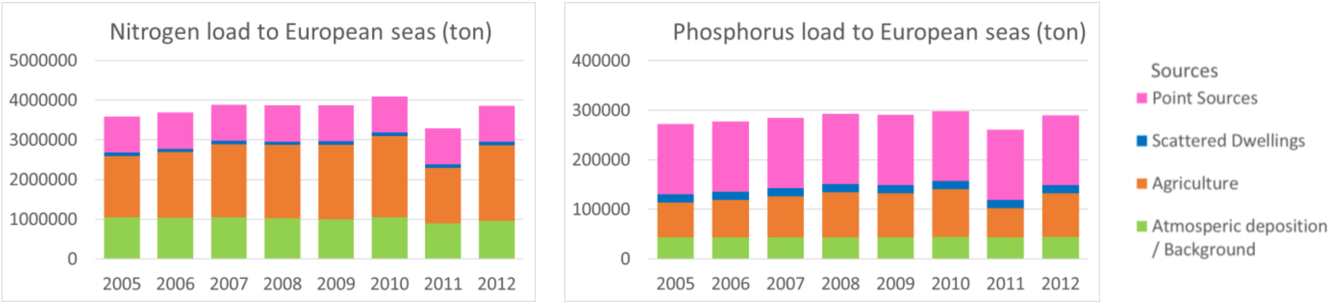
The last implementation report of the MSFD indicates that eutrophication and nutrient conditions are a problem in large part of coastal waters in the Baltic Sea, in southern North Sea, along the north-western coast of France and close to river outflows in the Mediterranean Sea, and that phytoplankton conditions pose a problem in the Black Sea([16]). Losses from agricultural sources are considered still too high (COM(2020) 259). In addition, the time lag between the reduction of nutrient input in the land and the effect on marine waters eutrophication might hamper the appreciation of improvements. An estimation

([15]) COM(2021) 1000 final. Report on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2016–2019.

([16]) COM(2020) 259 final. Report from the Commission to the European Parliament and the Council on the implementation of the Marine Strategy Framework Directive (Directive 2008/56/EC).

of N and P discharged to European seas, including sources contribution, is shown in Figure 19 (Grizzetti et al. 2021). For a very recent study, see also Vigiak et al. 2023 (For scenarios analysis of nutrient reductions to the European Seas see Section 4.6).

Figure 19. Total nitrogen (left) and total phosphorus (right) annual load to European seas (ton/y) for the period 2005-2012 (estimated by the GREEN model), and relative contribution of major sources: point sources, scattered dwellings, agriculture and background (P) or atmospheric deposition (N). Source: Grizzetti et al. (2021)[17].



3.3 The new EU Common Agricultural Policy (CAP)

In June 2018, the European Commission presented legislative proposals for a new CAP. The proposals outlined “a simpler and more efficient policy that will incorporate the sustainable ambitions of the European Green Deal”. After extensive negotiations between the European Parliament, the Council of the EU and the European Commission, agreement was reached on CAP reform and the new CAP was formally adopted on 2 December, 2021 (EU Regulation 2021/2116). The new CAP is due to be implemented from 1 January 2023[18].

3.3.1 Assessment and target-setting against common environmental and climate objectives

Three of the ten policy's specific objectives will concern the environment and climate (DG Agriculture and Rural Development 2019 p. 4). These objectives will be as follows:

1. Contribute to climate change mitigation and adaptation, as well as sustainable energy.
2. Foster sustainable development and efficient management of natural resources such as water, soil and air.
3. Contribute to the protection of biodiversity, enhance ecosystem services and preserve habitats and landscapes.

To address these (and other) CAP objectives, each Member State will draw up a "CAP strategic plan" (EU Regulation 2021/2115). In its plan, each Member State will analyse the situation on its territory in terms of strengths, weaknesses, opportunities and threats (SWOT) – as well as its related needs – in respect of these objectives. It will set quantified targets against the objectives and design "interventions" (types of action) for achieving them, on the basis of an EU-level menu. The Commission will approve the plan when satisfied with its quality. Year-by-year progress against the targets will be monitored and the plan will be adjusted as necessary.

The focus in this process will of course be on the CAP's own objectives. However, in its plan each Member State will have to show how, in pursuing the CAP's objectives, it will also make a contribution to achieving the objectives of various items of EU environmental and climate legislation (on biodiversity, water and air quality, greenhouse gas emissions, energy and pesticides) and how it will contribute to the Green Deal Objectives, including the reduction of 50% of the nutrient losses by 2030. In addition, when drawing up its CAP plan each Member State will take account of analysis and recommendations

[17] Vigiak et al. (2023) show timeseries 1990-2018.
[18] Source: https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/cap-glance_en

for action already made in the framework of that legislation (for example, analysis concerning water quality in lakes, rivers and groundwater).

Finally, an essential part of this framework will be an explicit obligation on Member States to clearly show greater ambition than at present with regard to care for the environment and climate.

Annex XIII of the CAP Strategic Plans Regulation provides the full list of legislation to be considered in the Member States Strategic Plans. The legislation referred to in Annex XIII concerning nitrogen and/or phosphorous in the environment is given below:

- Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.
- Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources.
- Directive 2008/50/EC of the European Parliament and of the Council on ambient air quality and cleaner air for Europe.
- Directive (EU) 2016/2284 of the European Parliament and of the Council of 14 December 2016 on the reduction of national emissions of certain atmospheric pollutants, amending Directive 2003/35/EC and repealing Directive 2001/81/EC.
- Regulation (EU) 2018/841 of the European Parliament and of the Council of 30 May 2018 on the inclusion of greenhouse gas emissions and removals from land use, land use change and forestry in the 2030 climate and energy framework, and amending Regulation (EU) No 525/2013 and Decision No 529/2013/EU.
- Regulation (EU) 2018/842 of the European Parliament and of the Council of 30 May 2018 on binding annual greenhouse gas emission reductions by Member States from 2021 to 2030 contributing to climate action to meet commitments under the Paris Agreement and amending Regulation (EU) No 525/2013.
- Directive 2009/28/EC on the promotion of the use of energy from renewable sources.

3.3.2 Conditionality in the new CAP

Conditionality is a system of linkage between area and animal-based CAP payments (in Pillar I or Pillar II) and a range of obligations. When recipients of these payments do not meet the obligations, the payments may be reduced (DIONE 2020).

These obligations originate either in CAP legislation (in the case of "standards for good agricultural and environmental condition" - GAEC) or in non-CAP directives and regulations (in the case of "statutory management requirements" - SMRs). An example of a non-CAP directive giving rise to SMRs is the "Nitrates Directive", which helps safeguard water quality. Farmers have to respect SMRs in any case, but their inclusion in the system of conditionality creates a link with CAP payments.

All the GAEC standards and some of the SMRs are environmental – concerning climate change, water, soil, and biodiversity/landscapes. The **provisional** list of GAEC and SMR proposed by the Commission impacting the balance of nitrogen and phosphorous in agriculture (directly and indirectly) is given in Table 10.

Table 10. SMR: Statutory Management Requirements (SMRs) and Standards for good agricultural and environmental condition of land (GAECs) impacting the nitrogen and phosphorous balance in agriculture (EU Regulation 2021/2115).

Main Issue	Requirements and standards		Main objective of the standard
Climate change (mitigation of and adaptation to) ([1])	GAEC 1	Maintenance of permanent grassland based on a ratio of permanent grassland in relation to agricultural area at national, regional, subregional, group-of-holdings or holding level in comparison to the reference year 2018 Maximum decrease of 5 % compared to the reference year	General safeguard against conversion to other agricultural uses to preserve carbon stock
	GAEC 2	Protection of wetland and peatland	Protection of carbon-rich soils
	GAEC 3	Ban on burning arable stubble, except for plant health reasons	Maintenance of soil organic matter
Water	SMR 1	Directive 2000/60/EC of 23 October 2000 of the European Parliament and of the Council establishing a framework for Community action in the field of water policy: Article 11(3)(e) and Article 11(3)(h) as regards mandatory requirements to control diffuse sources of pollution by phosphates	
	SMR 2	Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (OJ L 375, 31.12.1991, p. 1): Articles 4 and 5	
	GAEC 4	Establishment of buffer strips along water courses[1]	Protection of river courses against pollution and run-off
Soil (protection and quality) ([2])	GAEC 5	Tillage management reducing the risk of soil degradation, including consideration of the slope gradient	Minimum land management reflecting site specific conditions to limit erosion
	GAEC 6	Minimum soil cover to avoid bare soil in periods that are most sensitive	Protection of soils in periods that are most sensitive
	GAEC 7	Crop rotation in arable land, except for crops growing under water	Preserve the soil potential

([1]) GAEC1: Grassland conversion to cropland bears a risk of increased nitrate (NO_3^-) leaching and nitrous oxide (N_2O) emission due to enhanced nitrogen (N) mineralization. GAEC2: N_2O emissions from drained organic soils are significantly higher than from undrained peatlands. GAEC3: Burning of arable stubble is a source of NO_X.

([2]) GAEC5 –GAEC6: Preventing soil erosion reduces the risk of N and P loss. GAEC7: Crop rotation may impact the soil nutrient balance and fertilization requirements.

3.4 Regulatory framework on waste

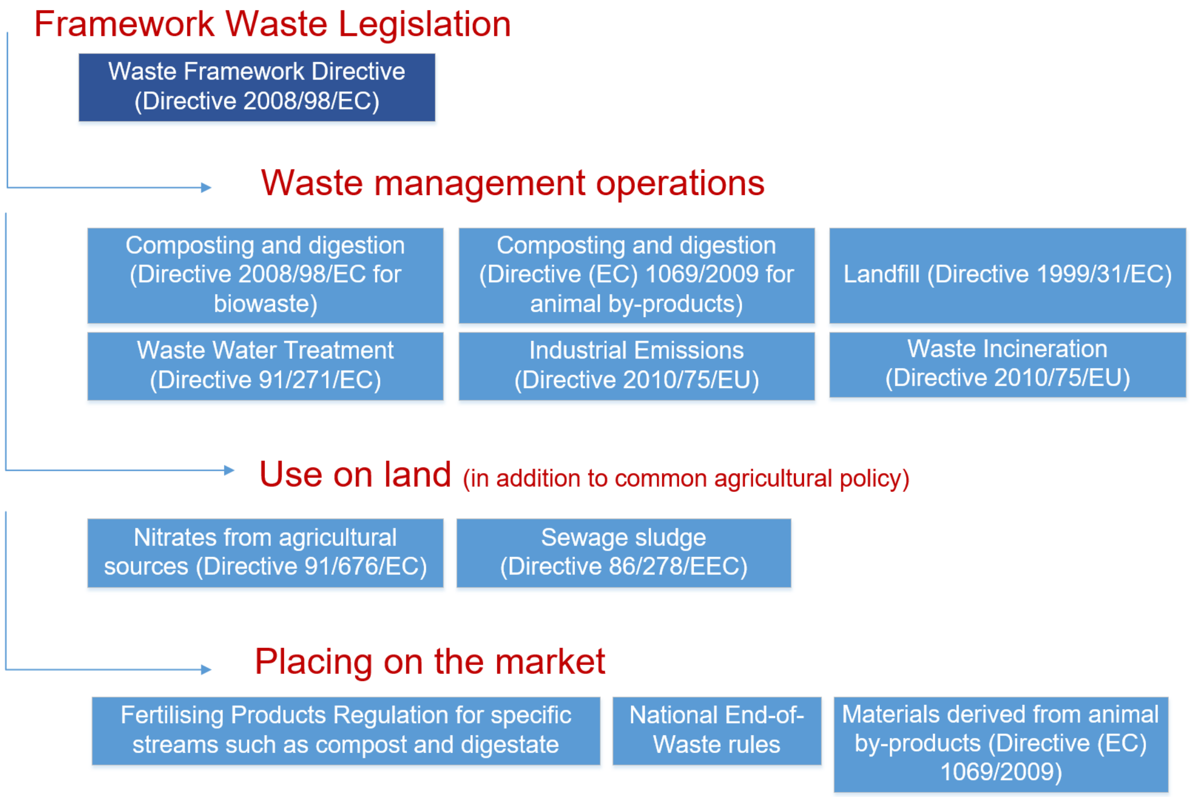
Waste is defined in the in the Waste Framework Directive (Directive 2008/98/EC) as ‘any substance or object which the holder discards or intends or is required to discard’. Animal by-products including manure, destined for incineration, landfilling or use in a biogas or composting plant, are also considered a waste material, and are subject to Directive 2008/98/EC. Albeit human faecal matter is excluded from the scope of Directive 2008/98/EC, it is covered under community legislation on sewage sludge (Directive 86/278/EEC) and on the operation of waste water treatment plants (Directive 91/271/EEC) and will therefore be covered in this section.

Nutrients in waste streams are mainly present in sludges from municipal and industrial waste water treatment plants, animal by-products including (excess) manure, and municipal bio-waste (see section “flows”). In addition, NH3 and NOx can be trapped from N-rich off-gases from specific facilities and industries (e.g. livestock stables, incineration plants) and end up as residues that are disposed of. This section will mostly focus on relevant legislation relevant to these nutrient-rich waste streams.

A vast set of EU policies influences nutrient flows and management of waste streams. Here, we have structured the most important pieces EU legislation related to nutrients in waste departing from the Waste Framework Directive 2008/98/EC as the framework waste legislation (Figure 20). Guided by the principle of the waste hierarchy, legislations that impact upon waste management operations, the use on land of waste derived materials, and the placing on the market of waste-derived materials will be reviewed (Figure 20), with a specific emphasis on quantitative targets when available.

The **waste hierarchy**, pronounced in the Waste Framework Directive, is one of the foundations of EU waste management. It promotes the prevention of waste, and regulates the collection and management of food and animal waste as well as other nutrient-rich waste to promote safe nutrient re-use and recycling. Materials that contain nutrients can cease to be waste and for instance be used as fertilising materials on agricultural land if compliant with certain quality requirements and conditions laid down in EU legislation. The **placing on the market of fertilising materials** that are derived from waste materials is regulated through Regulation (EU) 2019/1009 on the making available on the market of EU fertilising products. A set of **specific pieces of EU legislation deal** with specific waste management operations (e.g. Directive 1999/31/EC on the landfill of waste; Directive 2010/75/EU on industrial emissions) and regulate the use on land of specific materials with a view to enhance food safety and reduce nutrient losses (e.g. Directive 86/278/EEC on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture; Directive 91/676/EEC on the protection of waters against pollution caused by nitrates from agricultural sources).

Figure 20. Non-exhaustive list of EU legislation that impacts upon the management of nutrient-rich waste streams, with a particular emphasis on biowaste, sewage sludge, processed manure and industrial waste.



3.4.1 Framework waste legislation

The Waste Framework Directive 2008/98/EC sets the **basic concepts and definitions related to waste management, including definitions of waste, recycling and recovery** (Figure 21). It lays down some basic waste management principles, as well as the conditions for materials that cease to be waste. It requires that waste be managed without endangering human health and harming the environment without risk to water, air, soil, plants or animals and without adversely affecting the countryside or places of special interest. The **waste hierarchy** promotes the prevention of waste, and encourages the collection and management of waste to promote safe nutrient re-use and recycling.

Figure 21. The EU waste management hierarchy as outlined in Article 4 of Directive.

The **prevention of waste** is an important pillar of the Directive. Article 9 indicates that Member States shall take measures to prevent waste generation. Particularly important minimum measures that have an intimate link to nutrients are:

— “reduce the generation of food waste in primary production, in processing and manufacturing, in retail and other distribution of food, in restaurants and food services as well as in households as a contribution to the United Nations Sustainable Development Goal to reduce by 50 % the per capita global food waste at the retail and consumer levels and to reduce food losses along production and supply chains by 2030”. This target also forms part of the EU Farm to Fork Strategy and is of key importance given the contribution of food waste to the total N and P losses within the EU (see section flows).

— “target products containing critical raw materials to prevent that those materials become waste”. This is important as phosphorus has been added to the EU Critical Raw Material list in the year 2020.

— “reduce waste generation in processes related to industrial production, manufacturing, […] taking into account best available techniques”. This is important as some industrial activities may give rise to aqueous nutrient losses (e.g. food industry) and gaseous N losses in the form of NH_3 (e.g. intensive rearing of pig and poultry) or NO_x (e.g. paper and pulp industry, chemical industries that apply incineration processes).

Main legislation related to waste recycling, recovery and disposal options are operations are given in the subsequent sections.

3.4.2 Legislation and targets for waste management operations

3.4.2.1 Waste Framework Directive 2008/98/EC

— With relevance to nutrients, specific **targets** related to collection, re-use and recycling have been instated regarding municipal waste and bio-waste:

— Preparing for **re-use and the recycling of municipal waste to a minimum of 55%, 60% and 65% by 2025, 2030 and 2035, respectively** (Article 11). A Member State may reduce the targets by 5% and postpone the deadlines for attaining the abovementioned targets by up to five years provided that a Member State had an initially low degree of re-use and recycling (<20% of municipal waste in 2013) or high degree of landfilling (>60% municipal waste in 2013) and submits an implementation plan within the corresponding deadline. The rules and calculation methods for verifying compliance with these targets can be found in Commission Decision 2011/753/EU. Additional rules for the calculation, verification and reporting of data on waste in accordance with the amended Waste Framework Directive can be found in Commission Decision (EU) 2019/1004.

— Addressing municipal bio-waste is crucial for moving towards the targets on municipal waste. By 31 December 2023, bio-waste shall be either separated and recycled at source, or is collected separately and is not mixed with other types of waste of different biodegradability and compostability properties (subject to the text referenced above). Member States shall take measures in accordance with to: (a) encourage the recycling, including composting and digestion, of bio-waste in a way that fulfils a high level of environment protection and results in output which meets relevant high-quality standards; (b) encourage home composting; and (c) promote the use of materials produced from bio-waste.

3.4.2.2 Regulation (EC) No 1069/2009 on animal by-products

Process conditions apply to the processing of animal by-products in e.g. composting and digestion plants as well as to ensure the appropriate hygienisation of any animal by-product or derived material that will be placed on the market (Animal by-product regulation (EC 1069/2009) implemented by the 142/2011/EU regulation). The processing of animal by-products into compost or digestate is optional and unprocessed manure can be applied on agricultural land without treatment when no third party is involved. No targets are in place for use routes leaving the material holder the possibility to either re-use or discard animal by-products.

3.4.2.3 Directive 1999/31/EC on the landfill of waste

According to the EU's waste hierarchy, landfilling is the least preferable option and should be limited to the necessary minimum. Waste, and particularly organic waste, that is landfilled can have dangerous effects on the environment and on human health. The generation of leachate can contaminate groundwater and methane is produced, which is a potent greenhouse gas. In addition, where recyclable waste is landfilled, materials are unnecessarily lost from Europe's economy. The Landfill Directive sets out strict operational requirements for landfill sites with the objective to protect both human health and the environment.

Landfilling of bio-waste is addressed in the Landfill Directive which requires the **diversion of biodegradable municipal waste from landfills**. To support the EU's transition to the circular economy, the amending Directive (EU) 2018/850 also introduces restrictions on landfilling of all waste that is suitable for recycling or other material or energy recovery from 2030. The Landfill Directive obliged Member States to reduce landfilling of municipal biodegradable waste to a maximum of 75 % by 2006, 50 % by 2009 and 35 % by 2016, compared to a 1995 baseline. The revised Directive limits the share of **municipal waste landfilled to 10% by 2035**, and introduces rules on calculating the attainment of municipal waste targets. EU countries must implement national strategies to progressively reduce the amount of biodegradable waste sent to landfills. Finally, the Directive sets specific operational requirements such as permitting, waste acceptance, technical requirements in the operational and after-care phases and reporting.

3.4.2.4 Directive 91/271/EEC on waste water treatment

The urban waste water treatment directive is a 'basic measure' under the Water Framework Directive. Given the significant amounts of nutrients contained in waste waters from households and industries that discharge waters to municipal plants, it plays a significant role in improving the status of bodies of water in the EU. In view of the significant challenge to ensure good status for the EU's bodies of water by latest 2027, effective collection and treatment of urban waste water is very important.

The directive indicates that **sludge arising from waste water treatment shall be re-used whenever appropriate** (Article 14). Disposal routes shall minimize the adverse effects on the environment. Environmental targets are set for nitrogen and phosphorus discharges in effluents (Annex I of the Directive), for which reason increased implementation of the current targets and/or stricter targets for nutrient discharges in a possibly revised Regulation may impact upon the nutrient contained in the sludge waste materials.

3.4.2.5 Directive 2010/75/EU on industrial emissions

The Industrial Emissions Directive (IED) aims to achieve a high level of protection of human health and the environment taken as a whole by **reducing harmful industrial emissions** across the EU, in particular through better application of Best Available Techniques (BAT). Also the **conditions for the incineration of waste** are laid down in the Industrial Emissions Directive. In order to define BAT and the BAT-associated environmental performance at EU level, the Commission organises an exchange of information with experts from Member States, industry and environmental organisations. This process results in BAT Reference Documents (BREFs); the BAT conclusions contained are adopted by the Commission as Implementing Decisions. The IED requires that these BAT conclusions are the reference for setting permit conditions.

Around 50,000 installations undertaking the industrial activities listed in Annex I of the IED are required to operate in accordance with a permit (granted by the authorities in the Member States). This permit should contain conditions set in accordance with the principles and provisions of the IED. For certain activities, i.e. large combustion plants, waste incineration and co-incineration plants, the IED sets EU wide emission limit values for selected pollutants (e.g. NOx) that are emitted to the atmosphere. For other industries, e.g. intensive rearing of pigs and poultry, food and drink industries), good management practices (for instance to avoid N2O losses from manure storage) should be applied. The Directive aims on gradually reducing emissions based on the application and enforcement of techniques that reduce pollution. No long-term quantitative targets are present in the Directive. A recent study linked to the revision of the IED directive has analysed the options of including intensive cattle farms (in addition to large poultry and pigs farms), and of lowering the thresholds above which permits are required for poultry and pig farms (Hekman et al. 2021).

3.4.3 Legislation and targets for the use of land on waste and waste-derived materials

3.4.3.1 Directive 91/676/EC on nitrates

Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources aims to protect water quality across Europe by preventing nitrates from agricultural sources polluting ground and surface waters and by promoting the use of good farming practices. The Directive prescribes the establishment of action programmes to be implemented by farmers within so-called nitrate vulnerable zones or equivalent areas defined by Member States on a compulsory basis. These programmes must include measures already included in Codes of Good Agricultural Practice, and other measures, such as limitation of fertiliser application (across the spectrum of mineral and inorganic fertilising materials; taking into account crop needs, all nitrogen inputs and soil nitrogen supply), and **maximum amounts of livestock manure to be applied** (corresponding to 170 kg nitrogen/hectare/year; unless a derogation has been granted that allows the application of higher maximum limits of nitrogen from manure in specific areas and under particular conditions). The manure application limits correspond to the sum of the total manure applied, regardless of its legal status (solely manure destined for use in a biogas or composting plant is considered a waste material, in contrast to e.g. manure applied at the same entity of its production in livestock or mixed farming systems).

3.4.3.2 Directive 86/278/EEC on sewage sludge

The Regulation was adopted to encourage the correct use of sewage sludge in agriculture and to regulate its use in order to prevent harmful effects on soil, vegetation, animals and humans. The principal value of the Directive is its role in the protection of human health and the environment against the harmful effects of contaminated sludge, and particularly metal therein, in agriculture. In more qualitative terms, Article 8 of the Directive indicates that sludge “shall be used in such a way that account is taken of the nutrient needs of the plants and that the quality of the soil and of the surface and ground water is not impaired”. The Directive does not set targets related to use routes and fate of sewage sludge, but a reference in made in the Urban Waste Water Treatment Directive to promote sludge application on agricultural land “when appropriate”.

3.4.4 Placing on the market

Waste management systems can help to achieve a circular economy and ensure that waste materials containing nutrients that can safely re-enter the biosphere, returning as such embedded nutrients to the environment. In line with Article 6 of Directive 2008/98/EC, materials having undergone a recycling or other recovery operation can cease to be waste (End-of-Waste materials), amongst others on condition that a market/demand exists for such a substance, and the use of the substance or object does not lead to overall adverse environmental or human health impacts.

The **Fertilisers Regulation revision ((EU) 2019/1009)** aims at addressing barriers to free movement on the internal market. The main barrier has the form of diverging national regulatory frameworks for those fertilisers currently not covered by harmonisation legislation. In addition, the Regulation revision aims at establishing a regulatory framework enabling production of fertilisers from recovered bio-wastes and other secondary raw materials. The Regulation lays down criteria in accordance with which material that constitutes waste can cease to be waste, if it is contained in a compliant EU fertilising product. Presently, waste materials such as biowaste and different types of compost and digestate that were derived from waste materials can as such become EU Fertilising Products and traded as goods on the internal market. In addition, additional conditions are being developed to enable to placement on the internal market of waste derived materials (e.g. from sewage sludge, biowaste) that have undergone precipitation, thermal oxidation, pyrolysis and gasification as a processing step.

In view of the local nature of certain product markets, the EU Fertilising Products Regulation ((EU) 2019/1009) maintains the possibility that **non-harmonised fertilisers can be made available on the market** in accordance with national law, and the principles of mutual recognition of the European Union. Hence, EU Member States can still rely on the principle of optional harmonisation to make available non-harmonised fertilisers on the market in accordance with national law when compliant with End-of-Waste requirements at national level (in line with Article 6 of the Waste Framework Directive).

Finally, it is remarked that Article 31 and 32 of Regulation (EC) No 1069/2009 enable **animal by-products of category 2** (as organic fertilisers and soil improvers) and **category 3** (as feed and organic fertilisers and soil improvers) to be placed on the market, provided that certain conditions are met. When animal by-products are used as organic fertilisers and soil improvers they must, amongst others, have undergone a hygienisation treatment (e.g. pressure sterilisation) and originate from approved or registered establishments or plants. Member States may adopt or maintain national rules imposing additional conditions for or restricting the use of organic fertilisers and soil improvers, provided that such rules are justified on grounds of the protection of public and animal health. The re-use and recovery of category 1 animal by-products is not allowed within the legislation due to health concerns (e.g. prions).

3.4.5 Distance to targets for waste

The review of existing EU legislation mostly indicates qualitative objectives, rather than future quantitative targets for waste materials. Food waste and by extension municipal biodegradable waste is the sole waste stream that is subject to the following quantitative targets:

— The EU Farm to Fork Strategy states a target of halving per capita food waste at retail and consumer levels by 2030, in line with the Sustainable Development Goals Target 12.3. Using the new methodology for measuring food waste([19]) and the data expected from Member States in 2022, it will set a baseline and propose legally binding targets to reduce food waste across the EU. The distance to target will thus be dependent on the 2022 baseline;

— The Waste Framework Directive 2008/98/EC indicates that Member States shall prepare for re-use and the recycling of municipal waste to a minimum of 55%, 60% and 65% by 2025, 2030 and 2035, respectively. Addressing municipal bio-waste is crucial for moving towards the targets on municipal waste. By 31 December 2023, bio-waste shall be either separated and recycled at source, or is collected separately and is not mixed with other types of waste of different biodegradability and compostability properties;

— Directive 1999/31/EC on the landfill of waste introduces restrictions on landfilling of all waste that is suitable for recycling or other material or energy recovery from 2030. The amended Directive ((EU) 2018/850) limits the share of municipal waste landfilled to 10% by 2035.

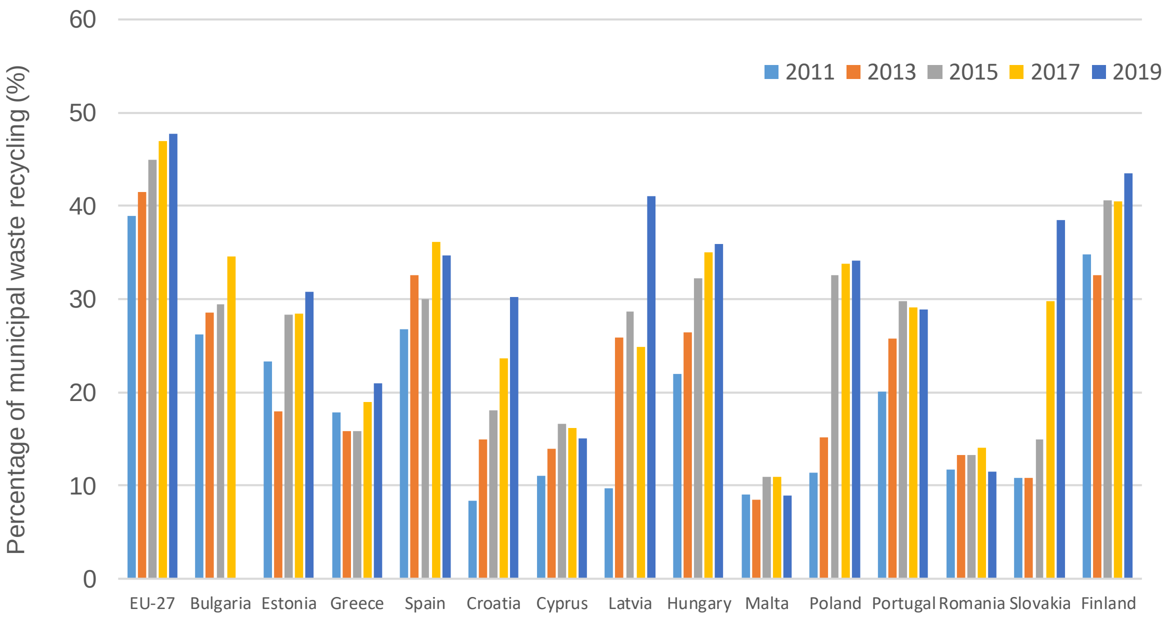
— In 2018, the European Commission published the latest implementation reports of the Waste Framework Directive([20]), giving an overview of progress and implementation challenges for several waste streams, including municipal waste. The latest Eurostat data for the year 2019 indicate an EU average recycling rate of 47.7% for municipal waste, an increase of more than 10% compared to a decade ago([21]). The latest implementations report indicated that 14 Member States were at risk of missing the (previously installed) 2020 target of 50% on separate collection. These are: Bulgaria, Estonia, Croatia, Greece, Spain, Cyprus, Latvia, Hungary, Malta, Poland, Portugal, Romania, Slovakia and Finland (Figure 22).

([19]) Commission Delegated Decision (EU) 2019/1597 of 3 May 2019 supplementing Directive 2008/98/EC of the European Parliament and of the Council as regards a common methodology and minimum quality requirements for the uniform measurement of levels of food waste (OJ L 248, 27.9.2019, p. 77).

([20]) Report from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the committee of the regions on implementation of EU waste legislation, including the early warning report for Member States at risk of missing the 2020 preparation for re-use/recycling target on municipal waste COM/2018/656 final.

([21]) Eurostat; dataset (emv_wasmun).

Figure 22. Recycling rate of municipal waste (Member States at risk of missing the 2020 target of 50% on separate collection).



While amounts of landfilled municipal waste have steadily fallen in the EU as a whole (dropping by 18% during the 2013-2016 period([22]), the average landfilling rate for municipal waste in the EU still stood at 24 % in 2016. Large differences across the EU persist: in 2016 10 Member States still landfilled over 50 % of municipal waste, while five reported rates above 70 %. According to the reported data, in 2015, half of Member States had already met the 35 % target for 2016. Croatia missed its 75 % target which was due in 2013. Cyprus, the Czech Republic, Greece, Latvia and Slovakia missed the 50 % target, also due in 2013. Malta, which has a high overall municipal waste landfill rate, has not reported recent data. A recent study carried out for the Commission([23]) found that 15 Member States were not fully meeting the obligation laid down in the Directive to treat waste before landfilling.

3.5 Monitoring progress

The EU legislation addressing nutrient losses to the environment or their impacts on ecosystems has established indicators to monitor progress towards the policy objectives (Table 11). Data on these indicators are collected by Member States with spatial and temporal resolutions that depend on the respective policy reporting cycle. Under the Nitrates Directive Member States monitor and report to the Commission the nitrates concentration in surface and ground waters and the eutrophication of surface waters every four years. In the past, there were issues related to changes in methodologies over time and discrepancies in method used by the countries (ENV pers.comm.). Information on the ecological status and chemical status of the WFD, as well as data on the descriptor eutrophication of the MSFD are reported every 6 years. The methodology for establishing the ecological status depends on the country. However, the biological classification methods were inter-calibrated across EU Member States (Birk et al. 2012; Poikane et al. 2015; 2016). Similarly, the descriptor of eutrophication of marine waters is based on different national methodological assessments (Araujo et al. 2018; Araujo and Boschetti, 2021). Recently, the new CAP has established a series of impacts indicators (Annex I) that will be monitored through annual performance reports. These indicators include soil erosion (P erosion); NH3 emissions (agriculture); gross nutrient balance (water pollution); and the share of ground water stations with NO3 concentrations higher than 50 mg/l (as per the Nitrates Directive data). However, the “data

([22]) Eurostat; dataset (emv_wasmun).
([23]) Milieu (2017), ‘Study to assess the implementation by the EU Member States of certain provisions of Directive 1999/31/EC on the landfill of waste’.

collection frequency is not always annual and there might be 2/3 years delay" (COM(2018) 392 final, Annex I).

The Gross Nutrient Balance has been considered a good proxy for nutrient pollution to air and water. However, it presents some limitations (discussion with MS, ENV report, pers.comm.). Nutrient balances estimate a risk of nutrient loss, not the nutrient losses. For example, denitrification is not be taken into consideration. Similarly, the land set aside or the presence of riparian areas are not reflected by the balance.

In the context of the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), the Expert Panel on Nitrogen Budgets, as part of the Task Force on Reactive Nitrogen, developed an integrated harmonized approach for estimating National Nitrogen Budget (NNB), including all sectors (Annex A3). The methodology is harmonised and has been applied in some European countries. However, its application requires data and expertise, and addresses only nitrogen, not phosphorus.

Table 11. Indicators in EU legislation addressing nutrient losses to the environment.

EU policy	Indicators
NECD	Emissions NOx, NH3
CAP	Impacts indicators (Annex I): • Soil erosion (P erosion); • NH3 emissions (agriculture); • Gross nutrient balance for N and P (water pollution); • Share ground water stations NO3>50mg/l (as per Nitrates Directive data)
WFD	Ecological status; Chemical status
ND	NO3 concentrations (in surface water and groundwater), eutrophication data
MSFD	Descriptor eutrophication

3.6 EU27 targets and planetary boundaries

The concept of planetary boundaries has been introduced by Rockstorm et al. (2009) to illustrate the impact of human activities on the Earth System functioning. Anthropogenic activities are destabilising the Earth Systems with severe and unpredictable consequences for the development of human society. The planetary boundaries framework proposes levels of anthropogenic perturbations below which the risk of destabilisation remains low (Steffen et al. 2015). Planetary boundaries for four out of eight key processes/features of the Earth Systems have been exceeded, including safe levels for climate change, biosphere integrity, biogeochemical flows (nitrogen and phosphorus) and land system changes (Steffen et al. 2015).

Planetary boundaries of N and P input as fertilisers have been proposed at the global scale, considering the harmful effects of nutrients for the eutrophication of aquatic ecosystems (Steffen et al. 2015). These boundaries have been downscaled considering pro-capita consumption boundaries (O'Neill et al. 2018) and regional boundaries (EEA, 2020a) (Table 12).

European boundaries for nitrogen and phosphorus were estimated using different allocation criteria (EEA, 2020a). The European losses of nitrogen and phosphorus exceed the estimated regional boundaries, by a factor of 3 for nitrogen and a factor of 2 for phosphorus (EEA, 2020a).

Recently, De Vries et al. (2021) proposed nitrogen regional boundaries for EU considering contemporary the impacts of nitrogen on biodiversity loss (critical limit: N deposition in natural areas), aquatic eutrophication (critical limit: N concentration in runoff) and drinking water pollution (critical limit: nitrates concentrations in leaching in agricultural soils). Their study highlighted that to respect all the environmental targets (critical limits), N input to the agricultural systems should be reduced up to 43% (aquatic eutrophication, being the more stringent requirement).

Planetary and regional boundaries are useful concepts to understand whether the current N and P flows in EU are within the 'safe operating space' (which is not the case) and how much possible intervention measures to reduce nutrient losses (analysed in the next Section 4) can help achieving this goal.

Rescaling the global planetary boundary of Steffen et al. (2015) for EU (based on cropland data from FAOSTAT) indicates a boundary of 4.4 TgN/y of N mineral fertiliser and intentional fixation and 0.4 TgP/y of P fertiliser input. Similar values are obtained when upscaling the nutrient per capita planetary boundary of O'Neill et al. (2018) for EU (based on EUROSTAT data on EU population in 2020): 4.0 TgN/y and 0.4 TgP/y. According to these boundaries and the values of new input of N and P estimated in this study (Section 2.1), the EU should reduce its annual mineral fertiliser input of N and P of about 60%.

Table 12. Planetary boundaries for nutrients proposed in the literature.

Biogeochemical indicator	Boundary	Description/Protection goal	Reference
Phosphorus	11 Tg P year–1 from freshwater systems into the ocean	prevention of a large-scale ocean anoxic event	Steffen et al. 2015
Phosphorus	6.2 Tg P per year from fertilizers (mined P) to erodible soils	avert widespread eutrophication of freshwater systems	Steffen et al. 2015
Phosphorus	0.06 Tg P per year (average for Europe)	Loss of P from fertilisers and waste	EEA (2020a)
Phosphorus	0.89 kg P per year (per capita boundary)	Consumption-based allocation of phosphorus from applied fertilizer	O'Neill et al. 2018
Phosphorus	0.11 kg P per year (per capita boundary for Europe)	Loss of P from fertilisers and waste	EEA (2020a)
Nitrogen	62 Tg N per year from industrial and intentional biological N fixation	Preventing eutrophication of aquatic ecosystems	Steffen et al. 2015
Nitrogen	2.1 Tg N per year (average for Europe)	Loss of N from fertilisers and waste	EEA (2020a)
Nitrogen	8.9 kg N per year (per capita boundary)	Consumption-based allocation of nitrogen from applied fertilizer	O'Neill et al. 2018
Nitrogen	3.5 kg N per year (per capita boundary for Europe)	Loss of N from fertilisers and waste	EEA (2020a)

4 Measures

Measures to reduce nutrient losses to the environment can be adopted at different intervention point of the N and P cycles. They range from technical measures for recovering and recycling nutrients in waste streams and improving nutrient use efficiency in agriculture, to policy measures at the EU level and broad societal changes, such as changes in human diet and agricultural system (food production-consumption system). This Chapter presents a (non-comprehensive) analysis of possible measures, considering both evidence from the literature and results of new modelling assessments (Figures 23 and 24). The latter were carried out adopting different modelling approaches, with an ensemble modelling perspective rather than a full integration, i.e. gathering evidence from different modelling assessments based on independent assumptions. In specific, we show:

- Nutrient recovery from manure, sewage sludge and bio-waste (Section 4.1)
- Effectiveness of measures to reduce nutrient losses in agriculture (Section 4.2)
- Scenario of reduction of atmospheric nitrogen deposition (EMEP model) (Section 4.3)
- Scenarios of reduction of nitrogen fertilisers in agriculture (DayCent model) (Section 4.4)
- Phosphorus erosion (P statistical model) (Section 4.5)
- Scenarios of reduction of nutrient losses to waters (GREEN model) (Section 4.6)
- Scenarios of changes in the agricultural system and diet (GRAFS model) (Section 4.7)

Figure 23. Modelling tools/approaches adopted in the analysis.

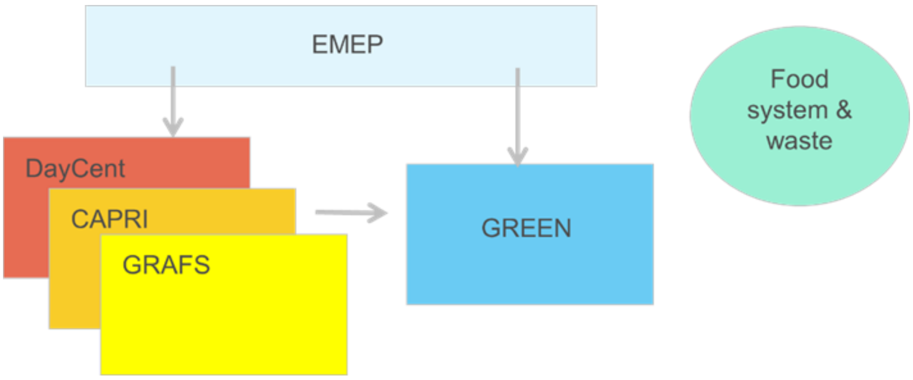
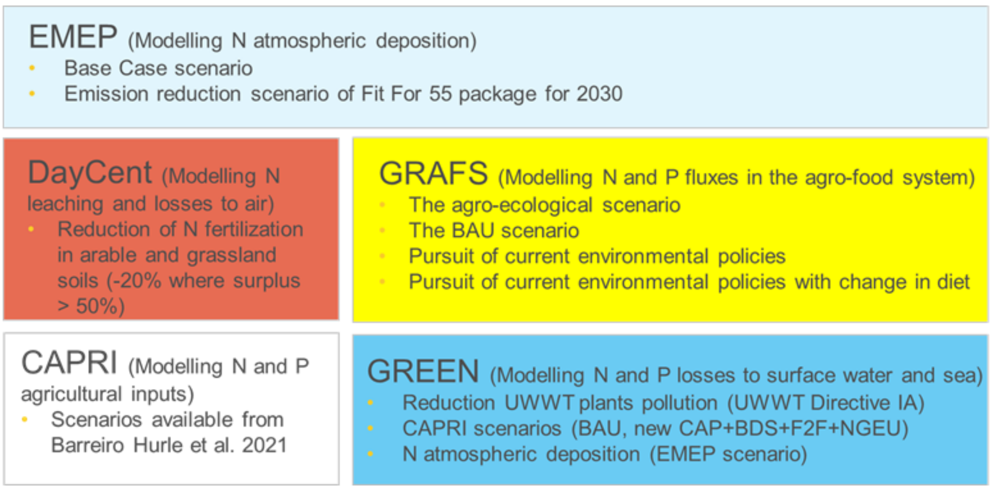


Figure 24. Overview of the scenarios of measures considered in the study.



4.1 Nutrient recovery from manure, sewage sludge, and bio-waste

Box 1. Abbreviations used in this section

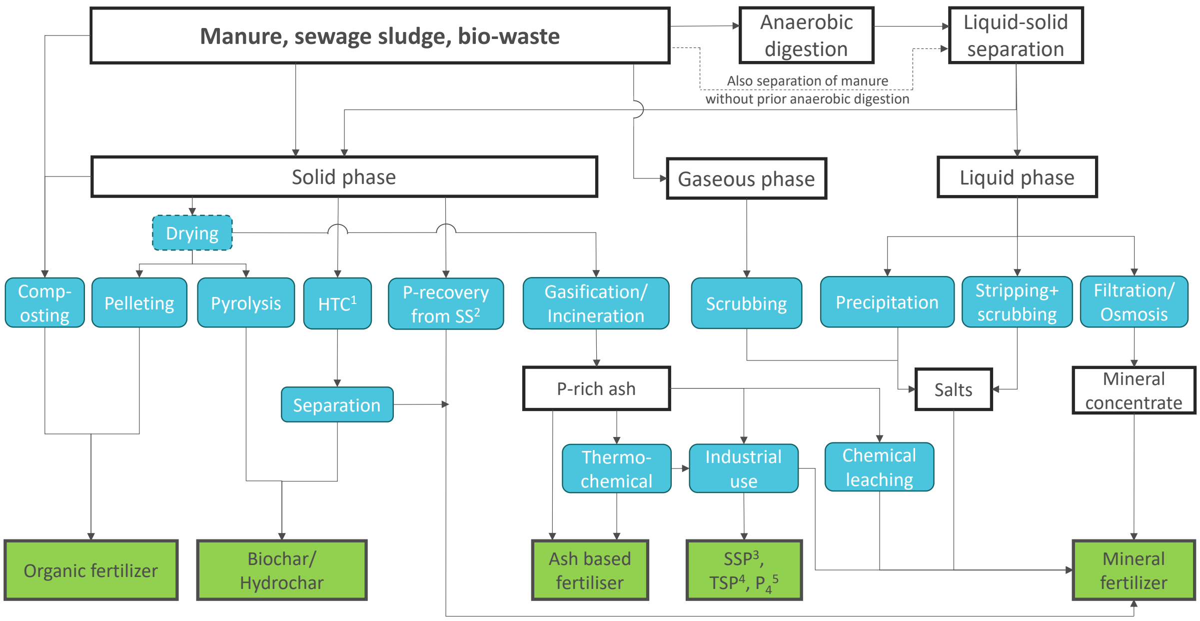
AD: Anaerobic digestion; HTC: Hydrothermal carbonisation; WWTP: Wastewater treatment plant; PE: Population equivalent; MBT: Mechanical-biological treatment plant; PO4: Orthophosphate; NH3: Ammonia; NH4: Ammonium; SS: Sewage sludge; SSA: Sewage sludge ash; CaP: Calcium-Phosphate; MAP: Magnesium-Ammonium-Phosphate; DAP: Diammoniumphosphate; SSP: Single-Superphosphate; TSP: Triple-Superphosphate; UAA: utilised agriculture area.

The growing leakage of nutrients into the environment is affecting the air and water quality, and at the same time, nutrients in waste streams as well as nutrient emissions to air are irreversible lost due to unsuitable collection, use, treatment and disposal. The three streams animal manure, municipal sewage sludge, and municipal bio-waste can offer an important contribution to improve the efficiency of nutrient management and support the EU in its transformation to a more circular economy (Buckwell and Nadeu, 2016).

To unlock this nutrient potential on the one hand improved separate collection schemes are crucial for e.g. bio-waste and on the other hand the implementation of recovery techniques.

Recovery technologies are capable of transforming biogenic waste streams into agronomical valuable organic and mineral fertilising products, and possibly remove containments at the same time. This technical overview focuses on recovery techniques of high TRL and on those currently already operating in the EU (Figure 25).

Figure 25. Overview on technologies to recover N + P from manure, sewage sludge, and bio-waste (Sommer et al., 2013, modified).



[1]HTC: Hydrothermal carbonisation; [2]SS: Sewage sludge; [3]SPP: Single Superphosphate; [4]TSP: Triple-Superphosphate; [5]P_4: Phosphorus in its purest form (e.g. white phosphorus)

4.1.1 Processing techniques

4.1.1.1 Anaerobic digestion and liquid-solid separation

Organic fractions of a low dry matter content from waste water treatment facilities (sewage sludge), livestock housing systems (manure) and other processing and collected systems (e.g. bio-waste) are often subject to anaerobic digestion (AD) to gain biogas (CH_4) and/or liquid-solid separation. During AD of organic substances, the biologically and chemically bound nutrients are partly transformed into dissolved nutrients (e.g. ammonium (NH_4) or orthophosphate (PO_4). After a mechanical liquid-solid separation of the digestate, the dissolved nutrients are available for recovery from the liquid phase. Especially for manure, liquid-solid separation is also applied without prior AD.

4.1.1.2 Liquid phase

4.1.1.2.1 Technologies to recovery N from liquid phase

Different technologies and/or technology combinations focus on recovering nitrogen in a plant-available form from this ammonium rich stream: gas-liquid stripping, chemical precipitation (e.g. struvite precipitation), ion exchange, absorption, reverse osmosis, electrodialysis, hollow fibre membrane contactor, and membrane distillation (Chen et al., 2021; Palakodeti et al, 2021; Beckinghausen et al., 2020). An excellent and detailed overview of production processes of ammonium-based fertilisers via reverse osmosis, liquid/gas separation and other techniques of lower technological readiness levels is given in Zarebska et al. 2015. The market research study within the EU funded "Systemic" projects gives a very good overview with 35 European biogas plants using different approaches to recover nutrients from liquid and solid manure streams with a high TRL level (Verbeke et al., 2021). Boehler (2018) highlights five wastewater treatment plants (WWTP) in Germany, two in Austria and four in Switzerland which recover N from the liquid phase through stripping and scrubbing technologies. Those technological approaches with noticeable high TRL level are described in detail in the following sections.

— Recovery of scrubbing salts

Full-scale application at wastewater treatment plants, manure- and bio-waste ADs in Europe mainly focus on the recovery of scrubbing salts([24]) via air- or membrane stripping and a subsequent capturing (acid scrubbing). These technologies achieve N-recovery efficiency of 80-90 % (Sigurnjak et al., 2019). The recovery technologies are energy (electricity, see Table 13) and chemical (NaOH, H2SO4) intensive. Compared to the Haber-Bosch process, which is the main industrial scale process to gain ammonia from the air, the energy demand is in the same range for air stripping, but it can be significantly higher for membrane stripping. Other approaches with a lower TRL (pilot scale) have the goal to adsorb ammonium on materials with high exchange capacity (e.g. zeolite) and could reduce the energy demand for stripping (EU Project ReNOx 2.0; Ellersdorfer et. al., 2019). In most cases, the primary objective of the implementation of nutrient recovery techniques at a WWTP is not the production of a fertiliser. The removal of N from the back flow water after the sludge digestion process has the benefit of a reduced energy demand for aeration and thus increases the overall treatment capacity.

Table 13. Energy demand for the recovery and production of 1 kg N (Boehler, 2018).

Ammonia recovery approach	**Energy demand (kWh/kg) (primary energy and electricity)**
Air-Stripping	8-13
Membrane stripping	10-27
Haber-Bosch (based on CH_4)	8

— Recovery of mineral concentrate

Filtration (micro, ultra and nano) can be used to remove suspended solids, bacteria and macromolecules from the liquid phase. Subsequently, reverse osmosis can be used to concentrate ammonium and other small compounds. The resulting concentrate is called mineral concentrate and is an ammonium containing fertiliser with a high pH (~pH 8) (Velthof, 2015; Ehlert et al., 2019). This approach is mainly applied during manure processing.

([24]) Defined throughout this report as a recovered N substance through the partial conversion of N into volatile NH3 ("stripping") followed by recapturing ("scrubbing") the extracted ammonia into soluble ammonium using a low pH solution (sulphuric acid, nitric acid or phosphoric acid to produce ammonium sulphate, ammonium nitrate, and (di-)ammonium phosphate, respectively).

— Recovery of precipitated salts

Besides scrubbing, it is also possible to precipitate N into a water insoluble form. However, precipitation processes mainly target the recovery of P as e.g. struvite (see also section 1.1.2). The potential to recover N is low because of the extent of the precipitation reaction is limited by the orthophosphate content of the liquid fraction, which is usually less that an ammonium.

— Cost

The cost to recover N are in the range of 1-10 €/kg N (Huygens et al., 2019; Fernandez and Hatzell, 2020; SYSTEMIC, 2020), being more cost efficient when the feedstock is characterised by a higher ammonium concentration (sewage sludge supernatant: up to 900 mg NH_4/L; manure supernatant: 400-4,300 mg NH_4/L pig manure and 300-3,200 mg NH_4/L cattle manure: Van Eckert et al., 2012; Menkveld & Broeders, 2018; Risberg et al., 2017). Often, this observation limits N recovery to N-rich waste streams that have been subject to anaerobic digestion (i.e. digested sewage sludge and manure). These costs are greater than current costs to purchase mineral N fertilisers produced via the Haber-Bosch process (0.5-3.0 €/kg N).

4.1.1.2.2 Technologies to recovery P from liquid phase

— Recovery of precipitated salts

Mainly precipitation technologies at WWTPs are applied to recover P from the liquid phase, with typically struvite([25]) being the phosphate salt that is obtained. Ghosh et al., 2019 and Kabbe (2021) give a good overview on the operating struvite unites on WWTP worldwide, highlighting that more than 50 full-scale struvite recovery units operate in the EU by 2021. Struvite precipitation can be combined with a subsequent ammonia recovery unit. This combination is already implemented full-scale (AV Braunschweig, 2019).

Precondition to recover P from the liquid phase is a mainly biological P-removal at the WWTP (Bio-P: surplus uptake by specific biomass). Under these preconditions, a certain percentage of P dissolves during AD due to the biological degradation processes, resulting in high PO4-concentrations of 100-900 mg/L.

Certain side stream adaption focus on the surplus release of P to increase the yield of precipitated salts (biological approach: WASSTRIP (Gysin et al., 2018); thermal hydrolysis: CAMBI (Abu-Orf, M., & Goss, T. (2014)).

Dissolved P then can be recovered in reaction tanks (fluidized bed or stirred reactors, Ghosh et al., 2019) by adding precipitants (e.g. magnesium or calcium salts) and pH-adjustment (pH > 8.5). Under alkaline conditions, dissolved PO_4 precipitates with NH_4 and/or magnesium and calcium to a water insoluble struvite and\or different forms of calcium phosphates (CaP).

More than 90 % of the dissolved P present in the liquid fraction can be recovered. However, related to the WWTP influent, the recovery rate is around 10-15 %. With optimised side stream processes (e.g. thermal hydrolysis), the recovery rate can be increased to up to 50 % (Kabbe, 2021). The recovered phosphate salts are non-water soluble, but have a proven long-term plant availability (Muys et. al. 2021).

— Cost

The operational cost to recover P from the liquid phase is in the range of 2-10 €/kg P (not taking account possible revenues; P-REX, 2016, Egle et al., 2016; Tonini et al., 2019) and thus similar (for technologies within the lower end of the cost spectrum) to substantially higher compared to raw phosphate rock (0.9-1.5 €/kg P) or already marketable mineral fertilisers (e.g. Diammoniumphosphate: 1-2 €/kg P; Triple superphosphate: ~2-3 €/kg P; Huygens et al., 2019, World Bank, 2022). However, the removal of P from the liquid phase has certain (economic) benefits for the WWTP (e.g. avoidance of clogging in pipes due to unpredictable struvite precipitation, reduction of P and N backflow to the WWTP, better dewatering properties of the sludge due to the reduced content of dissolved P; Ghosh et al., 2019).

([25]) Struvite: Magnesium-Ammonium-Phosphate (MAP: MgNH4PO4 6H2O, molar ration Mg:N:P = 1:1:1).

4.1.1.3 Solid phase

4.1.1.3.1 Direct land use

Direct use of manure and sewage sludge is the most direct way to recycle the contained nutrients. However, matching plant nutrient supply to nutrient demand is not always possible, particularly in regions of high population and livestock density. Especially during winter periods of low plant activity, there are risks with regard to the leaching of nutrients into the groundwater and surface water. In addition, concerns regarding the containing inorganic and organic contaminants (e.g. metals, pharmaceutical, hormones, and micro plastics) exist. The latter mainly exist for sewage sludge. Within the last years, the focus of attention regarding contaminants have changed from metals and pathogens to (micro-) plastics and organic pollutants (e.g. pharmaceuticals, hormones). With a direct land use, all contaminants present in the sewage sludge will enter into the soil matrix.

Bio-waste fraction can be obtained from separate collection systems or from mixed household waste by mechanical-biological treatment plants (MBT). Saveyn and Eder (2014) note, that the output of MBT is characterised with higher content of metals and visually noticeable physical impurities, for which reason EU legislation imposes separate collection of bio-waste. Typical treatment processes for the bio-waste fraction are anaerobic digestion and/or composting before soil application. The quality of the digestate or compost depends on the purity of the bio-waste but also the cleaning performance of the digestion and composting facility (e.g. sieves, sink-float processes).

4.1.1.3.2 Drying

The aim of drying processes is the reduction of mass and volume but also a certain stabilisation. This facilitates the long-term distant transport, at the expense of a somewhat lower nutrient availability. While P is chemically bound in the solid phase, during a drying process, NH_3 and consequently total N-concentrations can decrease significantly (Battista et al., 2021). Exhaust air condensates and ammonium rich vapour is produced, which could require further treatment to reduce adverse effects. The lower the share of ammonium in a fertilising material, the lower is the efficiency when used as a N-based fertiliser as ammonium is immediately available to plants.

With a subsequent pelleting, a further reduction of volume is achievable after drying and composting, reducing the cost for further handling and transporting.

4.1.1.3.3 Composting

Composting involves the mineralisation and partial humification of organic matter, leading to a stabilised final product with humic properties. Pathogen are reduced due to the high temperature in composting piles (up to 60-80 ºC) and also toxic organic substances including antibiotics can be partially degraded (Massé et al., 2014). Other unwanted substances as e.g. micro-plastics (especially present in sewages sludge) or metals are not removed. Composting helps to reduce volume and moisture content, making the material easier to handle, pelletise and to transport.

Depending on the input material and the process condition during composting (e.g. C:N ratio, water content), NH3 can be formed and lost to the atmosphere. It is even possible to recover N from this NH_3 rich air, which occurs e.g. in aerated tunnels for sewage sludge composting (Shen et al., 2020). One full-scale applied approach recovers the NH_3 via scrubbing technologies (see also Section 4.1.1.4.1). The primary goal of these techniques is the abatement of ammonia pollution, but also the reduction of odour (GMB BioEnergie, 2021).

4.1.1.3.4 Thermal transformation under reducing conditions

— Hydrothermal Carbonisation (HTC)

HTC reduces waste volume and transforms feedstock into carbonised materials. The HTC process acts as an acceleration of the natural coal formation process, working with feedstock with high water content, at moderate pressure (20-30 bar), and temperatures (200-230 ºC). Outputs are a coal slurry (hydrochar) which can be further treated (filter, dryer, pelletizing) and a water with nutrients, which in turn can be further processed by e.g. osmosis to gain a nutrient concentrate (liquid bio fertiliser). Studies show that approximately 50% of N and most of K are dissolved in the liquid product (Reza et al., 2016). However, the transfer of N into the hydrochar or the liquid phase depends on the process parameters (temperature, pressure, time) and can vary significantly (Djandja et al., 2021). Metals remain mainly in the ash fraction of the hydrochar. That is also the case for P, which can be extracted by acid leaching techniques (Tasca et al., 2019). With regard to metals, the quality of the hydrochar depends on the feedstock material (Leng et al., 2021). Several HTC industrial plants operate in Europe (e.g. Spain, UK, Italy, and Belgium).

— Pyrolysis

After pyrolysis, less than half of the original N is preserved in the char (Agar et al., 2018; Saud et al., 2021). N that is not transferred into the gaseous phase, is transformed into aromatic and heterocyclic N compounds or of low bio-availability. Phosphorus is retained in the solid material fraction, but its fertilising value is uncertain and dependent on the feedstock applied (Enders et al., 2012; Lehmann and Joseph, 2015; Huygens and Saveyn, 2018). The thermochemical conversion process produces a char-like material that is often referred to as "biochar". As with HTC, the quality of the biochar depends on the feedstock material, the process parameters and thus the further use.

4.1.1.3.5 P recovery from ashes

With thermal oxidation processes (>850 ºC, >2sec in the flue gas) a destruction of the organic substance and thus destruction of micro plastics, pathogens, and most organic pollutants can be reached. While N is lost into the flue gas, P and other nutrients (e.g. K, Mg, and Ca) end up in the ash. To avoid dilution of nutrients or contamination, a mono-incineration is preferred. P reacts with different elements from the feedstock material, mainly calcium, forming chemical compounds with poor solubility and low phosphorus use efficiency (except poultry litter ash: Huygens and Saveyn, 2018). With regard to metals, removal is possible for some metals with a low evaporating temperature (e.g. 95 % removal for Hg).

— Recovery from mono-incinerated sewage sludge ashes (SSA)

To recover P from incineration residues, the incineration should take place without other substances to produce ashes with a high P contents as well as to avoid ash contamination from other feedstocks applied.

A direct application as a fertiliser without pre-treatment is in many cases not possible due to high metals concentration and low P availability for plants. In the recent years, several technologies from existing industrial processes were adopted and further developed to recover P from SSA. The technological approaches can be grouped as follows:

— **Acidic wet chemical mixing**: Transformation of the P into a plant available form by mixing the ash with mineral acids as e.g. sulphuric or phosphoric acid. All the other compounds of the ash are fully incorporated into the fertiliser, so no removal of contaminants takes places (Full-scale implementation in Haldensleben (Germany) producing 60 kt fertiliser out of 35 kt of SSA; SERAPLANT, 2021). The fertiliser industry follows this approach to produce single-or triple-superphosphate from raw phosphate rock and could use a limited percentage of SSA to substitute raw phosphate rock (ICL Amsterdam, 2019).

— **Acidic wet chemical leaching**: Aim is the transformation of P in different uniformly usable and marketable forms (e.g. phosphoric acid, calcium phosphates). P is leached with mineral acids (e.g. hypochloric, sulphuric or phosphoric acid) and as such separated from the ash. Depending on the technological approach, metals are removed by e.g. ion-exchange, liquid-liquid separation or precipitation. Certain technologies also aim for the recovery of iron- and aluminium as iron- and aluminium salts, which can be used as by-products (e.g. as coagulants at WWTP). Due to the specific removal processes, the P-rich output materials contain less contaminants. A recycling plant for 20 kt of SSA is already installed in Hamburg (TetraPhos®, Remondis, Rak, 2018), and further full-scale implementations are expected within the next 5 years (e.g. Ash2Phos (Easymining, 2021), Phos4Life, City of Zurich (Schlumberger, 2017).

— **Thermo-electric**: Certain technologies focus on the production of phosphoric acid of even white phosphorus (P4) with technologies operating at temperatures above the ash melding point (> 1.500 ºC). Under this condition, P enters into the gaseous phase and can be recovered (Rapf and Raupenstrauch, 2021). Recovery of P in industrial scale was done by Thermphos (Schipper, 2001), but the plant ceased its activities for unknown reasons.

— **Thermo-chemical**: Aim of this approach is the partial removal of metals and the transformation of the P into a better plant available form. This can be achieved by adding Cl and a treatment temperature of 750 ºC to 1,000 ºC (below ash melting temperatures: Adam et al., 2009). Latest developments of this technology focus on the further improvement of the plant availability of SSA by adding Na instead of Cl, with the trade-off of significantly lower metal removal (Herzel et. al., 2021).

- **Bioleaching and bioaccumulation of P**: The process to recover P via bioleaching consists of three main steps; 1) Bioleaching of P and metals by specific microorganisms creating an acid environment (leaching); 2) Accumulation of the dissolved P by microorganisms; 3) Microbial induced P precipitation (Zimmermann and Dott, 2009). This approach is state-of-the-art to recover e.g. Cu from ore, however is not applied on SSA so far.
- The cost to recover P from SSA is in the range of 1-5 €/kg P. By taking into account possible revenues for the products, the cost to recover P is thus close to or higher than the cost of producing a mineral fertiliser produced from raw phosphate rock.
- The P recovery efficiency of these technologies is in the range of 85 % (wet chemical leaching with targeted removal of contaminants) to even 100 % for approaches with the goal to only improve P plant availability but no removal of contaminants.
- Recovery from manure ashes

In principle, the incineration of different forms of manure is possible, but so far, mainly incineration of poultry manure is observed. The higher dry matter content poultry manure compared to swine or cattle manure enables a net positive energy recovery from this material. The ash of poultry manure contains valuable minerals such as P (12-13 %) and K and is homogeneous since the feedstock is of consistent quality (Ehlert, 2020; Adamczyk et. al., 2021). In 2010, 3 countries incinerated poultry manure (4 % of total poultry manure incinerated in Ireland, 30 % in the Netherlands and 36 % in the UK; AMEC 2014).

4.1.1.3.6 P recovery from raw and dewatered sewage sludge

In the recent years, several technologies were developed with the specific objective to recover P from sewage sludge, either before or after sludge dewatering. Literature shows complex technological approaches to recover P directly from sewage sludge (e.g. acidic wet chemical leaching, wet oxidation, super critical water oxidation, metallurgic melt-gassing; Kabbe, 2021) however, none of these technologies has reached full-scale application.

The cost to recover P from sewage sludge is significantly higher (> 10 €/kg P; P-REX, 2016, Egle et al., 2016) than other approaches addressing (e.g. from SSA, or through precipitation). The high cost is the result of complex technological processes but even more the need for great amount of chemicals and energy. Possible revenues from the produced P materials can by far not cover the operational cost. The P recovery efficiency of these technologies is in the range of 60 % related to the sewage sludge input.

4.1.1.4 Gaseous phase

4.1.1.4.1 N recovery from off-gases from stables and manure storage/processing facilities

Emissions from manure are responsible for a large share of the total ammonia emissions to air in the EU (3.5 Mt N yr^{-1}), with housing and storage being the main stages of emissions. Scrubbing technologies enable to recover 85 % of the ammonia present in exhaust gases. Ammonia scrubbers remove pollutants via a chemical reaction using strong acids (typically sulphuric acid, but also phosphoric- and nitric acid can be used) to neutralize the ammonia. Depending on the technology applied, the resulting material contains nutrients in plant-available form (e.g. following forced oxidation techniques). This byproduct from the gas cleaning system, a pure salt, is then collected and subsequently used as a fertiliser. The application of an air cleaning system is listed as one of the Best Available Techniques (BAT) that applies to the intensive rearing of poultry or pigs (Giner et al., 2017). Nonetheless, it is indicated that this technique may not be generally applicable to existing plants due to technical and economic considerations. Costs of 5-15 €/ kg N recovered have been indicated (ECE, 2014).

4.1.1.4.2 Non-recoverable nitrogen losses

Off-gas emission and pollution control systems

Nitrogen oxides (NO_x consists mainly of NO, NO_2) are formed in the combustion process of fossil fuels, biomass or waste. Modern power plants reduce NOx emission by injecting ammonia or urea at temperature of 760 and 1,090 ºC (SNCR - Selective Noncatalytic Reduction Process or SRC – Selective catalytic reaction). This results in a chemical transformation of NOx into N2, CO2 and H2O and the nitrogen is irretrievable lost into the air (Vehlow, 2013). SNCR has demonstrated NOx reductions of about 40-70 %, whereas SCR performs significantly better with removal efficiencies ≥ 80 % (Rogoff and Screve, 2012; Sarkar, 2015).

The BAT reference document for waste incineration (Neuwahl et al., 2019) lists techniques to decrease NO_X emissions:

— For SCR: Use of a larger catalyst surface, installed as one or more layers. 'In-duct' or 'slip' SCR combines SNCR with downstream SCR, which reduces the ammonia slip from SNCR;

— For SNCR: The performance of the SNCR system can be increased by controlling the injection of the reagent from multiple lances with the support of a (fast-reacting) acoustic or infrared temperature measurement system so as to ensure that the reagent is injected in the optimum temperature zone at all times;

— Flue gas recirculation: Recirculation of a part of the flue-gas to the furnace to replace a part of the fresh combustion air, with the dual effect of cooling the temperature and limiting the O_2 content for nitrogen oxidation, thus limiting the NO_X generation.

N recovery from flue gas

Certain patents describe technological approaches to recover N as pure N2 or as NH3 from the flue gas.

- Process and system for the recovery of ammonia when separating nitrogen oxides from flue gases (EP 0 264 041 A2; Frey, 1986)
- Adsorptive process for recovering nitrogen from flue gas (US 4988490; Nicholas et al., 1991)
- Separation of carbon dioxide and nitrogen from combustion exhaust gas with nitrogen and argon by-product recovery (EP 0 469 781 A2; Krishnamurthy and Andrecovich, 1992)

However, no full-scale implemented technology can be found in literature.

N recovery from condensate treatment

In cases where a wet scrubber is used for acid gas abatement, and in particular with SNCR, unreacted ammonia can be absorbed by the scrubbing liquor and, once stripped, the ammonia rich stream could be recycled as SNCR or SCR reagent (Neuwahl et al., 2019).

Biomass power plants with dry and a subsequent wet separation could recover ammonium salts through a five-step treatment cascade. First, the flue gas condensate passes two filtration steps (micro- and ultrafiltration), reverse osmosis, ion exchange and finally a transmembrane chemisorption to produce ammonium salts. The gained salts can be used again to reduce CO and NOx emissions (Liqui CelTM; 3M, 2016).

For techniques, which address the condensate of flue gas treatment, the entrapped N is not present in a plant available form and therefore cannot be recovered on agricultural land, even not after further processing. Therefore, the following section on impacts of measures does not focus on the recovery of N following combustion and incineration processes.

4.1.1.5 Potential to remove contaminants

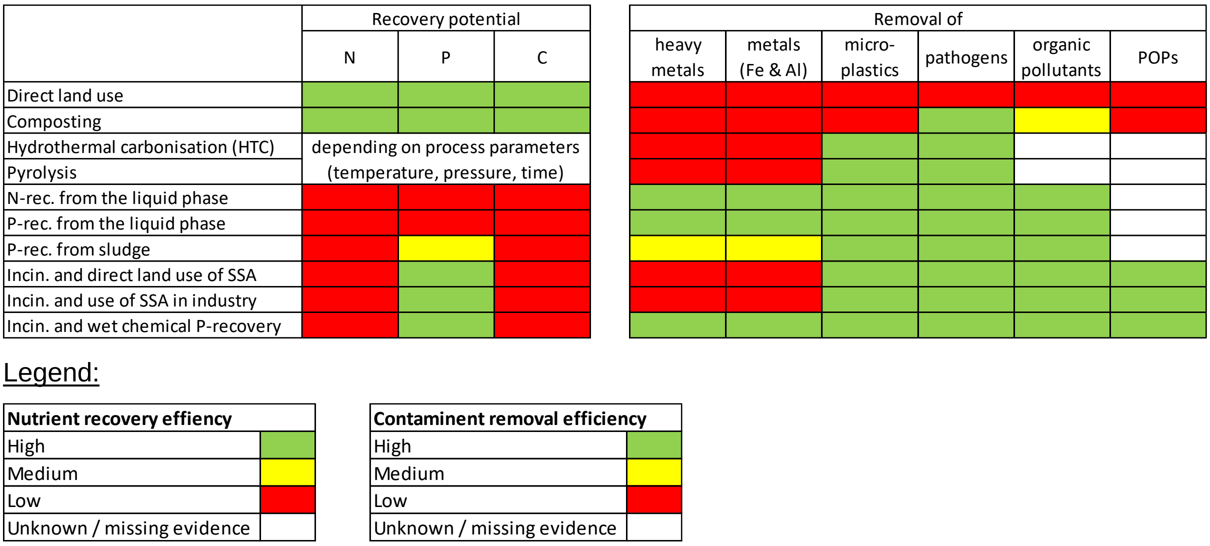
The agricultural use of waste streams is closely linked to discussion(s) about the potential harmful substances that has gained force in view of the recent Commission Zero Pollution Action Programme. The challenge for technologies is to align the efficient recovery of nutrients from complex and inhomogeneous waste stream with the demand for clean products. This section offers a qualitative assessment on the recovery and removal performances of the different technological approaches.

The transformation of wastewater burden streams but also manure into chemically new materials as e.g. ammonium salts (Huygens et al., 2020), phosphate salts (Foletto, 2013; EC, 2016; Huygens et al. 2019) and phosphoric acid or P salts from sewage sludge ashes (Amann et al, 2018) guarantees clean products with a significant removal of contaminants and unwanted substances as e.g. iron and aluminium (Figure 26).

In principle, the incineration of organic material guarantees the destruction of the overall share of organic contaminants. Combined with selective recovery technologies also metals can be removed. Some technologies allow even the recovery of e.g. iron and aluminium salts from materials generated at waste water treatment plants (Kabbe, 2021). Other technologies (e.g. direct land application of sludge, direct substitution of rock phosphate with sewage sludge ashes in present-day P-fertiliser production processes), do not result in the removal of metals and other compounds, thus finally involving their return to agricultural soils.

For technologies, operating under limited oxygen level (HTC, pyrolysis) the fate of nutrients varies depending on the process parameters (temperature, pressure, time: Meesuk et al., 2013, Djandja et al., 2017). Regarding the transformation and/or removal of the organic pollutants and POPs, further investigations are necessary.

Figure 26. Recovery potential and removal of contaminants (SSA: sewage sludge ashes).



4.1.2 Description of scenarios

4.1.2.1 Limitations and uncertainties on scenario analysis

The main objective is to assess the potential impact of the implementation of state-of-the-art nutrient recovery technologies from currently dissipated nutrients originating from sewage sludge, manure and bio-waste. Different scenarios were developed for each stream that altogether provide a first estimate to budget the potential of nutrient recovery from sewage sludge, manure and bio-waste to close nutrient cycles. The scenarios take into account (i) the state and limitations of technologies (e.g. recovery efficiencies), (ii) the estimated future available feedstock for recovery processes, and (iii) the estimated implementation potential in the EU settings (e.g. limit ammonia scrubbing to large stables).

The number provided are best estimates, associated to significant uncertainties, and should by no means be interpreted as a final outlook on the potential of these technologies. The reasons therefore relate to (i) the impossibility to forecast technological and the legal framework e.g. on pollution prevention and waste management, and (ii) uncertainties to the techno scientific information base that is applied in this exercise. Rather, they should be interpreted as a proxy to estimate the overall contribution of recovery and recycling technologies to contribute to the overall objective to reduce nutrient losses and mineral fertiliser applications.

In order to be able to determine the impact of the different scenarios and get the dimension right, the recoverable nutrients from each scenario are compared with the annual mineral fertiliser consumption. According to EUROSTAT (2020) and Fertilisers Europe (2019), the annual fertiliser consumption in the EU consumption is 10.2 Mt N and 1.1 Mt P, respectively.

4.1.2.2 Nutrient contents in feedstocks

The annual sewage sludge production is 7.2 Mt of dry solid per year (data from 2016) (EUROSTAT, 2017). Assuming an average content of 3.15% N and 1.8% P, sewage sludge contains about 226 kt N and about 129 kt P. These contents may slightly (e.g. by 10-15%) increase in case of more stringent quality measures for waste water treatment effluents, for instance for smaller waste water treatment plants. In 2018, around 35% of sewage sludge was applied in agriculture and 12% was composted or applied differently. 17% was classified as ‘other use‘, mostly used outside of agriculture (e.g. backfilling, in forestry). Consequently, 40% of N and P from sewage sludge is irretrievable lost (landfill: 10%, co-incineration without P-recovery: 30% (EUROSTAT, 2021b).

Foged at al. (2011) highlight that in the EU-28 (UK included) annually about 1.4 billion tonnes of livestock manure is produced. The 1.4 billion livestock manure correspond to 7.2 Mt of N and 1.8 Mt of P per year (Foged et al., 2011), of which approximately 67% is collected for possible manure processing (Asman et al., 2011; De Vries et al. 2021). For the reference year 2010, less than 8% of the total manure produced underwent processing as e.g. liquid-solid separation, AD, treatment of the liquid or solid fraction. The application of technologies which produce ammonium salts or P-rich salts struvite are limited to a very few plants.

Calculating with an average municipal waste generation of 487 kg/inh/y (EUROSTAT, 2021a), 447 Mio. inhabitants, and a current average share of 37% bio-waste in the total municipal waste generation, the total mass of bio-waste is estimated at around 80 million tonnes per year. In addition, about 40-45 million tonnes per year of bio-waste is produced by the food industry (Buckwell and Nadeu, 2016). Assuming a dry matter content of 35%, an N content of 2.5% and P content of 0.5%, about 0.95 Mt N and 0.19 Mt P would be present in bio-waste.

4.1.2.3 Selection of scenarios

This analysis focusses on following promising technologies for nutrient recovery, which can be broadly classified into four groups:

- Mineral N and P fertilisers recovered from liquid digestates though stripping/scrubbing and precipitation;
- Mineral P fertilisers recovered from biomass ashes;
- Mineral N fertilisers recovered via scrubbing exhaust air from stables, storage facilities and composting plants;
- Organic fertilisers obtained following composting and/or anaerobic digestion.

It is noted that three out of four scenarios involve the recovery of mineral fertilisers, whereas only a single scenario focuses on the recovery of organic nutrients and organic matter. As outlined in the section on flows, nutrient losses and excesses in soils occur in some (mostly Western) EU regions (e.g. the Netherlands, Germany), mainly because of the significant livestock manure generated and applied locally. Other EU regions are characterised by a lower livestock density and thus more neutral gross nutrient balances. Therefore, nutrient-dense mineral fertilisers are more suitable to transfer excess nutrients from specific EU region to another one without excessive transport cost burdens. Organic fertilisers of a lower nutrient-density and dry matter content are more suitable to be applied locally close to their place of generation, and therefore have a lower potential to address nutrient excesses at regional EU level.

4.1.3 Results

4.1.3.1 Nutrient recovery potential

4.1.3.1.1 Mineral N and P fertilisers recovered from liquid digestates through stripping/scrubbing and precipitation

Anaerobic digestion is the door-opener for separation processes and nutrient recovery from manure and other biogenic materials (Foged at al., 2011). Importantly, it transforms an important share of the nutrients into directly plant-available mineral N, and can thus increase the fertilising value of the feedstock. As part of the recent Joint European action for more affordable, secure and sustainable energy (REPowerEU), it is aimed to double anaerobic digestion from biogenic waste materials. Therefore, it is assumed that significant shares of manure, bio-waste and sewage sludge will be digested:

- For manure we assumed that up to 20% of the total manure, equalling about 1.45 Mt N yr^{-1}, 0.46 Mt P yr^{-1}, will be digested. The nutrient amounts for manure correspond to approx. 40% of only a few selected MS (Germany, the Netherlands, Denmark, Belgium) characterised by higher nutrient excess, and seem therefore plausible;

- Given the high methane production potential of bio-waste and legislative requirement on the separate collection of bio-waste, it is assumed that up to 75% of the available bio-waste can be digested. Hence, 0.71 Mt N and 0.14 Mt P would be digested annually in such scenario;
- For sewage sludge, it is assumed that 70% of the sewage sludge produced (0.15 Mt N yr^{-1}, 0.10 Mt P yr^{-1}; corresponding to sewage sludge produced at WWTP with a treatment capacity ≥ 50,000 PE (OEWAV, 2016)) will be digested.

Subsequently, it is assumed that N stripping/scrubbing processes are applied at plants that process 50% of these feedstocks, and that 30% of the N in the manure/waste stream is ammonium-N that can be scrubbed from the digestate at an efficiency of 85%. Alternatively, reverse osmosis could be applied to obtain a mineral concentrate.

Incorporating this technology would produce 0.34 Mt N yr^{-1}. Costs for these technologies have been estimated at 2.5 - 10 billion €/Mt N recovered [11]. In addition, about 20% of the P can be precipitated from these streams, e.g. as calcium phosphates or K-struvite. This enables recovering 0.14 Mt P yr^{-1} under this scenario, at an estimated cost of 2-3 billion €/Mt P recovered [12].

It is noted, that without a process in place to recover nutrients in mineral form, some of these nutrients may also end up on agricultural land as organic fertilisers (digestate). Still, the transformation of nutrients into mineral fertilisers may increase fertiliser efficiency, and facilitate enhanced management and long-distance transport to areas characterised by a higher nutrient demand, likely resulting in an overall higher nutrient use efficiency. On top, new feedstocks for organic fertiliser (e.g. from separately collected bio-waste) are now becoming available (see scenario 4 that may compensate the loss of nutrients that are stripped and precipitated as described above).

We also refer to the RENURE project report (Huygens et al, 2020) that evaluated the potential of the scrubbing salts (as well as other materials with a high mineral:N or low TOC:N ratio) to be considered as manure-derived nitrogen (N) fertilisers that can be used in areas subject to the ceiling of 170 kg N/ha/yr prescribed in Annex III of the Nitrates Directive (91/676/EEC). Based on agronomic efficiency data and considering supplementary measures (e.g. to avoid NH_3 losses upon RENURE application on agricultural land, appropriate storage conditions), it was confirmed that scrubbing salts and materials of a similar chemical composition show a N use efficiency similar to mineral fertilisers. This demonstrates the potential of this measure to effectively reduce N losses through manure processing, and to substitute Haber-Bosch derived mineral N fertilisers by manure-derived materials.

4.1.3.1.2 Mineral P fertilisers recovered from biomass ashes

Increasing amounts of sewage sludges, poultry litters, and pulp are being incinerated and combusted. Phosphorus is currently already being recovered from P-rich sewage sludge mono-incineration ashes and poultry litter ash. This scenario assumes that mono-incineration followed by the transformation of the ashes into a mineral P fertiliser will become the default route for sludge that is currently disposed and used in agriculture (75 % of the generated sewage sludge). This scenario is aligned to e.g. future legislative requirements in Germany and Austria that would prohibit sludge use on agricultural land due to (i) concerns on contaminants that may cause a risk for human health and the environment, and (ii) the relatively low P bioavailability in sludge, especially when strongly bound to Al/Fe complexes. In addition, significant amounts of poultry litter are currently already being incinerated, mainly for energy production in MS characterised by a high P surplus (e.g. the Netherlands). In 2015, a total amount of 0.15-0.20 Mt of poultry litter ash was produced, with an estimated P content of 0.02 Mt P. In view of an expected increase in renewable energy production from solid biomass, Huygens et al. (2019) projected a further increase in the amounts of solid manures and poultry litters that will be incinerated with energy recovery. Innovative techniques now enable to recovery P in mineral form such ashes.

This scenario assumes that 75% of the sewage sludge (0.10 Mt P) and increasing amounts of poultry litter (0.04 Mt P) will be incinerated, generating biomass ashes with a total P content of 0.14 Mt P. Nitrogen is lost during the incineration process, but the P present in the ashes can then be recovered (technology efficiency with regard to input material: 90%), with possibilities for the removal of contaminants. In total, these processes have a potential to recover about 0.13 Mt P yr^{-1}. The recovery of mineral P salts can be performed at a cost of about 1.5-2.5 billion €/Mt P recovered (Egle et al., 2016; Tonini et al., 2019).

4.1.3.1.3 Mineral N fertilisers recovered via scrubbing exhaust air from stables, storage facilities and composting plants

Manure from livestock farming is responsible for more than 70% of the total ammonia emissions in the EU (3.5 Mt N yr^{-1}). Housing and storage are the main stages in the manure chain that cause ammonia emissions (50% of all manure-derived emissions). Exhaust gas scrubbing technologies enable to recover 85% of the ammonia present in exhaust gases. In case such technologies were to be additionally implemented in 50% of the stables and storage facilities, 0.52 Mt N yr^{-1} could be recovered. In addition, ammonia can also be scrubbed at housed composting facilities that process separately bio-waste and other solid digestate fractions (estimated supplementary recovery potential of 0.15 Mt N yr^{-1}). Costs of 5-15 billion €/Mt avoided N loss (displaced mineral N import) have been indicated (ECE, 1014).

4.1.3.1.4 Organic fertilisers obtained following composting and/or anaerobic digestion

— Based on manure processing date for the year 2010, it can be estimated that maximum about 0.58 Mt N and 0.14 Mt P ended up as composted and/or digested manure on agricultural land. Presently, it is estimated that, through the production of digestate and compost from 47.5 Mt (40-50%) of separately collected bio-waste, over 0.13 Mt N and 0.04 Mt P and 3.5 Mt of organic carbon were recycled in the 18 European countries (ECN, 2019). In addition, composted and sewage sludge are also applied on agricultural land, although these numbers are likely much smaller (<0.10 Mt N; < 0.4 Mt P).

— Increasing separate collected of municipal solid bio-waste and the ban of landfilling for biodegradable materials could further increase the amounts of organic materials that can be composted and/or digested for the production of organic fertilisers. At the same time, nutrient losses may occur, especially during the compositing process that causes a loss of ~30-40% of N, including ammonia (Wong et al., 2017; Yang et al., 2019; Witter & Lopez-Real, 2013). Note that particularly at larger plants, emission control systems may be in place that can capture these losses and turn them into a fertilising material (see section 4.1.3.3). Nutrient losses are minimal during anaerobic digestion, a process that transforms part of the organic N into more plant-available mineral N. Both processes also remove a substantial part of the organic carbon (40-70%) present in the feedstock material

— For manure it is assumed that manure processing will increase from 8% to 20% of the total manure, leading to additional possibilities to recover about 0.84 Mt N yr^{-1} and 0.28 Mt P yr^{-1}, e.g. as a digestate or compost material. It is noted that presently manure is anyway returned to agricultural land, and therefore double-counting of nutrient streams is to be avoided. Nonetheless, turning excess manure fractions into a drier and hygienised material may improve the handling and efficient use of the material.

— At present about 10% of the generated sewage sludge is landfilled, causing a loss of 0.02 Mt N yr^{-1} and 0.01 Mt P yr^{-1}.

To unlock the nutrient potential of bio-waste, source segregation is necessary to gain a clean bio-waste fraction and to avoid the incineration or landfilling of the bio-waste fraction with the municipal solid waste. According to an EEA survey (EEA, 2020) the separate collection rate of bio-waste is 50% (ranging from 10% to 85%). To reach the ambitious 65% recycling rates in 2035 for municipal waste according to the Waste Framework Directive (2018/851), the separate collection of organic waste is a key element to reach this goal. It is assumed, that separate collection of bio-waste increases from the current 50% to 65% by 2035 mainly due to improved collection infrastructure (e.g. additional collection opportunities, door-to-door collection) and implementation of additional economic instruments on e.g. incineration or landfill tax to promote recycling. An increase in separate collection of bio-waste from 50% to 65% results annually in an additional 12 million tonnes of organic waste (containing 0.09 MT N yr^{-1}; 0.02 Mt P yr^{-1}) that is available for anaerobic digestion and/or composting.

If organic materials were re-allocated to biological treatments plants (e.g. anaerobic digestion followed by composting; with an assumed loss of 35% of the incoming N), 0.67 Mt N yr^{-1} and 0.31 Mt P yr^{-1} could be returned to agricultural land as a value-added organic fertiliser.

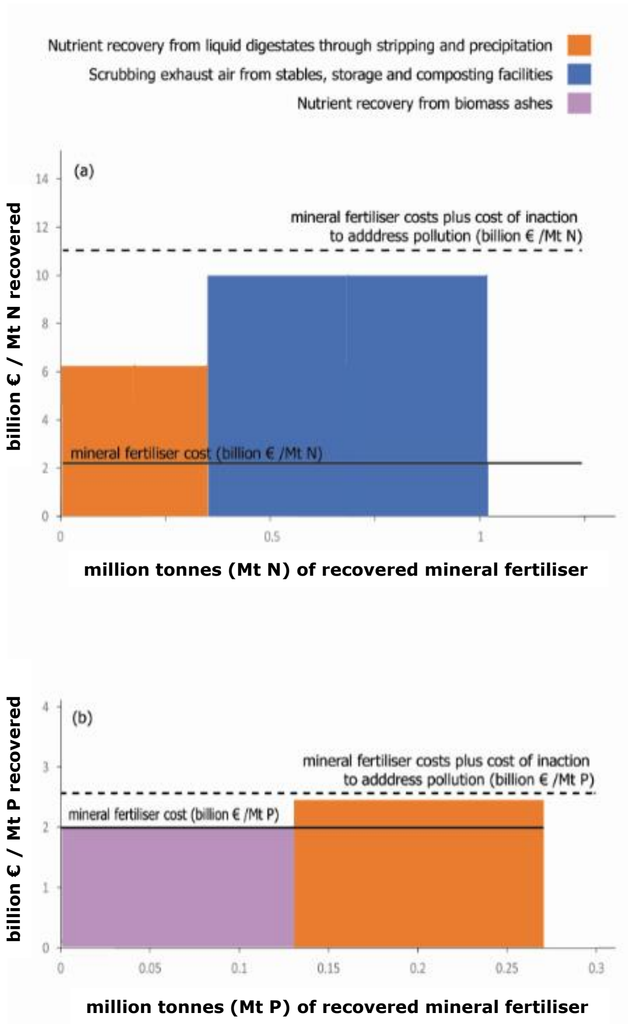
4.1.3.2 Opportunity costs

Three pathways have been identified that transform nutrients in waste and other dissipated nutrient streams (e.g. emissions to air) into mineral N and P fertilisers. In total, about 1.02 Mt N yr^{-1} and 0.27 Mt P yr^{-1} can be recovered as mineral N and P fertilisers, respectively (Figure 27). According to EUROSTAT (2020) and Fertilisers Europe (2019), the annual fertiliser consumption in the EU consumption is 10.2 Mt N and 1.1 Mt P, respectively. Hence, the proposed measures, may enable to substitute up to 10% and 25% of the N and P annually used in mineral fertilisers, respectively.

Actual mineral fertiliser acquisition costs are in the range of 1.5-3 billion €/Mt N and 1-3 billion €/Mt P (World Bank, 2022). At current fertiliser prices, some measures are cost-effective as they not only reduce fertiliser imports, but also production costs for farmers or waste managers. These measures may already reduce mineral N and P demands by 0.34 Mt N yr-1 and 0.14 Mt P yr-1, respectively (Figure 1).

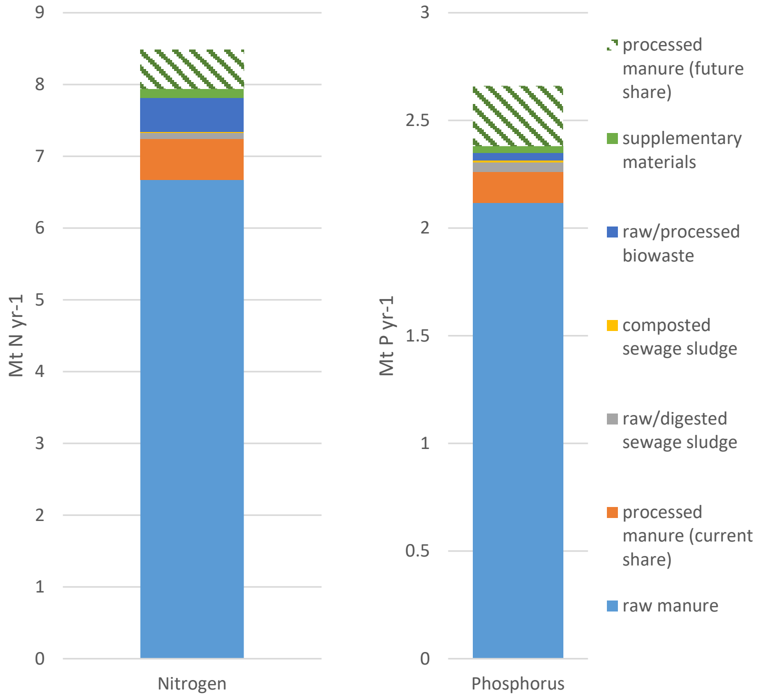
Importantly is that the measures will also proportionally reduce N and P losses of 1.02 Mt N yr^{-1} and 0.27 Mt P yr^{-1}. Avoided emissions include losses of ammonia (0.67 Mt N yr^{-1}), and N and P losses in case of material disposal and suboptimal use of digestates on agricultural land (i.e. not aligned to plant nutrient demands in application rates or timing). Hence, some of the measures not only produce new fertilisers, but may also abate pollution. This is particularly relevant as, for instance, emissions of 1 Mt ammonia-N to air cause societal damages that are equivalent to 2-20 billion € (Brink et al., 2011). To be cost-beneficial to society, the cost of the measure should be lower than the sum of its economic implementation costs and external costs in the absence of its application (Figure 27). Ammonia emissions to air (average 11, range 2–20 billion €/Mt N) and supplementary nitrate losses to water bodies from digestates relative to variable rate fertilisation with mineral N fertilisers (average 0.3; range 0.1–0.6 billion €/Mt N; assuming on average of 30% additional losses to waters in case of inaction) are considered. For phosphorus, external costs involve supplementary phosphate losses to water bodies from digestate relative to variable rate fertilisation with mineral P fertilisers (average 0.7; range 0.2 – 1.2 billion €/Mt P; assuming on average of 30% additional losses to waters in case of inaction) (Brink et al., 2011; CE Delft, 2017). All measures proposed here are thus cost-beneficial to society.

Figure 27. Marginal mineral fertiliser recovery cost curve. Histogram of recovered nitrogen N (a) and phosphorus P (b) fertiliser and net marginal costs associated with each of these measures. Stepping up on emerging circular economy actions, the graphs rank the future potential of the measures to recover mineral fertilisers from most cost-effective measures (left-hand side) to most cost-prohibitive measures (right-hand side) on the X-axis. The bar width indicates the estimated recovery potential of the individual measures. The estimated (average) cost of the measures, expressed as billion € per Mt N and P displaced, is indicated on the Y-axis. Cost-effective measures cost less than the mineral fertiliser cost (solid horizontal black line). However, when the cost-benefit analysis also considers the benefits of the measure to reduce external costs (e.g. from ammonia emissions to air), measures that are cost-beneficial from a societal perspective are positioned below the dotted black horizontal line.



The fourth measure, focused on diverting organic-rich biogenic waste streams from disposal options by returning them agricultural land, may also have an effect on nutrient recycling and recovery (0.67 Mt N yr^{-1} and 0.31 Mt P yr^{-1}). These numbers are small relative to the current return of organic nutrients, e.g. present in manure, bio-waste and sewage sludge (Figure 28). In addition to nutrients, these materials also contain organic matter, a key component of soil. Organic fertiliser as e.g. compost deliver stable organic matter to soils, which is linked to soil health and as a further consequence results in less soil erosion, less leaching of nutrients as well as improved plant yield (Celik et al., 2004; Diacono and Montemurro, 2009). Repeated applications of good-quality organic fertilisers can improve the soil's ability to retain water and nutrients and to store carbon, as well as raising its fertility, particularly in soils of low organic carbon content in the EU. However, the improvements are strongly linked to the local situation (Hijbeek et al., 2017). In addition, organic fertilisers and soil improvers (e.g. compost, digestate) are less suitable for long-distance transport, and therefore have a reduced potential to address nutrient-excess at regional level. As a result, nutrients in certain organic-rich materials (e.g. sewage sludge, poultry litter) are being “destroyed” in some nutrient-rich EU regions as a waste management strategy.

Figure 28. Estimated amounts of nutrients presently contained in raw and digested manure, sewage sludge and bio-waste. The supplementary nutrients amounts that be returned to agricultural land in this scenario are indicated in green at the top of the bars. The diagonally striped blocks on top refer to the future share of processed manure, considered to involve a transformation of materials that are currently applied on agricultural land as raw manure.



4.1.4 Conclusions

Novel recycling techniques capture and transform N and P from organic waste into nutrient-dense concentrated and safe (mineral) fertilisers, and may enable to transfer nutrients from nutrient-excess to nutrient-demanding EU regions. Such actions are able to substitute about 10% and 25% of the N and P mineral fertilisers, respectively. In addition, increased efforts to collect and re-use current discarded biogenic materials may contribute marginally to make available supplementary amounts of organic fertilisers. The proposed measures to increase nutrient circularity will involve additional private costs to operators involved, estimated at about 6 billion € yr^{-1} because manufacturing fertilisers from secondary raw materials is generally more expensive compared to production process from primary sources. However, their application is rationalised and cost-beneficial to society because of (i) the additional cost savings incurred by third parties (e.g. citizens that experience better air and water quality due to reduced nutrient losses; leading to estimated total savings in external costs of more than 7 billion € yr^{-1}), and (ii) the reduced need to deplete fine raw materials, more particularly rock phosphate. This also explains why these measures are already applied in specific local contexts. The challenge is now to scale up the implementation of the measures to reach their full-scale potential as simulated in this document, as well as to ensure an equitable distribution of costs between parties.

4.2 Impacts of agricultural farming practices on environment & climate - systematic literature review

Under the IMAP4Agri[26] Administrative Arrangement the Joint Research Centre provides scientific support and tools to DG Agriculture and Rural Development for implementation, monitoring and evaluation of the post 2020 CAP

In this context the JRC Food Security Unit is conducting an extensive systematic literature review of farming practices in order to better understand the impacts of these farming practices on the environment and climate change. This initiative will support Member States to better program their interventions, quantify their results and link them to the CAP objectives.

To understand the implications of different farming practices in a scientifically robust manner, a large amount of data is synthesized to assess whether farming practices have positive or negative effects on the environment, climate and productivity. Starting point is the relatively large number of meta-analyses (MAs) published in agricultural science. A meta-analysis is the systematic statistical synthesis of the results of many independent individual experiments. Therefore, MAs allow to explore general trends beyond the context-dependence of large numbers of experimental studies and identify key moderating factors. (Makowski et al., 2021). More details regarding the methodology and the results of the meta-analysis are available on the **public WIKI webpage** ([27])**.**

Currently (status 12.08.2022) the WIKI provides information on the effect of 15 main farming practices, which are:

- Sustainable fertilisation practices
- Agroforestry
- Organic farming systems
- Soil amendment practices
- Pesticide reduction strategies
- Landscape features
- Fallowing
- Manure management techniques
- Livestock housing techniques
- Livestock dietary manipulation techniques
- Intercropping
- Crop rotation
- Grassland
- Cover and catch crops
- No tillage and reduced tillage

In some cases, the farming practices may be further sub-divided into more specific practices e.g. “Sustainable fertilization practices” comprises the specific practices:

- Nitrification Inhibitors, Enhanced-efficiency fertilisers
- Green manure
- Organic Fertilisation
- Low-ammonia emission techniques for mineral fertilisation

([26]) IMAP = ***I***ntegrated ***M***odelling platform for ***A***gro-economic resource ***P***olicy analysis
([27]) https://wikis.ec.europa.eu/display/IMAP/IMAP+Home+page

For each of the practices (or sub-practices) so-called “fiches”, which are documents providing the results of the meta-analysis, are available at three levels:

1. **General fiches** summarise all environmental and climate impacts of a single farming practice. As example, the fiche reporting on the impacts of Nitrogen inhibitors is presented in Annex A4 (Figure A4.1)
2. **Single impact fiches** provide a deeper insight into each individual environmental and climate impact of a farming practice. As example, we have selected a fiche reporting on the specific impact of Nitrogen inhibitors on Nitrogen leaching and run-off” (Figure A4.2 in Annex A4).
3. **Summary fiches** (Figure A4.3 in Annex A4) are individual reports that provide fuller information about the results reported in each reviewed meta-analysis, in particular about the modulation of effects by factors related to soil, climate and management practices. As example, we have selected a Summary fiche about the impact of Nitrogen inhibitors on Nitrogen leaching and run-off, extracted from the meta study Li et al. 2017.

While the general and single impact fiches provide a **qualitative evaluation** of the impact of the farming practices, the summary fiches may include also **quantitative data** extracted from the reviewed meta-analyses.

4.3 Scenario of reduction of atmospheric nitrogen deposition (EMEP model)

Within the JRC C5 unit, the EMEP Air Chemistry Transport Model is used for different projects related to air quality, together with different anthropogenic emission inventories, such as the EDGAR V5 2015 inventory.

For this project we provide nitrogen (N) deposition values over Europe, that serve as input to the GREEN model. The GREEN model needs total nitrogen input from atmospheric deposition (ton N per year). The N deposition varies according to land cover type (land covers available in EMEP are Conif, Crops, Decid, Grid, Seminat, Water). In the past this was not considered in GREEN and an average value was used. Data were downloaded from the EMEP model Norwegian website. The total atmospheric nitrogen deposition values should correspond to the sum of the EMEP output variables:

— Wet Depositions:

— WDEP_OXN ((Oxidised Nitrogen [mg(N)/m2])) +

— WDEP_RDN ((Reduced Nitrogen [mg(N)/m2])) +

— Dry Depositions:

— DDEP_OXN_m2grid (Oxidised Nitrogen [mg(N)/m2]) +

— DDEP_RDN_m2grid (Reduced Nitrogen [mg(N)/m2]).

We performed two simulations for the year 2015:

1. Base Case scenario where the latest state-of-the-art EDGAR anthropogenic emissions were used and
2. a Scenario that includes anthropogenic emission reductions over Europe as described in the Fit For 55 package for the year 2030, also known as EU Reference Scenario (integrated with the National Emission Ceiling Directive reductions, for the pollutants not covered by the Fit For 55; see below for more details).

This EU Reference Scenario (Fit for 55) is one of the European Commission's key analysis tools in the areas of energy, transport and climate action. It allows policy-makers to analyse the long-term economic, energy, climate and transport outlook based on the policy framework in place in late 2020. This scenario can provide policy-makers with a comprehensive analytical basis against which they can assess new policy proposals. National experts from all EU countries contributed to the Reference Scenario 2020 through a consultation process, and stakeholders have also contributed on technology assumptions. More information about EU Reference Scenarios can be found here: https://ec.europa.eu/energy/data-analysis/energy-modelling/eu-reference-scenario-2020.

The EMEP model is run on the JRC in-house JEODPP computing platform.

4.3.1 EDGAR emissions

In this study we used the emissions for aerosol and aerosol precursor gases from the EDGAR version 5.0 inventory (available at https://edgar.jrc.ec.europa.eu/dataset_ap50; Janssens-Maenhout et al., 2019, Oreggioni et al., 2021, Crippa et al., 2020, Oreggioni et al., submitted to Energy Policy (2021). The Emissions Database for Global Atmospheric Research (EDGAR) is a global inventory providing greenhouse gas and air pollutant emissions estimates for all countries over the time period 1970 to the recent present, covering all IPCC reporting categories, with the exception of Land Use, Land Use Change and Forestry (LULUCF). More detailed information about the emission inventory is given in Thunis et al., 2021 and references therein.

4.3.2 Model description and set-up

In this study we use the off-line regional transport chemistry EMEP model version rv_34 (Simpson et al., 2012; https://github.com/metno/emep-ctm), to calculate atmospheric deposition (wet and dry) values of nitrogen (N) over Europe for the year 2015. The domain stretches from -15.05° W to 36.95° E longitude and 30.05° N to 71.45° N latitude with a horizontal resolution of 0.1° x 0.1° and 20 vertical levels, with the first level around 45 m. The EMEP model uses meteorological initial conditions and lateral boundary conditions from the European Centre for Medium Range Weather Forecasting

(ECMWF-IFS) for the meteorological year 2015. The temporal resolution of the meteorological input data is daily, with 3-hour timestep. The meteorological fields for EMEP are retrieved on 0.1° × 0.1° longitude latitude coordinate projection. Vertically, the fields on 60 eta (η) levels from the IFS model are interpolated onto the 20 EMEP sigma (σ) levels. The MARS equilibrium module is used calculate the partitioning between gas and fine-mode aerosol phase in the system of SO4=, HNO3, NO3-, NH3-, NH4+ (Binkowski and Shankar, 1995). More information on the gas and aerosol portioning is given in Simpson et al. (2012), section 7.6. Detailed information on the meteorological driver, land cover, model physics and chemistry are described in Simpson et al. (2012) and in the EMEP Status Report 2017 (https://emep.int/publ/reports/2017/EMEP_Status_Report_1_2017.pdf).

4.3.3 Description scenario REF 2030

The EU Reference Scenario 2020 is the baseline scenario on which specific policy scenarios and variants used to assess options informing the policy initiatives in the European Green Deal package adopted by the European Commission in July 2021 have been developed (source: https://ec.europa.eu/energy/data-analysis/energy-modelling/eu-reference-scenario-2020). The EU Reference Scenario (REF) is available at 2015 and 2030, with values at country level.

In particular, the following emission values are available for:

— Air quality

— PM2.5.

— SO2.

— NOx.

— Non CO2 GHG

— CH4.

— N20.

— Fgases.

Fit For 55 package doesn't provide projections for NH3 and NMVOCs. Therefore, we use 2030 projections from the 'National Emission Ceiling Directive' (NEC) projections (https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32016L2284&from=EN#d1e32-19-1).

The EDGAR emissions are multiplied by a reduction factor, to project the emission changes between 2015 and 2030 based on REF 2030 (PM2.5, SO2 and NOx) and NEC (NH3 and NMVOC) for EU27 (UK+NO+CH are not included). We consider EU27, because the model results will be used to show the expected effects on Nitrogen deposition in the EU from the implementation across the EU27 of the policies in the Fit For 55 package.

In Table 14 the emission reductions for PM2.5, SO2 and NOx are shown, which are obtained from the Fit For 55 package for each single EU country. The ratio indicates how much the emissions in each country is projected to change from the 2015 base, by 2030.

Table 14. Emission totals (2015) and projections (2030) for PM25, SO2 and NOx (in kt) based on Fit For 55 package. Together with the ratio in the emissions between the two years.

		PM25 (kt)			SO2 (kt)			NOx (kt)		
Country		REF 2015	REF 2030	ratio	REF 2015	REF 2030	ratio	REF 2015	REF 2030	ratio
	EU	1361.2	813	0.60	2473.2	1045.1	0.42	7215.9	3283.9	0.46
Austria	AT	17.4	11.9	0.68	13.3	9.2	0.69	160.1	65.3	0.41
Belgium	BE	26.1	15	0.57	39.7	35.1	0.88	201.9	95.3	0.47
Bulgaria	BG	34	21.5	0.63	140.8	56.3	0.40	138.6	80.7	0.58
Cyprus	CY	1.4	0.9	0.64	13.8	2.2	0.16	14.5	7	0.48
Czech Republic	CZ	43.2	19.5	0.45	124.8	38.2	0.31	195.7	104.7	0.54
Germany	DE	95.2	82	0.86	342.3	174.7	0.51	1200	525.7	0.44
Denmark	DK	21.9	13.7	0.63	9.8	10	1.02	117.4	67.3	0.57
Estonia	EE	10.2	2.9	0.28	36.7	7.9	0.22	30.1	15.5	0.51
Greece	EL	30.8	19.2	0.62	86.4	26.4	0.31	249.3	92.5	0.37
Spain	ES	115.3	77.6	0.67	251	87.9	0.35	829.7	337.1	0.41
Finland	FI	21.5	19.1	0.89	41.2	23	0.56	143.9	83.7	0.58
France	FR	179.4	123.4	0.69	158.3	88.5	0.56	1025	418.2	0.41
Croatia	HR	22.2	11.6	0.52	15.1	7	0.46	57.2	27	0.47
Hungary	HU	53.4	34.5	0.65	25.4	8.6	0.34	118	71.6	0.61
Ireland	IE	15	8.4	0.56	15.8	8.5	0.54	105.4	59.4	0.56
Italy	IT	158.9	85.5	0.54	129	70.4	0.55	831.4	342.2	0.41
Lithuania	LT	16.5	7.1	0.43	14.3	10.5	0.73	48	29.7	0.62
Luxembourg	LU	1.7	1	0.59	1.4	1.2	0.86	31.8	7.9	0.25
Latvia	LV	17.2	7.4	0.43	3.7	2.9	0.78	38.6	25.2	0.65
Malta	MT	0.3	0.2	0.67	2.2	0.7	0.32	4.7	2.2	0.47
Netherlands	NL	15.2	13.8	0.91	31.9	16.4	0.51	267.2	123	0.46
Poland	PL	246.9	120.7	0.49	691.1	258.1	0.37	748.1	369.9	0.49
Portugal	PT	48	30.1	0.63	46.2	21.7	0.47	176.9	84.2	0.48
Romania	RO	117.4	47.3	0.40	147.9	48.5	0.33	232.7	130.9	0.56
Sweden	SE	22.2	17.7	0.80	17.8	14.6	0.82	139.1	59.2	0.43
Slovakia	SK	18.6	11.9	0.64	67.6	13.1	0.19	74.8	40.7	0.54
Slovenia	SI	11.4	9.2	0.81	5.8	3.5	0.60	35.7	17.8	0.50

Emission reduction commitments for NMVOCs and NH3 are as indicated in the NEC Directives. The reduction commitments have the year 2005 as base year. We have recalculated the reductions relative to the year 2015 as shown in Table 15, by considering a linear implementation of the NECD over the entire period 2005-2030.

Table 15. Emission ratios between the base year 2005 and projected year 2030 for NMVOC and NH3 based on NEC Directive. Together with the ratios between 2015 and 2030 for NMVOC and NH3.

		NEC directive Reduction compared with 2005		**Reduction compared with 2015**	
Country		2005 to 2030		2015 to 2030	
		NMVOC	**NH3**	**NMVOC**	**NH3**
Austria	AT	0.36	0.12	0.216	0.072
Belgium	BE	0.35	0.13	0.210	0.078
Bulgaria	BG	0.42	0.12	0.252	0.072
Cyprus	CY	0.48	0.25	0.288	0.150
Czech Republic	CZ	0.5	0.2	0.300	0.120
Germany	DE	0.5	0.22	0.300	0.132
Denmark	DK	0.37	0.24	0.222	0.144
Estonia	EE	0.28	0.01	0.168	0.006
Greece	EL	0.48	0.2	0.288	0.120
Spain	ES	0.52	0.13	0.312	0.078
Finland	FI	0.28	0.29	0.168	0.174
France	FR	0.62	0.1	0.372	0.060
Croatia	HR	0.58	0.32	0.348	0.192
Hungary	HU	0.32	0.05	0.192	0.030
Ireland	IE	0.46	0.16	0.276	0.096
Italy	IT	0.38	0.01	0.228	0.006
Lithuania	LT	0.47	0.1	0.282	0.060
Luxembourg	LU	0.42	0.22	0.252	0.132
Latvia	LV	0.27	0.24	0.162	0.144
Malta	MT	0.15	0.21	0.090	0.126
Netherlands	NL	0.26	0.17	0.156	0.102
Poland	PL	0.38	0.15	0.228	0.090
Portugal	PT	0.45	0.25	0.270	0.150
Romania	RO	0.32	0.3	0.192	0.180
Sweden	SE	0.53	0.15	0.318	0.090
Slovakia	SK	0.39	0.16	0.234	0.096
Slovenia	SI	0.36	0.17	0.216	0.102

4.3.4 Results

The results of the EMEP model are presented in Figure 29 and 30. Figure 29 shows the dry deposition quantities of Oxidised and Reduced Nitrogen, and the wet deposition values of Oxidised and Reduced Nitrogen for the Base Case 2015 simulation for EU27 countries. Figure 30 shows the difference between the Base Case (2015) and EU Reference Scenario (2030) in dry deposition quantities of Oxidised and Reduced Nitrogen, and in the wet deposition values of Oxidised and Reduced Nitrogen.

Figure 29. Dry deposition quantities for (a) Oxidised Nitrogen and (b) Reduced Nitrogen, together with the wet deposition values of Oxidised Nitrogen (c) and Reduced Nitrogen (d) for the Base Case 2015 simulation for EU27 countries (in mgN/m^2).

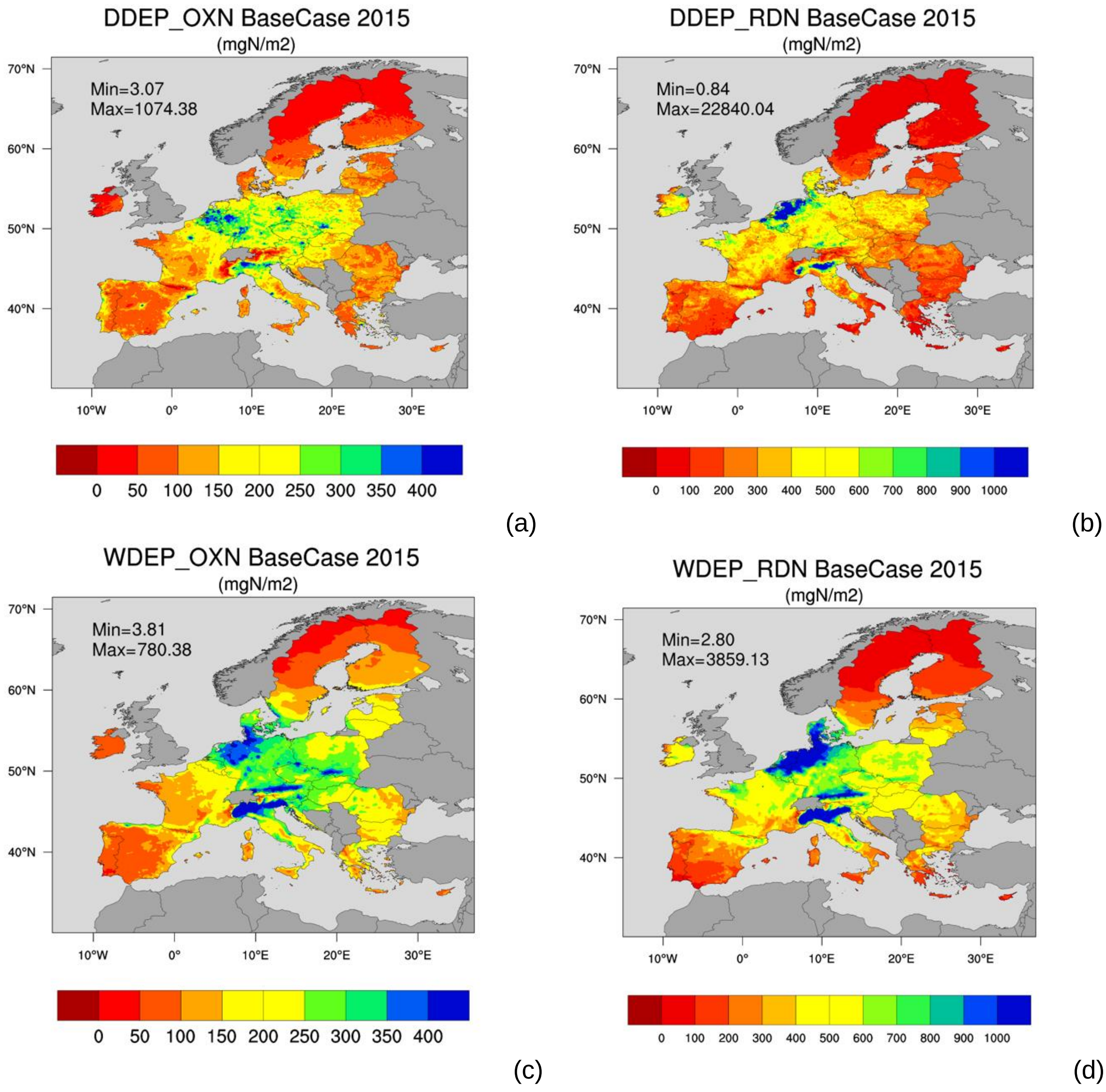
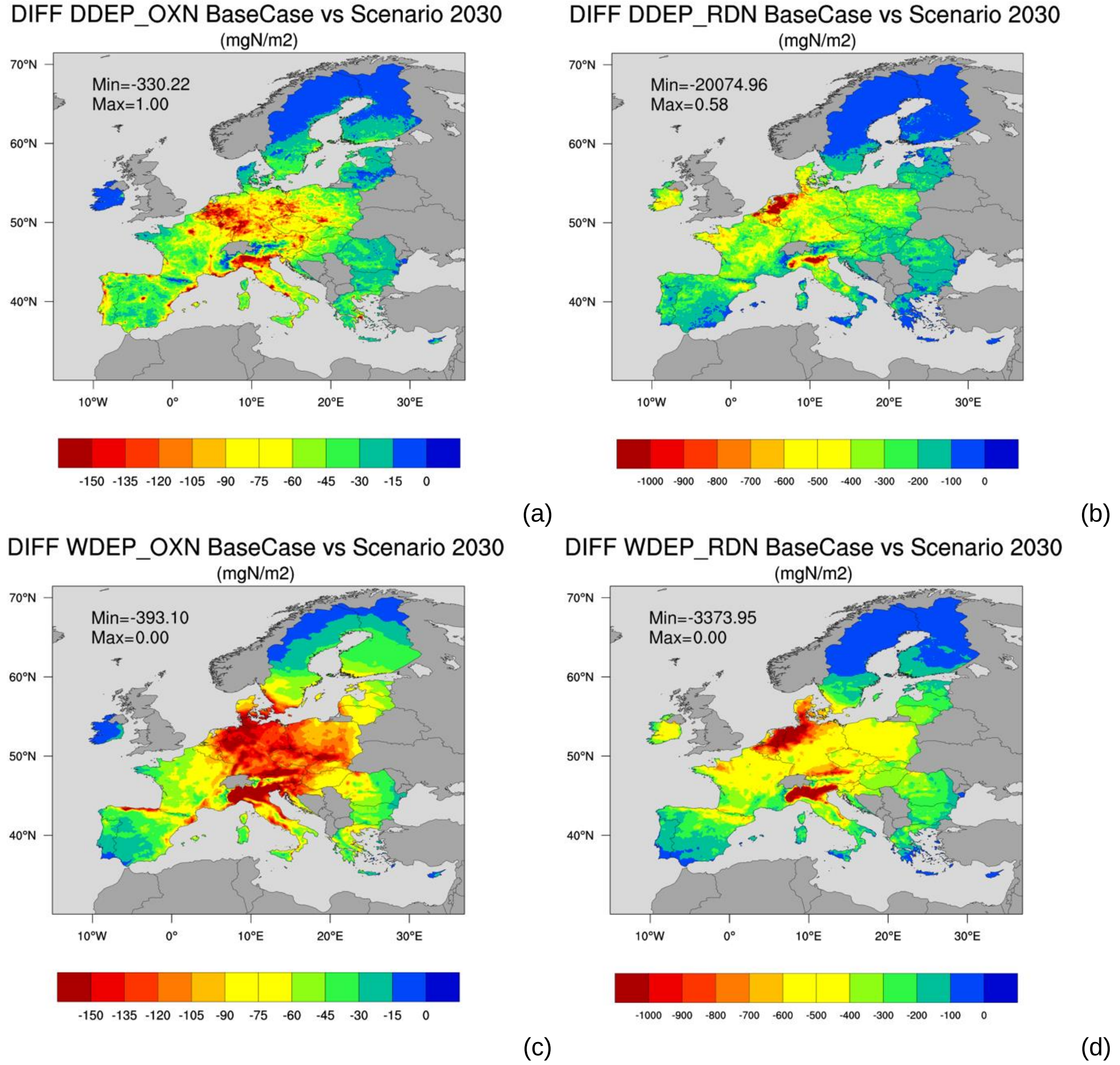


Figure 30. Difference in dry deposition quantities between the Base Case (2015) and EU Reference Scenario (2030) for (a) Oxidised Nitrogen and (b) Reduced Nitrogen, together with the differences in wet deposition values of Oxidised Nitrogen (c) and Reduced Nitrogen (d) between the Base Case and EU Reference 2030 simulation for EU27 countries (in mgN/m^2).



4.4 Scenarios of reduction of nitrogen fertilisers in agriculture (DayCent model)

4.4.1 Modelling framework: biogeochemical soil model

The JRC has developed a state-of-the-art process-based European biogeochemical modelling platform that **simulates carbon (C) and nitrogen (N)** flows within soil and between soil, the atmosphere and vegetation.

Key sub-models include decomposition of organic input and soil organic matter, mineralisation of nutrients, N gas emissions from nitrification and denitrification, soil water content and temperature by layer, plant production and allocation of net primary production (NPP) and CH_4 oxidation in non-saturated soils and CH_4 production in flooded soils. Flows of C and N between the different soil organic matter pools are controlled by the size of the pools, C/N ratio and lignin content of material, and abiotic water/temperature factors. Plant production is a function of genetic potential, phenology, nutrient availability, water/temperature stress and solar radiation. NPP is allocated to plant components (e.g., roots vs. shoots) based on vegetation type, phenology, and water/nutrient stress. Nutrient concentrations of plant components vary within specified limits, depending on vegetation type, and nutrient availability relative to plant demand. Decomposition of litter and soil organic matter and nutrient mineralization are functions of substrate availability, substrate quality (lignin %, C/N ratio), and water/temperature stress. N gas fluxes from nitrification and denitrification are driven by soil NH_4 and NO_3 concentrations, water content, temperature, texture, and labile C availability (Parton et al., 2001).

In this project, DayCent is run on a 1 km^2 grid using the following data (Figure 31):

— soil properties available for ESDAC and derived from spatial interpolation of LUCAS soils (https://esdac.jrc.ec.europa.eu/);

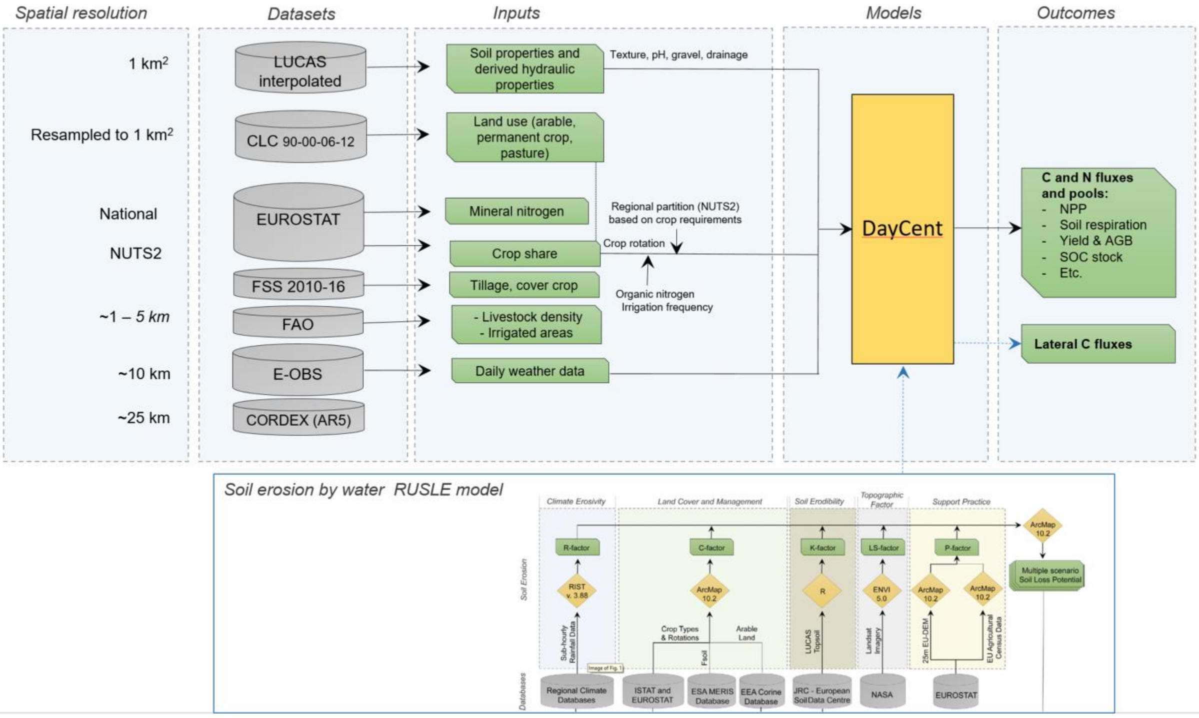
— land cover from the CORINE LAND COVER 1990, 2000, 2006, 2012;

— official statistics (EUROSTAT, FAO, Farm Structure Survey) and spatial datasets, which were used to describe the current management (i.e. crop rotation, mineral and organic N fertilization, tillage, irrigation, cover crop, etc.);

— meteorological data from the E-OBS gridded dataset (http://www.ecad.eu). The dataset provided daily data of maximum and minimum temperature and precipitation on a grid of 0.1° resolution (v22). For the climatic projection, we used the general circulation model CNRM-CM541 run with a RCP4.5 (Thomson et al. 2011) and downscaled with the RCM CCLM4-8-17, available at the WCR-CORDEX portal (WCR-CORDEX, 2019);

— average wet and dry deposition N depositions from the EMEP model (rv 4.5)

Figure 31. Flow chart showing the datasets utilized and their spatial resolution, the inputs derived and the model integration.



As inputs, the amount and timing of nutrient amendments is required. The current (baseline) N fertilization was characterised as follow:

— Mineral N fertiliser: it was partitioned in two applications at planting (30%) and standing crops (70%). In each fertilization the proportion of NH_4 and NO_3 was assumed to be equal to 75 and 25%, respectively;

— Organic: applied generally after harvest or during standing crop in highly demanding crops such as maize. The territorial rates calculated was limited to the maximum rate of 170 kg/ha of N per year.

Model **outputs** include: daily N fluxes (N_2O, NOx, N_2, NO_3- leaching), CO_2 flux from heterotrophic soil respiration, soil organic C, NPP (partitioned into residues, grains and harvested root crops). The model takes into account land management and cropping practices. Since it is driven by a range of climate scenarios, as simulated by Global Climate Models, the model can provide long-term policy perspectives.

The **ability** of DAYCENT to simulate NPP, soil organic carbon, N_2O emissions, and NO_3- leaching has been tested with data from various native and managed systems (e.g. Del Grosso et al., 2001, 2006). The DAYCENT model is currently being used by the United States Environmental Protection Agency, United States Department of Agriculture and Colorado State University to develop a national inventory of N_2O emissions from U.S. agricultural soils. This inventory is compared and contrasted with the existing Intergovernmental Panel on Climate Change (IPCC) agricultural N_2O emissions inventory for the USA.

The **JRC Units D3 and D5** has developed and continuously improved the modelling framework in the EU in the last decade, initially using CENTURY (the monthly version model) and then DayCent (the daily version model), running it both at LUCAS point and gridded 1 km level (Lugato et al., 2014a,b, 2015, 2016, 2017, 2018a,b, 2020; Monforti et al., 2015; Scarlat et al., 2019; Quemada et al., 2020a). This framework was used for many scientific studies and policy scenarios, receiving a scientific recognition. For more information on the general architecture, model performances and different scenarios and agricultural management simulated, we refer to the reference section.

4.4.2 Modelling scenarios

One of aim of the Farm to Fork (F2F) and Biodiversity strategies is acting to reduce nutrient losses by at least 50%, while ensuring that there is no deterioration in soil fertility. This will reduce the use of fertilisers by at least 20% by 2030

However, in the agricultural land, complex interactions between biogeochemical cycles, environmental and management conditions make very difficult to estimate this cause-effect relationship in a quantitative way. Nitrogen losses originate and flow from different environmental compartments (air, water, soil) and are interlinked, therefore, is not possible to constrain "a priory" complex models to get targeted losses reductions. Conversely, we can use the DayCent modelling framework to run a scenario involving a reduction of N fertilization in arable and grassland soils of the EU (+UK), in order to assess the overall effect in term of losses and cause-effect relationships.

The first step involved the identification of the N operative space (Figure 32) as illustrated by Quemada et al., 2020b. Plotting total N input vs N outputs (from crop harvest including grassland), we can define a spatially explicit operative space at the EU level. The concept relies on the fact that some environmental impacts may arise from both:

— Excessive surplus on N, leading to detrimental losses to air and water,

— Very high N use efficiency (NUE) that can mine soil fertility and reduce the productivity.

According to Quemada et al., 2020b and expert opinions, we define an operative N safe space having NUE<0.9 or surplus <50 kg/ha/y. The areas (pixels) under those conditions are identified using N flows from the DayCent modelling framework (Figure 33) and reported in Figures 34 and 35. We did not consider a minimum N removal level.

The second step consists in simulating a change of N mineral fertilization in agricultural soils as following:

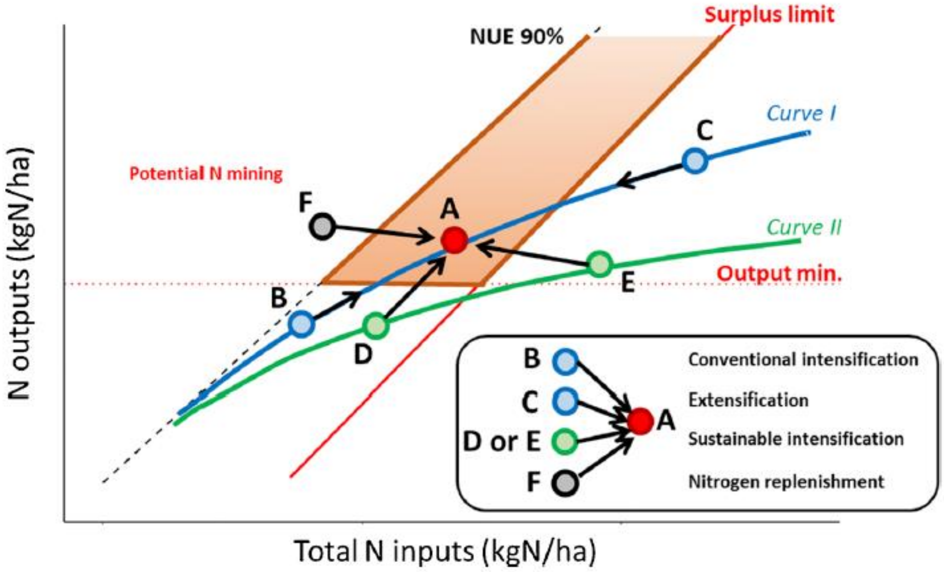
— Decrease of 20% of mineral N in surplus areas,

— Increase of 20% of mineral N in potential mining areas.

The objective of the BDS and F2F is the reduction of nutrient losses, which goes with increased nutrient efficiency that can be achieved by different strategies, including a reduction of fertilisers in excess. The scenario cannot test "a priori" the 50% reduction of losses, which is a result of many interacting ecosystem components, but the effect of possible management like the reduction of mineral fertilisers. The 20% change is selected according to one of the foreseen effect of the F2F strategy of "reducing fertilizer use at least 20% by 2030". This represents also a cost-effective scenario, which has likely a prompt effect on N losses reduction.

Organic fertilization was not included in any scenario since it requires more deep analysis on potential leakage, indirect land use change and/or dietary habits.

Figure 32. Nitrogen operative space. Within the safe operative space the agroecosystems guarantee food security minimizing environmental impacts and reduced soil fertility.



Source: Quemada at al. (2020b).

Figure 33. Nitrogen surplus and NUE at 1km² spatial resolution from DayCent modelling framework. The surplus is defined as total N input – N exported by crop harvest. The NUE is the ratio between the latter and former flows.

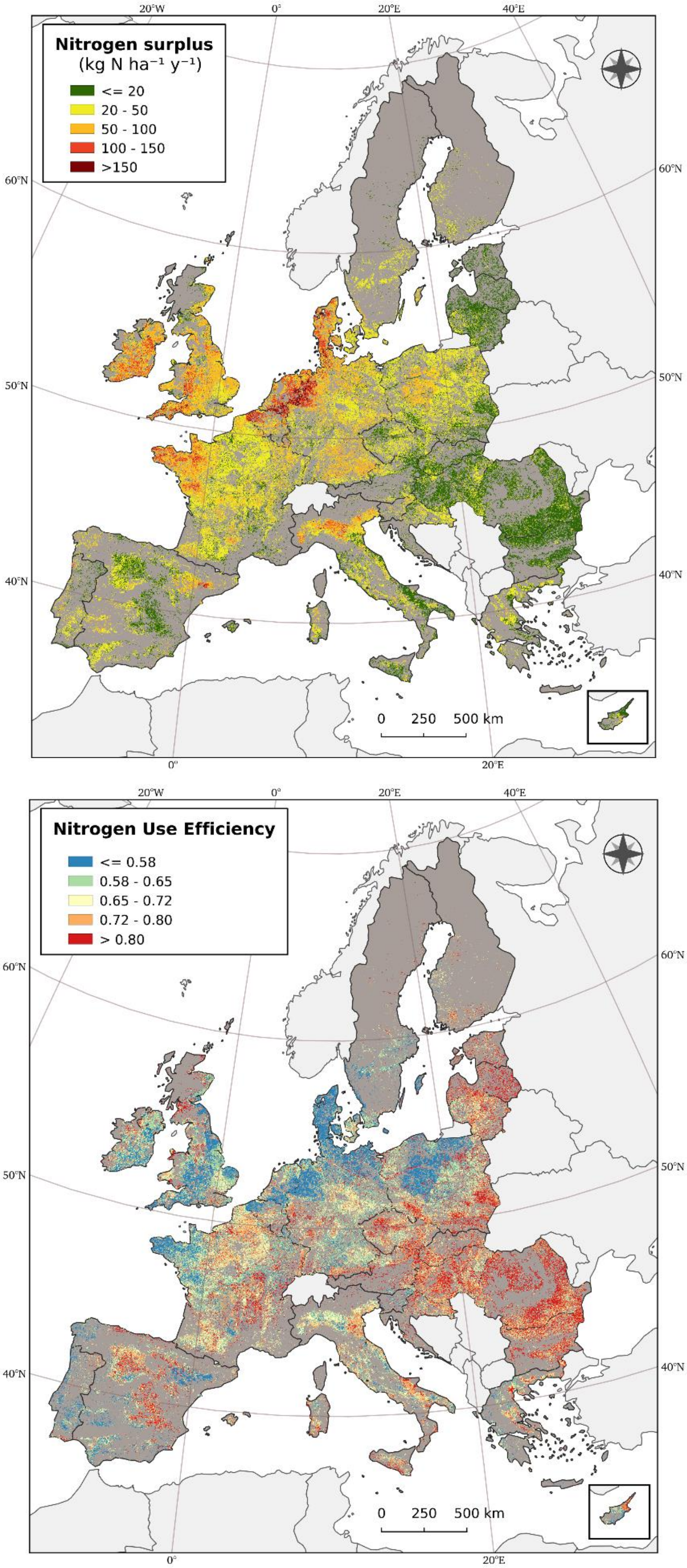


Figure 34. Spatial distribution of N safe operative space, potential N mining and N surplus. Bars indicate the relative proportion of areas under different condition at Country level.

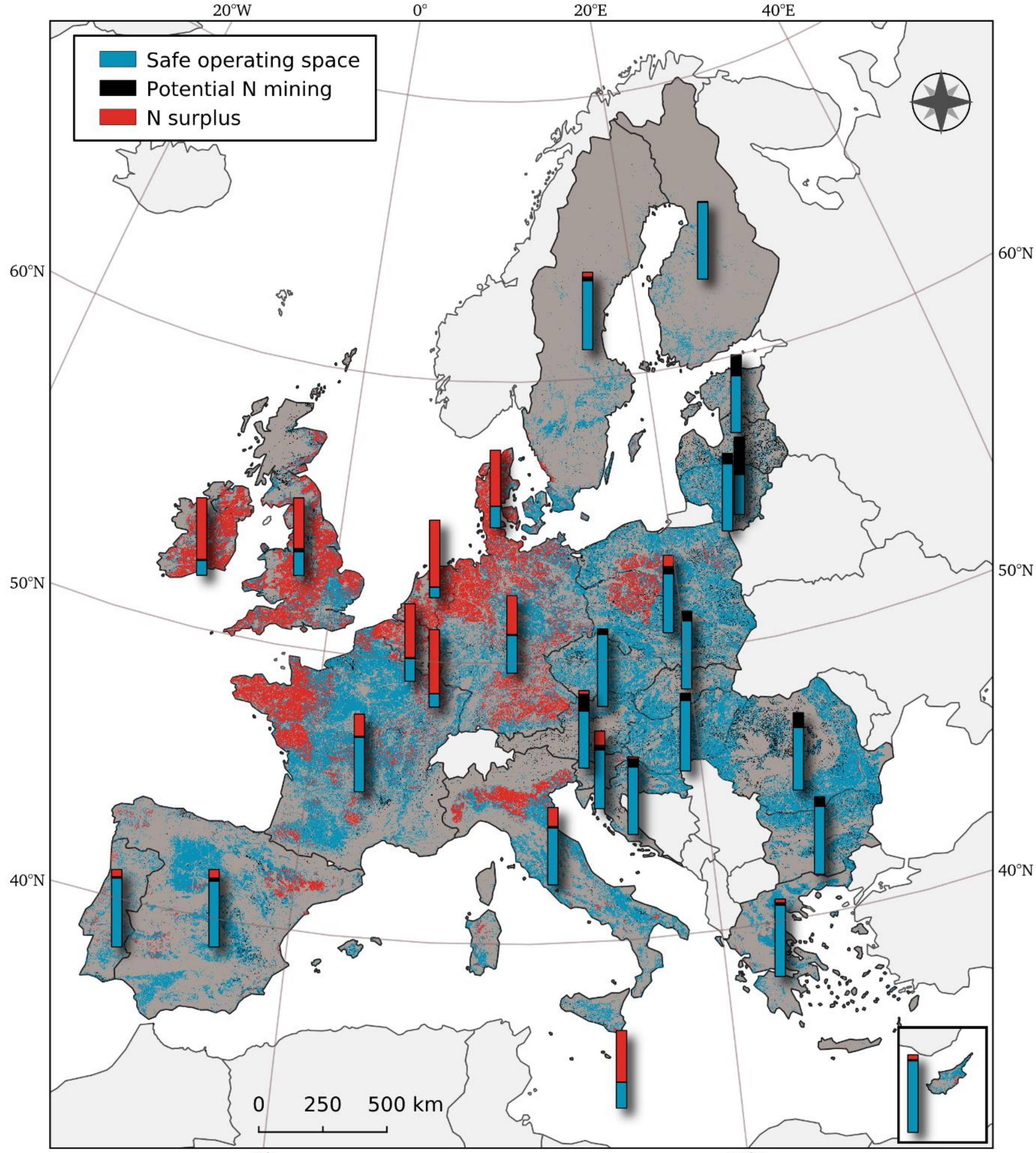
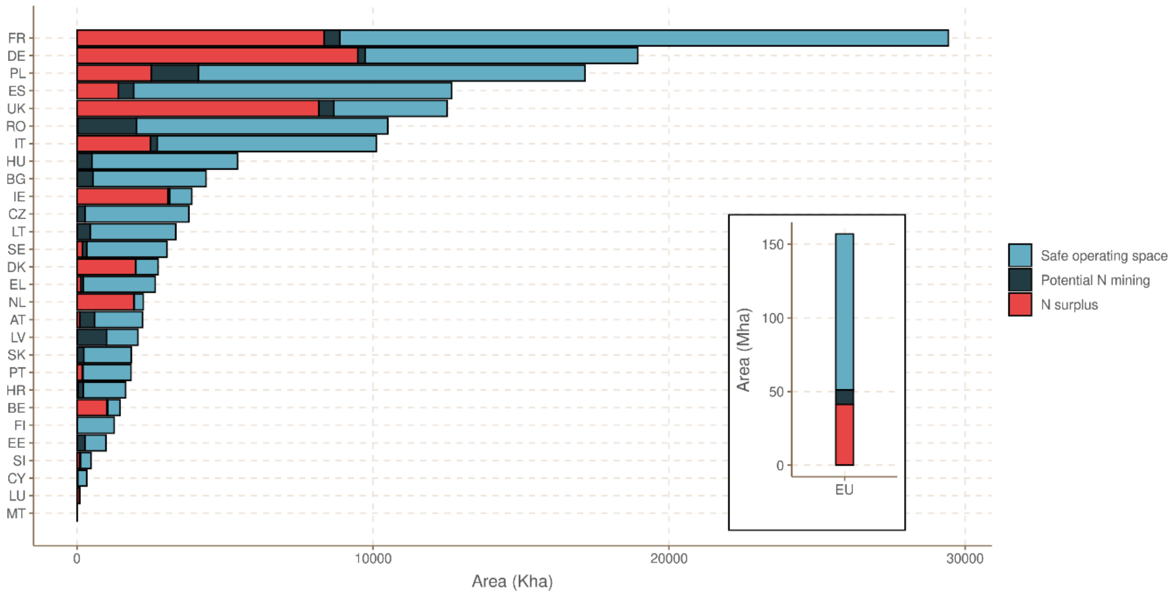


Figure 35. Cumulative areas of N safe operative space, potential N mining and N surplus. The inlet is the cumulative values at EU+UK.



4.4.3 Results

The results of the scenario analysis showed that:

- a more balanced mineral N fertilization may allow a consistent saving of mineral N, especially in some countries (DE, FR and UK) (Figure 36). Cumulative, the N fertilizer redistribution could reach up to 7% saving of current total application (Figure 40 and Table 16).
- nitrogen leaching decreased to a lower extent (compared to change in fertilization) to about 6% of the current NO3 losses (Figure 37 and Table 16). Central Europe plus IE and UK showed higher benefit, while very marginal effects were predicted in Eastern countries.
- The N2O emissions patterns followed those of change in mineral N fertilization (Figure 38). The cumulative change was about -4% of the current direct soil emissions (Table 16).
- SOC slightly increased in areas current affected by N mining, and the opposite in surplus area due to feedbacks in primary productivity (Figure 39). However, the total SOC change (-14 Mt C) represents only 0.1% of the current topsoil SOC stock (Table 16), which can be easily recovered by appropriate best practices (cover crop, residue management, agroforestry etc.).

Further analyses should be worth in the future to assess more targeted reduction-increase fertilization levels at regional scale and run additional scenarios.

Figure 36. Spatial explicit change in N mineral fertilization (kg ha^{-1} yr^{-1}) and cumulative values at National level. The scenario involves the decrease and increase of 20% of mineral N in surplus and in potential mining areas, respectively, as calculated in Figure 34.

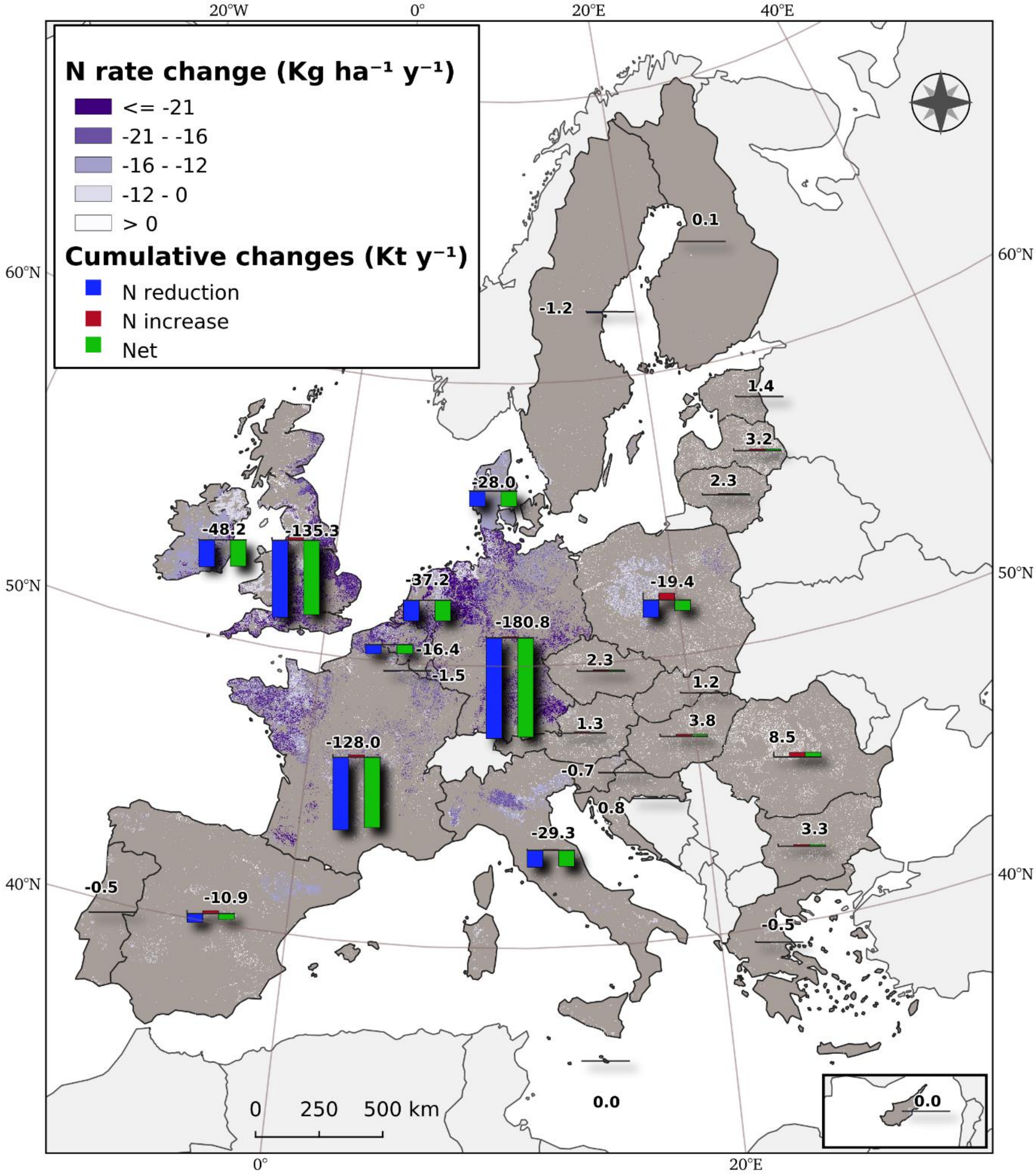


Figure 37. Spatial explicit change in NO_3 leaching (kg NO_3-N ha^{-1} yr^{-1}) and cumulative values at National level. The scenario involves the decrease and increase of 20% of mineral N in surplus and in potential mining areas, respectively, as calculated in Figure 34.

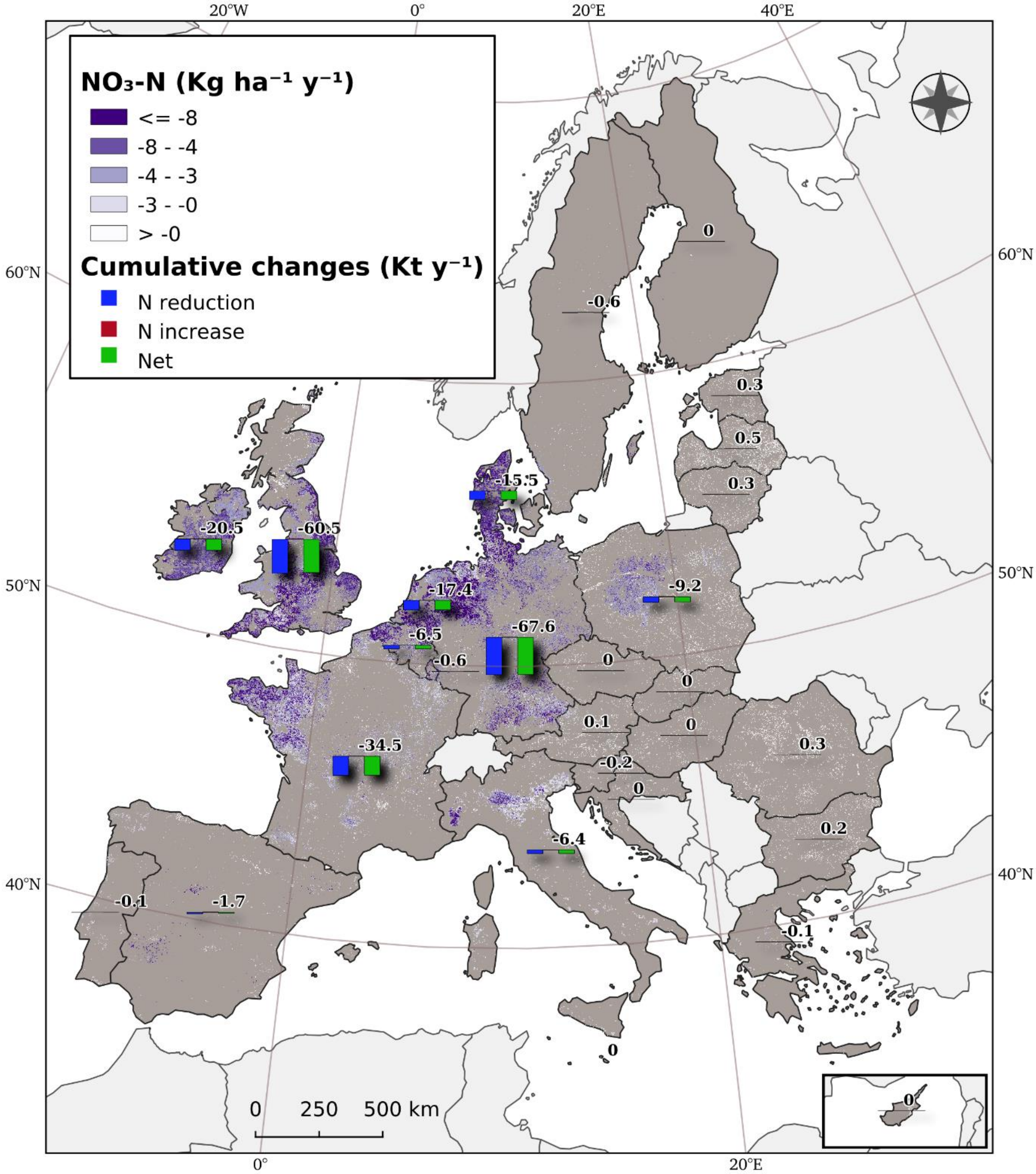


Figure 38. Spatial explicit change in N_2O emissions (kg N_2O-N ha^{-1} yr^{-1}) and cumulative values at National level. The scenario involves the decrease and increase of 20% of mineral N in surplus and in potential mining areas, respectively, as calculated in Figure 34.

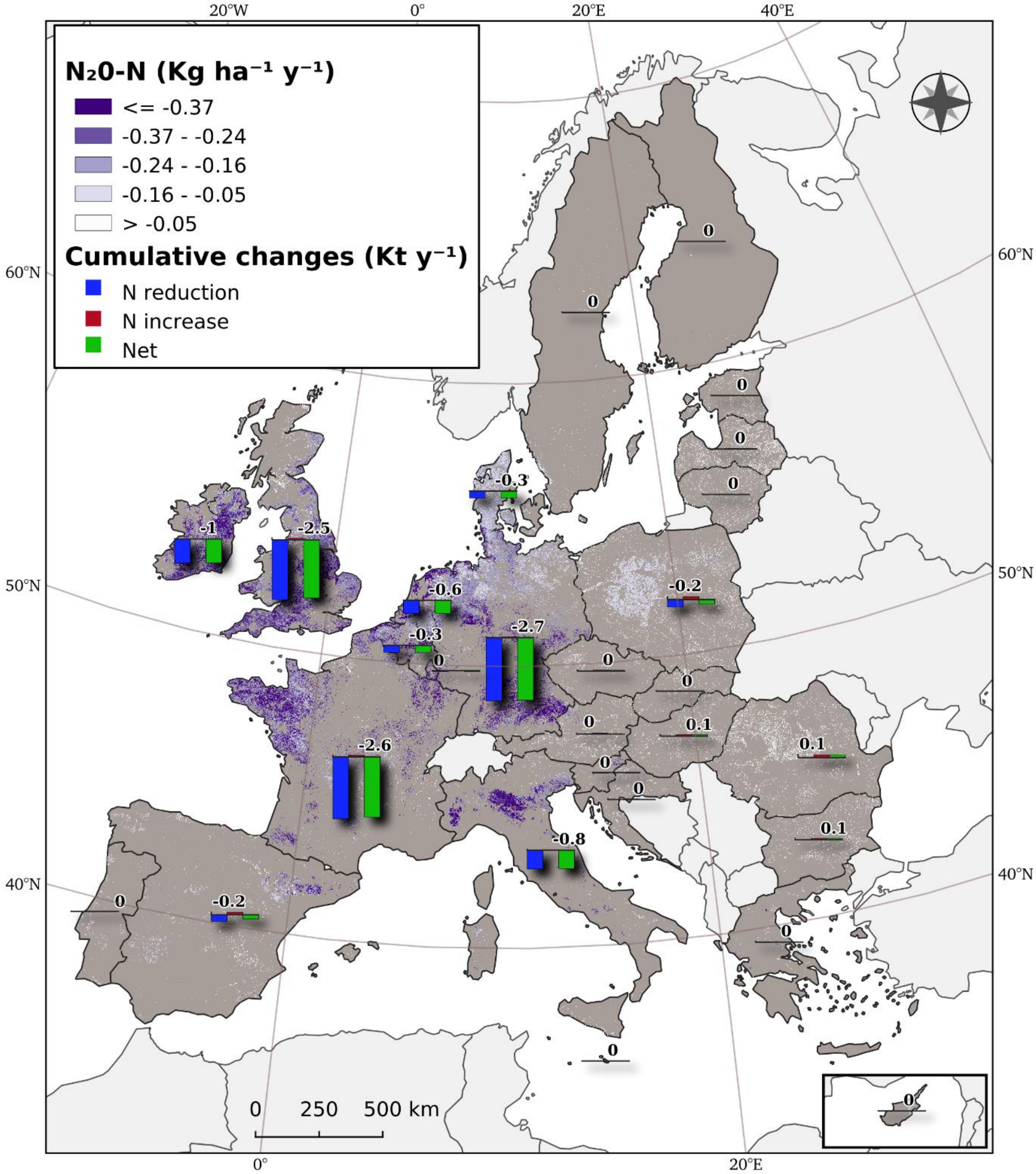


Figure 39. Spatial explicit change in SOC (t ha^{-1} in the topsoil 0-30 cm layer) at equilibrium and cumulative values at National level. The scenario involves the decrease and increase of 20% of mineral N in surplus and in potential mining areas, respectively, as calculated in Figure 34.

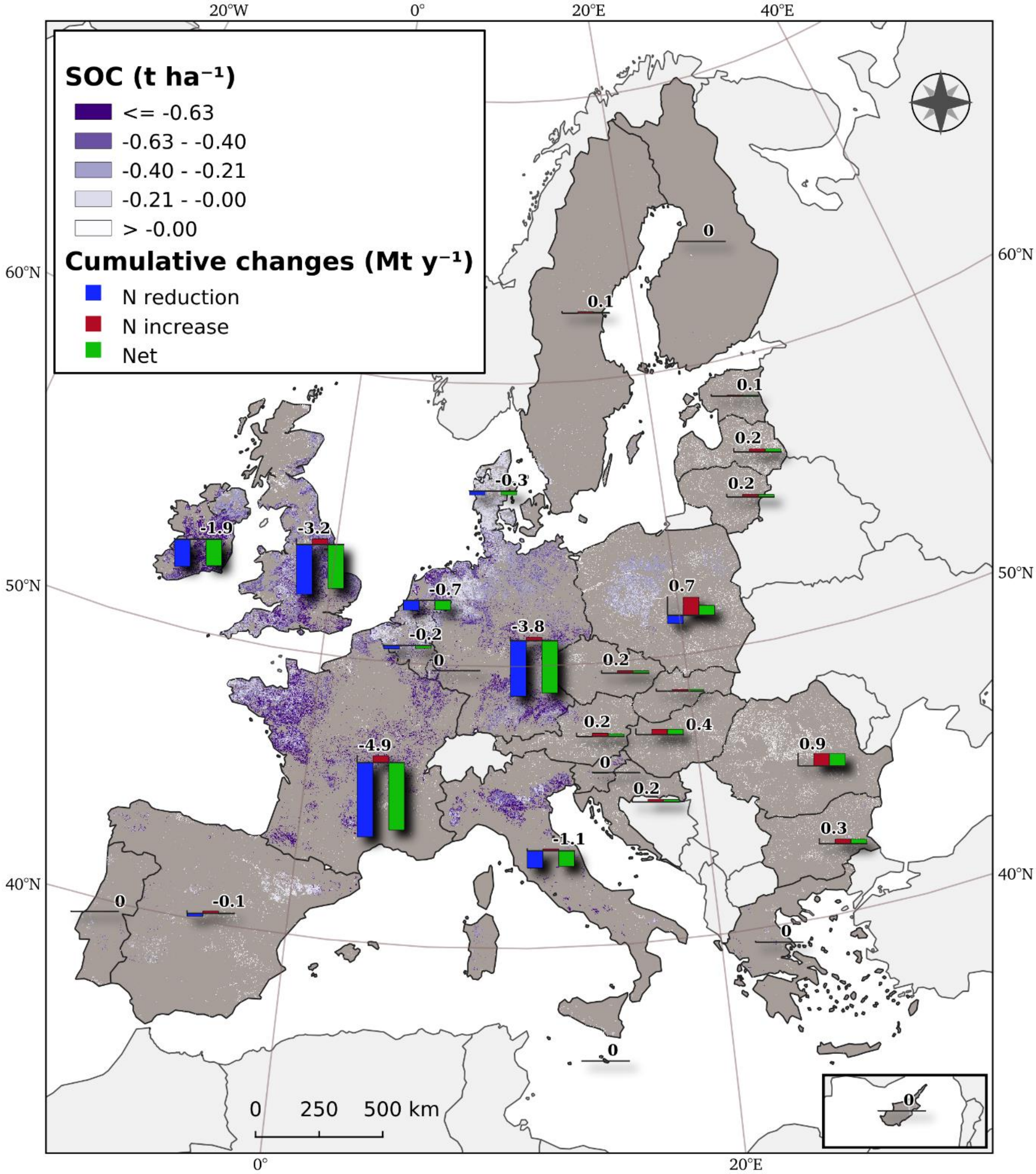


Figure 40. Cumulative changes in N flows and SOC at Member State and EU+UK level.

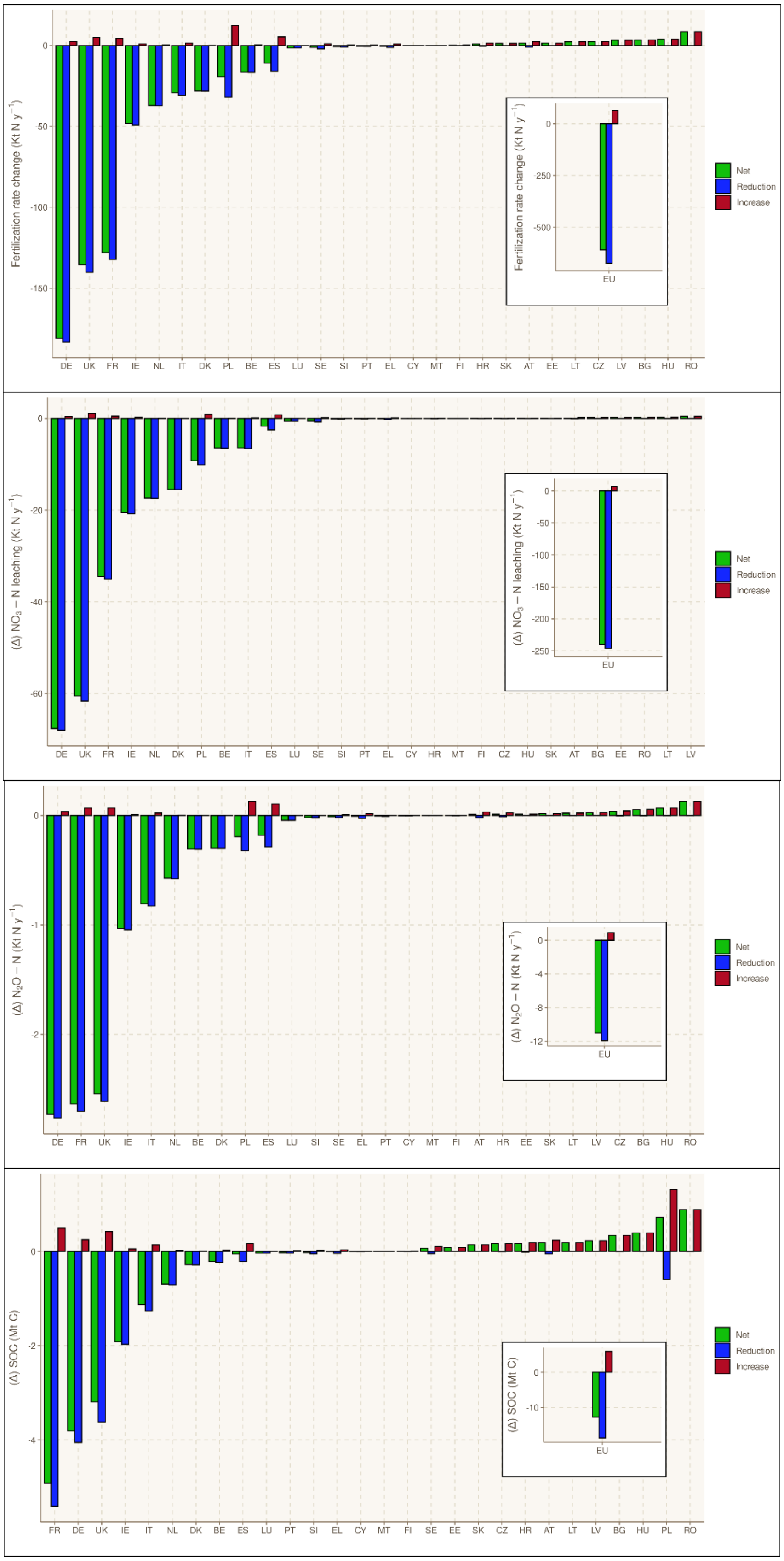


Table 16. Cumulative change in N and C flow at EU+UK level and their relative variation with current estimates.

Flow-stock	**current**	**change**	**%**
Mineral N fertilization (kt)	9075	-612	-6.7
NO_3-N leaching (kt)	4179	-241	-5.8
N_2O-N emissions (kt)	283	-11	-3.9
SOC (Mt)	12300	-14	-0.1

4.5 Phosphorus budget in European agricultural soils with an empirical model

JRC D3 and D5 Units have developed a research study to a) investigate the main inputs and outputs of phosphorus in European agricultural topsoils; b) assess the phosphorus budget and c) develop an empirical model which can simulate the phosphorus budget at regional level. Finally, we make some considerations on limiting excess of nutrients relevant to recent policy developments in the EU legislation such as the Farm to Fork strategy (F2F).

4.5.1 Methods and data inputs

4.5.1.1 Study area

The study area includes all agricultural lands of the European Union (EU) plus the United Kingdom (UK). These cover about 41.5 % of the total land area in EU and UK and we will focus on about 180 million ha. The fraction of the total area occupied by agricultural land is not equally distributed amongst European countries. For example, this value exceeds more than ¾ in Ireland, the Netherlands, and Denmark while this share is less than ¼ in Malta, Finland and Sweden, showing substantial variations between countries (Panagos et al., 2021).

4.5.1.2 Empirical model for phosphorus balance

We develop an improved Empirical Model Phosphorus Balance (EMPBa) framework, which is based on the equation (1):

Phosphorus budget = P_Fert + P_Man + P_Atm + P_Che – P_Grain – P_Res – P_Eros (eq. 1)

With four phosphorus (P) inputs to topsoils:

- P_Fert is the P fertilizer input;
- P_Man the P manure application input;
- P_Atm the P atmospheric deposition;
- P_Che the P deposition due chemical weathering.

The three outputs of the phosphorus balance are:

- P_Grain the output from crop harvesting;
- P_Res the output with crop residues removal;
- P_Eros the phosphorus losses with water erosion;

The units applied in equation (1) are tonnes. The EMPBa is applied at different scales starting from a regional to country level and finally at continental (European).

4.5.1.3 Phosphorus inputs

The fertilizers inputs are based on European Union (EU) agri-environmental indicator mineral fertiliser consumption (EUROSTAT, 2020). According to this indicator, Eurostat publishes the data on Nitrogen and Phosphorus consumption per Member State (MS) for the period 2011-2019. We used the mean value of this indicator per MS. The agro-environmental indicators, such as the ones provided by Eurostat, are widely used for policy analysis, communication to farmers, research purposes and monitoring/evaluating progress (Langeveld et al., 2007).

Over the past few decades, manure application in EU agricultural soils replaces the inorganic fertilizer inputs (Pagliari and Laboski, 2012). In European Union and UK, livestock production generates about 1400 million tons of manure annually (Köninger et al., 2021). We took into account the livestock distribution per region and then we apply Excretion coefficient rates per animal type. Six large countries (DE, ES, FR, IT, PL, UK) produce ca. 68% of the total manure while than 75% of the produced manure derives from cattle.

Anthropogenic activities have changed the global atmospheric chemistry and this has also an impact in global mobility of nutrients (Brahney et al., 2015). Therefore, we have noticed developments in compiling global datasets of atmospheric phosphorus deposition (Mahowald et al., 2008).

4.5.1.4 Phosphorus outputs

Removal of phosphorus with harvesting of crops is the major output of P from soils. To estimate this removal we developed a module which takes into account:

- Crop type
- Crop production (t ha-1 as fresh matter) (Cpr)
- Agricultural utilized area (ha) (AUA)
- Humidity rate (%) (Hum)
- P concentration (%) in plant tissue as dry production (Pc)

$$P\_CropUptake\ (tonnes) = Cpr \times AUA \times (1-Hum) \times Pc \quad \text{(eq. 2)}$$

$$P\_Grain = \sum_{n=1}^{37} (P\_CropUptake) \quad \text{(eq. 3)}$$

The mean crop production rates (Cpr) originate from the Common Agricultural Policy Regional Impact Analysis (CAPRI) model with the reference year 2016 (Himics, 2018; Panagos et al., 2021). However, the crop production rates were also compared with those from Eurostat and data from the Crop Growth Monitoring System (CGMS) used in Monitoring Agricultural Resources (MARS) (Biavetti et al., 2014). In addition, we used the Agricultural utilized area (AUA) per crop and region from CAPRI.

The humidity rates are used to extract the dry production and the data source were Eurostat statistics. The phosphorus uptake rates by different crops (P_Uptake) are existing coefficients in the literature. They vary from high ones as oilseeds (c.a 0.58-0.70%) to lower ones as in fruits (0.07%) or fodder crops (0.21%).

Crop residues include straw, head leaves and stems and other crop residuals which are removed from the field. The removal of plant residues contributes to phosphorus uptake from soils (Erinle et al., 2018). The crop residues removal takes into account:

- Crop type
- Crop production (t ha-1) (CropProd)
- Agricultural utilized area (ha) (AUA)
- Humidity rate (%) (Hum)
- Ratio of residue production per tonne of crop production (RatioResidue)
- Ratio of residues removal from the field (ResidueRemoval)
- P uptake from removed dry residues (P_ResiduesDry)

$$P\_Residues\ (Tonnes) = RatioResidue * (CropProd * AUA) * (1- Hum) * ResidueRemoval * P\_ResiduesDry$$

The ratio of residue production per tonne of crop production is used from experimental sites and works (García-Condado et al., 2019; Scarlat et al., 2010). Ratio of residues removal is stable at 50% while the P uptake show variation between crops. The mean phosphorus uptake from plant residues is around 0.10% on the dry residues content.

4.5.1.5 Phosphorus losses due to soil erosion

The Phosphorus content in topsoils was recently mapped at high resolution (Ballabio et al., 2019) highlighting the main drivers of phosphorus distribution as well as the influence of fertilization in agricultural areas. This is the latest state of the art in Phosphorus concentration in soils at European scale and has advanced both in the number of input samples compared to past assessments and in the machine learning techniques.

We estimated the loss in soil stock of the upper 20cm by using the Bulk Density (BD) of the LUCAS physical properties (Ballabio et al., 2016). The bulk density (range: 1–1.4 Mg m-3; mean: 1.22 Mg m-3), the total weight of 1 ha topsoil has a range of 2000–2800 Mg. Therefore, the P stock depends on both the P content and the bulk density of the topsoils. The soil loss stock (%) is based on the erosion (rill and interill erosion) divided by the total Soil stock of the topsoils.

The P stock is calculated based on P concentration of LUCAS (Ballabio et al., 2019) and the Bulk Density. However, the P concentration in LUCAS is the Olsen (labile fraction). In order to estimate the total phosphorus, we used the ration of P Total / P Labile found in Ringeval et al., (2017). This ratio is between 16 for arable crops and 32 for pastures.

In the next step, we combined spatially explicit estimates of hillslope riverine system sediment fluxes (Borrelli et al., 2018) with the P stocks to compute the amount of P potentially displaced together with soil particles; therefore drained into the nearest river.

4.5.2 Results

4.5.2.1 Phosphorus budget

At global scale, the P input from inorganic fertilizers are estimated to about 14.2 million tonnes per year, the P input from manure is about 9.6 million tonnes while the P removal from harvested crops is about 12.3 million tonnes per year (MacDonald et al., 2011).

The annual consumption of phosphorus with inorganic fertilization inputs are on average c.a 1,310,000 tonnes for EU-27 and UK for the period 2011-2019. This is about 7.2 kg of P ha-1 yr-1 with Netherland, Estonia and Sweden showing lower than 4 kg of P ha-1 yr-1 and Slovenia, Ireland and Poland more than 8 kg of P ha-1 yr-1.

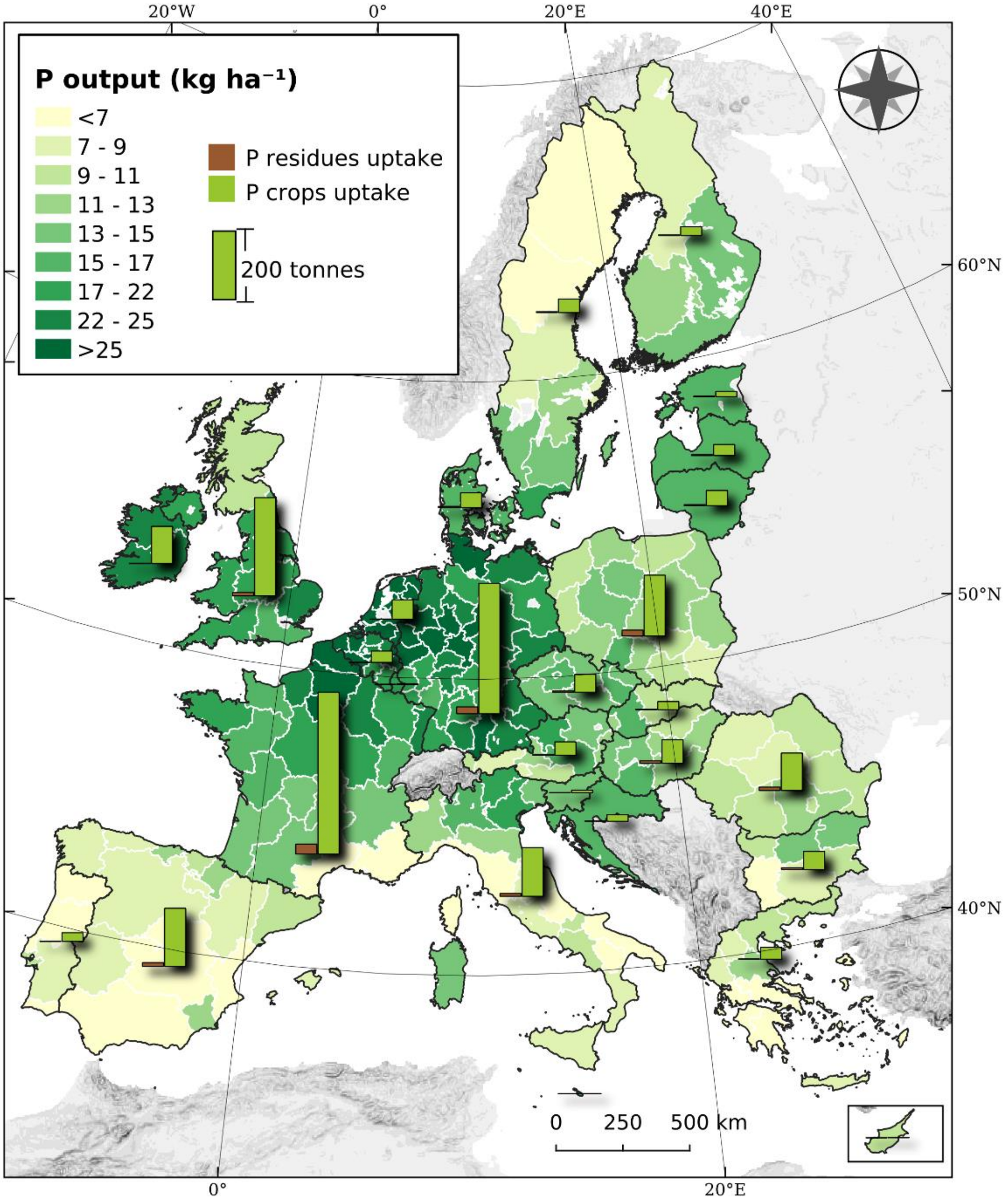
The annual organic P inputs with manure are estimated to about 1,300,000 tonnes in EU and UK. The mean P organic input in EU and UK is about 6.7 kg ha-1 yr-1. Netherlands, Belgium, Denmark and Ireland have the highest P inputs (> 10 P Kg ha-1 yr-1). The lowest P organic application rates are noticed in Baltic States, Scandinavia and Bulgaria (< 2.5 P Kg ha-1 yr-1).

Summing up the inputs from inorganic fertilizers, the manure input and atmospheric deposition, we estimate the annual P input to agricultural soils in EU and UK around 2,720,000 tonnes with an uncertainty of ± 9%. This is about a mean input of 15 kg P ha-1 yr-1. In most of the North-western European regions, the rates of P removal are higher than 20 kg ha−1 year−1, while rates are lower than 10 kg ha−1 in Mediterranean regions and South-East EU countries (Panagos et al., 2022; Figure .41).

The main output in the plant uptake with almost 2.4 million P tonnes removal from soils (Panagos et al., 2022). Also, a small portion of P output are the plant residues which can reach almost 150,000 P tonnes. The P lost to river-basins from agricultural lands due to soil erosion is about 60,000 tonnes.

Therefore, we conclude that the P budget is still a surplus for the whole EU and UK area with about 0.7 kg ha-1 yr-1. Further analysis is available in Panagos et al. (2022b).

Figure 41. Total phosphorus removal per country and region. Green bars aggregate P crop removal per country and brown ones are the aggregated P removal with residues



4.5.2.2 Phosphorus losses due to soil erosion

Coupling soil erosion with P stock allows to have the total P displacement due to water erosion in EU Agricultural lands (Figure 42). However, just a small part of this displaced phosphorus ends to the river basins and furthermore to the sea outlets. The phosphorus which ends to the sea outlets is estimated if we use the deposition/displacement WATEM/SeDEM model. The total P losses to river basins and sea outlets is about 60,000 tonnes P (without taking into account the P enrichment with sediment process) which is about 15-20% of the total P displacement (Figure 43).

Figure 42. Phosphorus displacement (kg P ha^{-1} yr^{-1}) due to water erosion in agricultural lands of the EU and UK. The vertical bars show the annual gross P losses (blue), and P deposited (green) per country (tonnes).

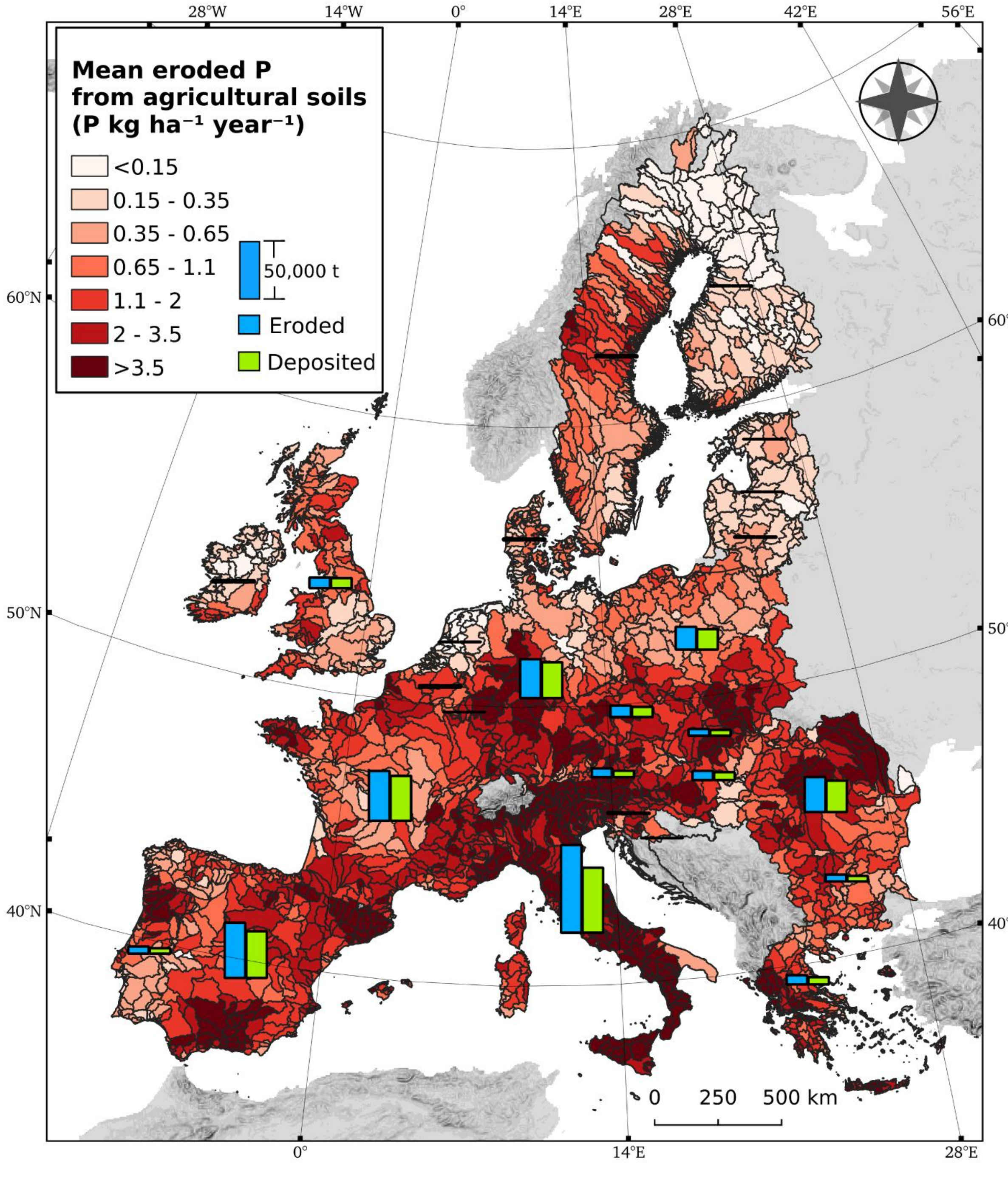
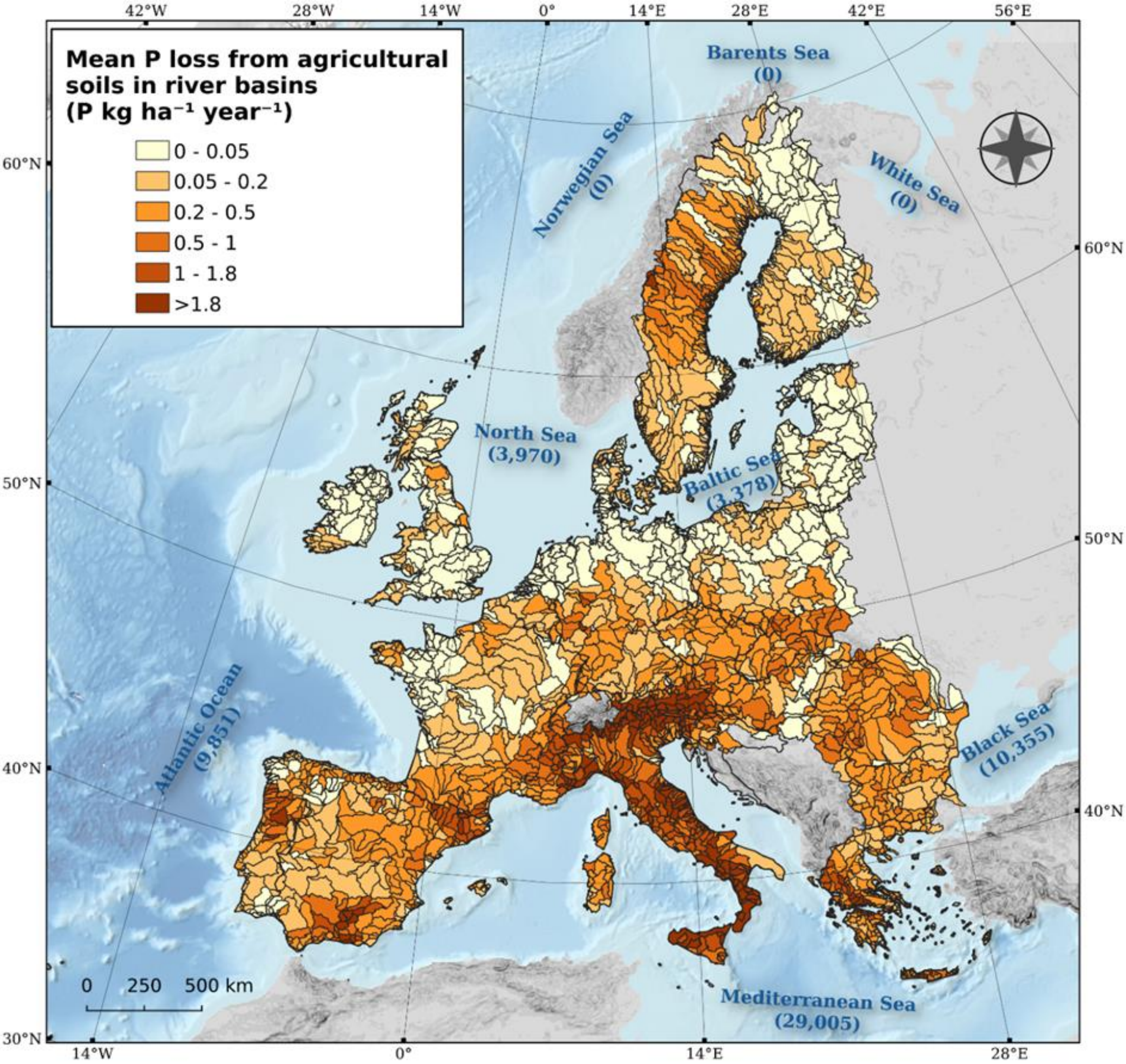


Figure 43. Phosphorus losses to river basins (kg ha−1 yr−1) and sea outlets (Kt yr−1) due to water erosion in agricultural areas. Further analysis is available in Panagos et al. (2022b).



4.6 Scenarios of reduction of nutrient losses to waters (GREEN model)

Nitrogen and phosphorus emissions to European rivers, lakes and coastal waters were estimated over the period 1990-2018 and under different scenarios of nutrient reduction, correspondent to possible measures adopted in the EU. The assessment and scenarios analysis were carried out building synergies between the impact assessment for the revision of the Urban Waste Water Treatment Directive (UWWTD, Pistocchi et al. in preparation), the Blue2.2 project supporting the Marine Strategy Framework Directive (MSFD) (administrative agreements ENV-JRC, Grizzetti et al. 2022), and the INMAP project (administrative agreements ENV-JRC), with the aim of considering measures coherent across different EU policies.

4.6.1 Methodology

The spatial extent covered by the analysis encompasses all river basins draining in European seas, covering in part or completely 44 countries (Figure 44), namely 27 EU countries and 17 non-EU countries. The spatial resolution of the analysis is catchments of 7 km^2 average size (CCM2 catchments and river network, lakes from Ecrins dataset). Results are also aggregated at level of administrative (countries, NUTS2) and hydrological units (river basin districts, marine regions/sub-regions).

Figure 44. Extent of the modelling of nutrient emissions to waters.

The model GREEN was applied for estimating Total N and Total P load from land to surface waters and the seas (Grizzetti et al. 2021; Vigiak et al. 2023). Diffuse nutrient inputs to the river network were estimated spatializing nutrient sources available at administrative level (regional or national) based on spatial maps of land cover (Corine Land Cover and ESA CCI Land Cover time-series v2.0.7). Point discharges of nutrients from domestic and industrial waste waters were quantified following the approach of Vigiak et al. (2020) updated with the latest data reported by Member States under the UWWT Directive (Table 17). A data time series from 1990 to 2018 was built, including gap filling. The time series of annual precipitation, irrigation and water flow was retrieved from the model LISFLOOD (Gelati et al. 2021).

Monitoring data of Total Nitrogen and Total Phosphorus available in the EEA WaterBase for the whole period 1990-2018 (the data are reported by countries to EEA) were georeferenced and used for model calibration (namely, 29560 observations were available for N and 49845 observations were available for P). The model GREEN was calibrated per marine regions, to account for specific biogeographical condition (Vigiak et al. 2023).

Table 17. Nutrient sources considered in the GREEN model.

Type	Nutrient source	N	P	Spatial allocation	Data source
Diffuse	Mineral fertiliser	√	√	Agricultural	CAPRI model historical time series (Barreiro Hurle et al. 2021), FAOSTAT data for countries not covered by CAPRI
	Manure fertiliser	√	√	Agricultural	CAPRI model historical time series (Barreiro Hurle et al. 2021), FAOSTAT data for countries not covered by CAPRI
	Crop fixation	√		Agricultural	CAPRI model historical time series (Barreiro Hurle et al. 2021)
	Soil fixation	√		Agricultural	Fix value set at 4 kgN/ha
	Atmospheric deposition	√		All catchment	EMEP model data from the Norwegian Meteorological Institute
	Background losses		√	All catchment	Fix value set at 0.15 kgP/ha
	Scattered dwellings	√	√	All catchment	Estimated according to Vigiak et al. (2020) updated with most recent data reported by EU27 countries to EEA
Point	Urban waste water discharges + Industrial emissions	√	√	Point discharges	Estimated according to Vigiak et al. (2020) updated with most recent data reported by EU27 countries to EEA Industrial emissions reported in E-PRTR dataset

4.6.2 Scenarios construction

We developed the spatial input data and run the model simulation for several scenarios, correspondent to nutrient reduction measures under different EU policies:

1. Reduction of nutrient discharges from domestic wastewaters. Five scenarios (PS1-PS5) were prepared for the Impact Assessment of the revision of the UWWT Directive. They include the full compliance with the measures established in the UWWTD (PS1), and a combination of additional measures for extending the efficiency of the level of treatment and the extent of the Sensitive Areas (where more stringent treatments are necessary) (PS2-PS5) (Pistocchi et al. in preparation);
2. Reduction of nutrient emissions from agricultural sources. Two CAPRI model scenarios of nutrient reduction were considered, in specific the current CAP (business as usual scenario, capriBAU) and the implementation of the new CAP legislative proposal plus measures to achieve the Green Deal targets also using New Generation EU Funds (capriHAS) (based on Barreiro Hurle et al. 2021);
3. Reduction of nitrogen input from atmospheric deposition (ATM). The scenarios of N atmospheric deposition reduction developed by the EMEP model (described in Section 4.3) was used. It considers the measures adopted by the Commission to reduce atmospheric emissions by 2030 in the Fit for 55 package.

A combined scenario, called INMAX, was run considering the measures of scenario PS5, capriHAS and ATM, i.e. most ambitious reduction measures for domestic wastewaters, agriculture and atmospheric deposition.

4.6.3 Results

The estimation nutrients concentration in European surface waters and nutrients export to the European seas under current condition (year 2016) are shown in Figure 45 and 46, respectively. These maps indicate areas more at risk of N and P water pollution (Figure 45) and the contribution of river basins to the nutrient export to the sea (Figure 46).

Figure 45. Nitrogen (above, mgN/l) and phosphorus (below, mgP/l) concentration in surface water estimated by the model GREEN (year 2016).

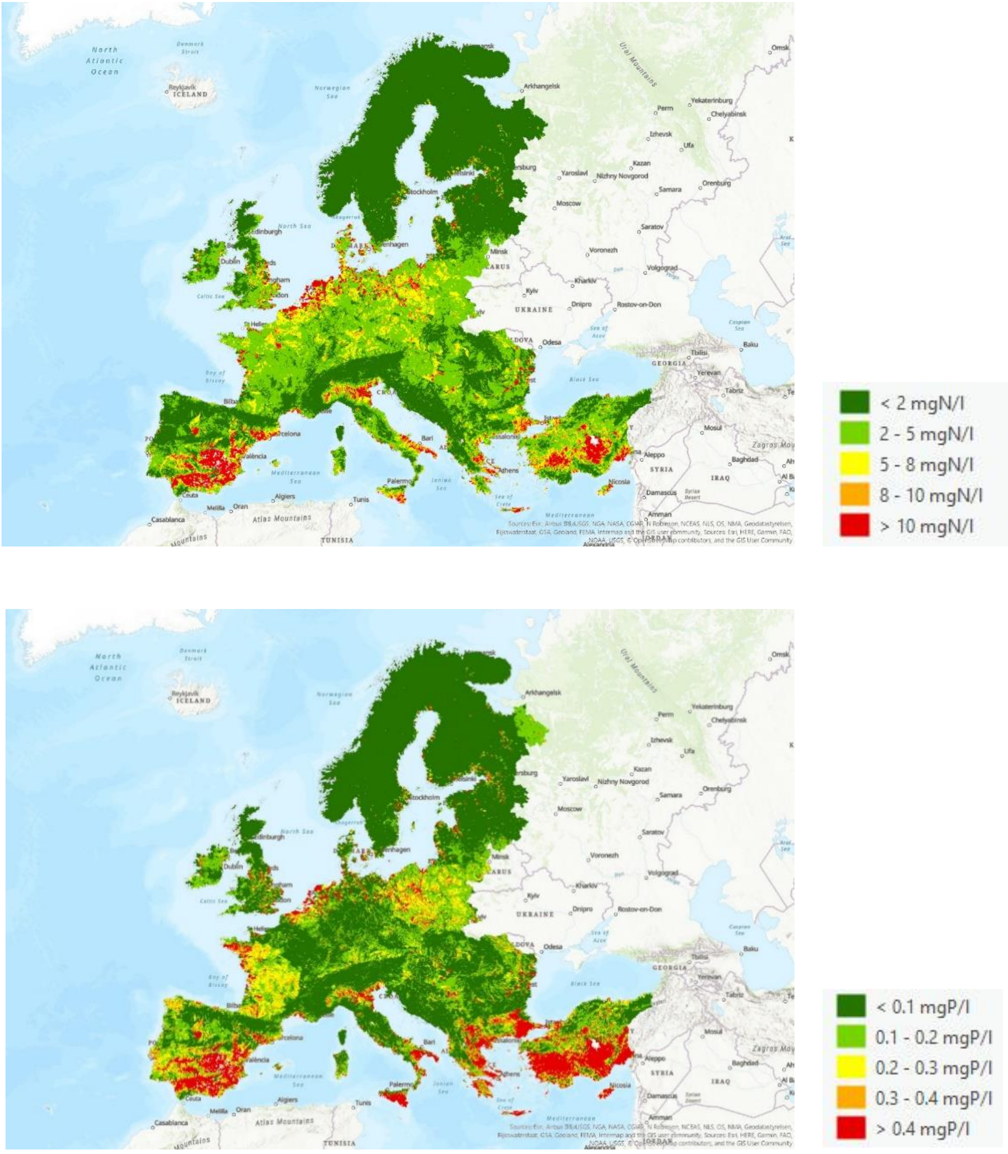
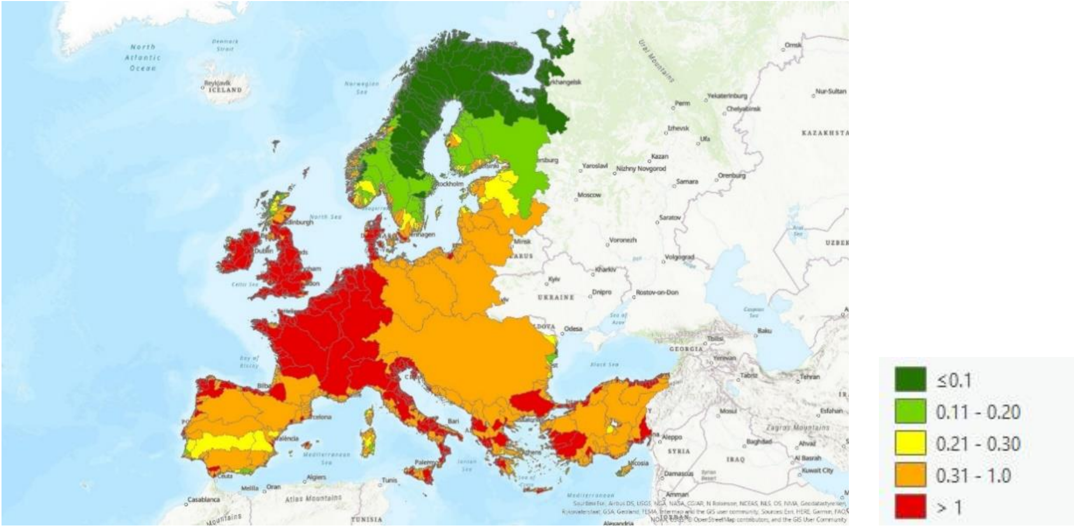
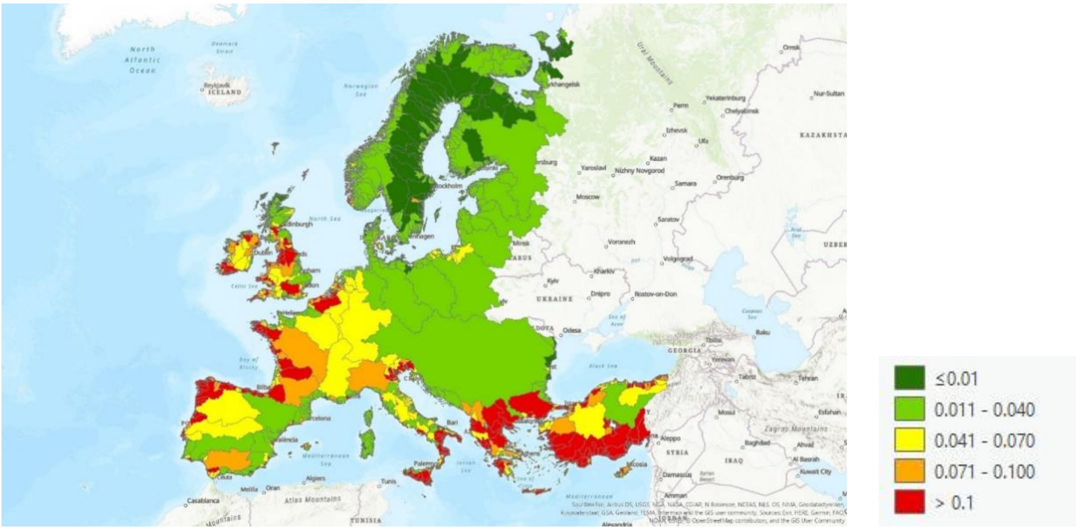


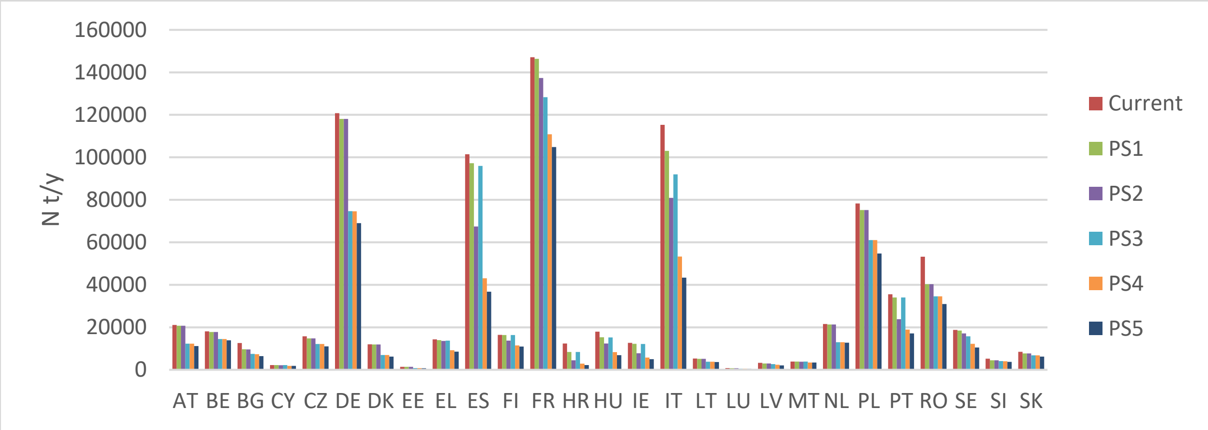
Figure 46. Nitrogen (above, tN/km^2) and phosphorus (below, tP/km^2) specific load to the sea estimated by the model GREEN (year 2016).





The reduction of N and P input per EU27 countries in the different scenarios are shown in Figures 47, 48 and 49. The scenarios represent a reduction of N input compared to current values up to 45% for domestic wastewaters (PS5), 65% for atmospheric deposition (ATM), and 39% for mineral and manure fertiliser application (capriHAS) (average 2014-2018). Concerning P, the reduction is up to 52% for domestic wastewater (PS5) and almost 10% for mineral and manure fertiliser application (capriHAS) (average 2014-2018).

Figure 47. Nitrogen (above) and phosphorus (below) input to surface water from domestic wastewaters (point sources plus scattered dwellings) per EU27 countries under current situation (Current, data of 2016) and five scenarios of reduction: full compliance UWWT Directive (PS1) and a combination of additional measures for extending the efficiency of the level of treatment and the extent of the Sensitive Areas (PS2-PS5).



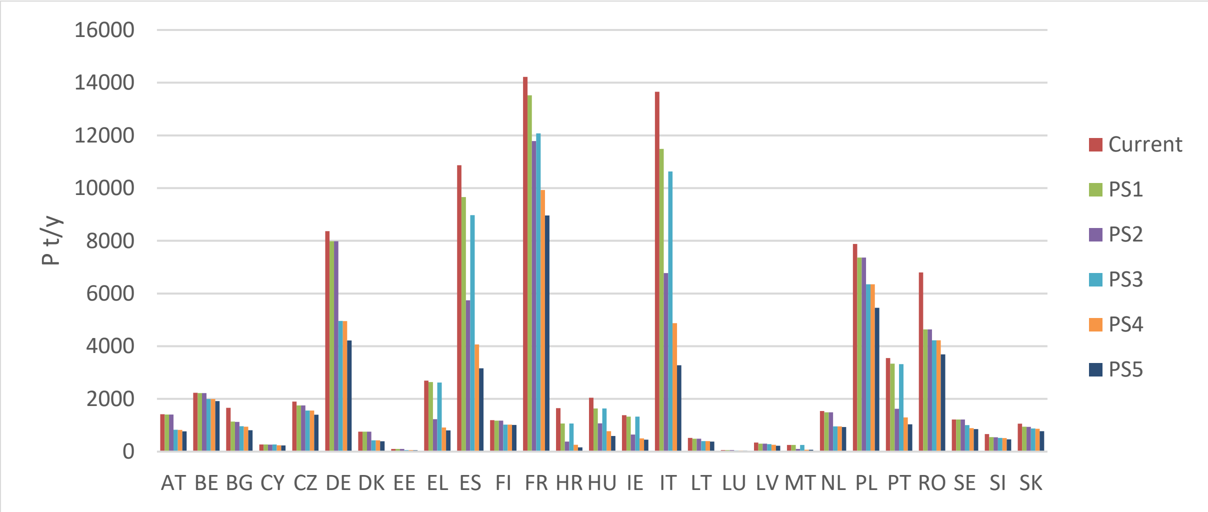


Figure 48. Nitrogen input to land from atmospheric deposition per EU27 countries under current situation (Current, average values 2014-2018) and the scenario Fit for 55 package (ATM).

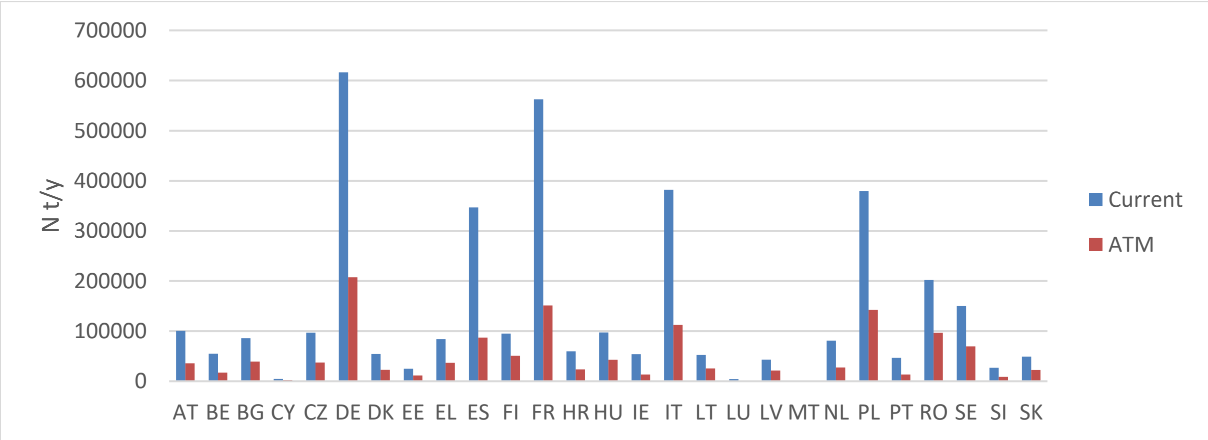
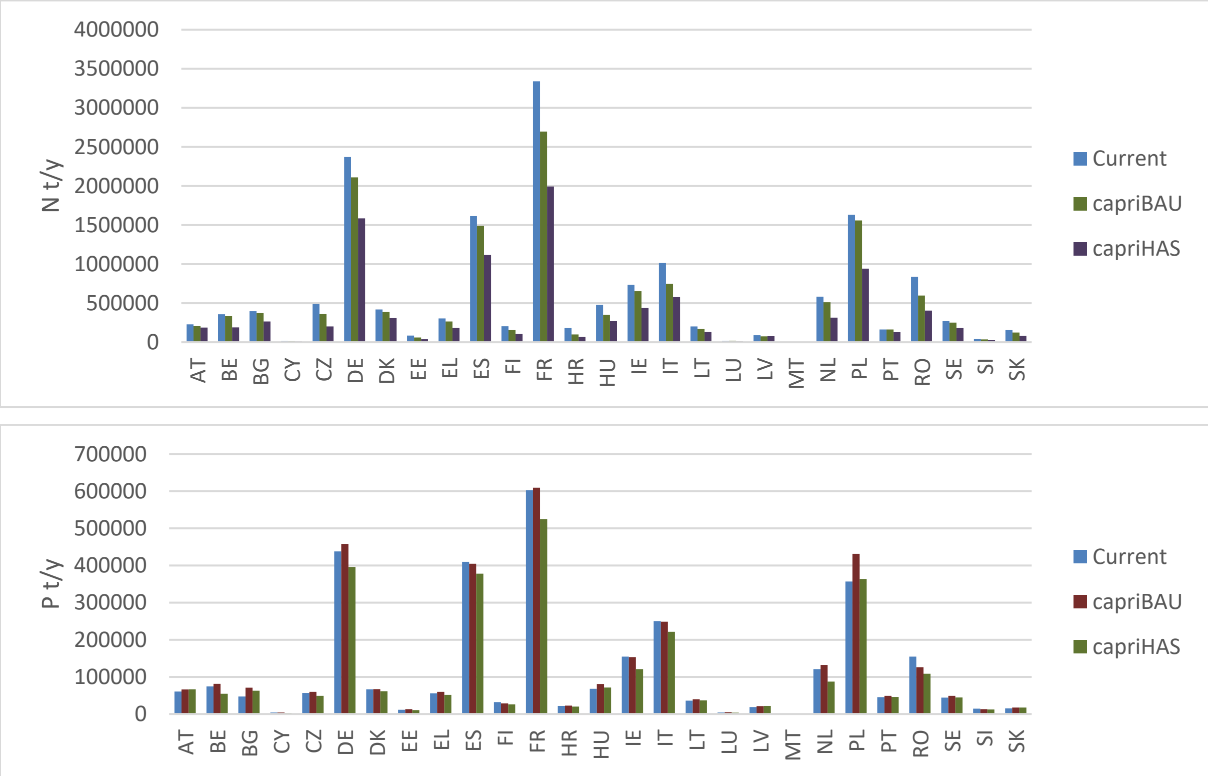


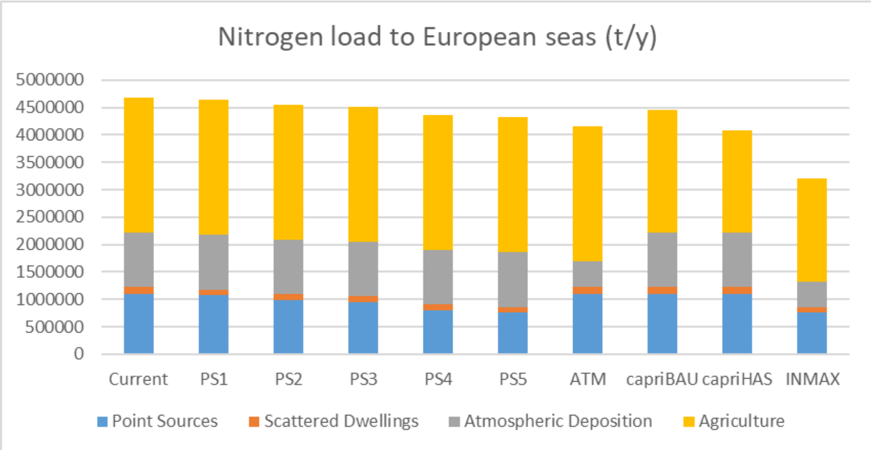
Figure 49. Nitrogen (above) and phosphorus (below) input to agricultural land from mineral and manure input from agriculture per EU27 countries under current situation (Current, average values 2014-2018) and two CAPRI scenarios of reduction: Business As Usual (capriBAU) and implementation of the new CAP legislative proposal plus measures to achieve the Green Deal targets also using New Generation EU Funds (capriHAS).

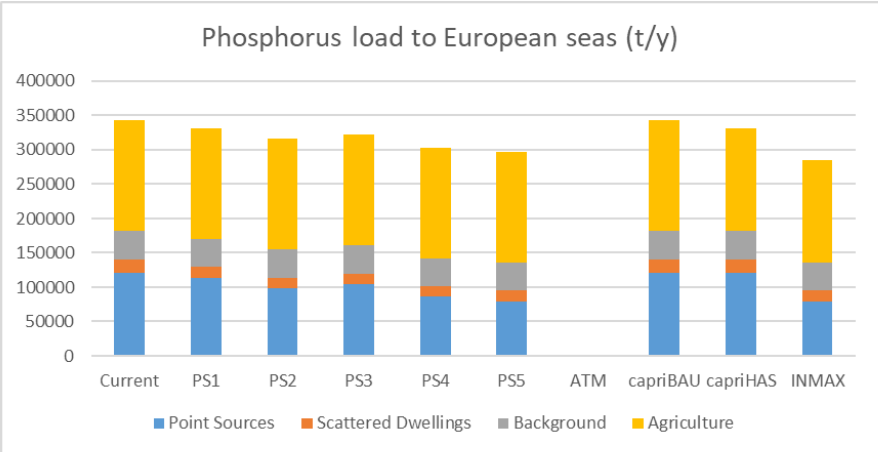


Annual N and P load delivered to European seas under the different scenarios were estimated by the model GREEN (Figure 50). Improvement of domestic wastewaters treatment (PS1-PS5) decreases the nutrient export to the European seas by 8% for N and 13% for P. Reduction of N atmospheric emissions (ATM) could lower the N export to the sea by 11%. Measures under the new CAP and to achieve BDS and F2F strategy targets (capriHAS) could lead to a decrease of N and P load to the seas of 13% and 3%, respectively. Adopting all the measures together (INMAX scenario) could reduce the nutrients load to the European seas around 32% for N and 17% for P. The effect of measures on the N:P ratio need to be considered for the potential impact on coastal and marine ecosystems.

The reductions of nutrient load to the sea are estimated considering the extent of the modelling application (Figure 44), which was established to cover all European marine regions. The reductions are slightly higher (39% for N and 23% for P) if computed considering riverine input only from river basins containing EU territory (as measures take place only on EU27 territory).

Figure 50. Nitrogen (above) and phosphorus (below) annual riverine export from land to European seas (from all study area in the GREEN model) under current condition of nutrient inputs (Current) and the scenarios of measures analysed in the study: improvement of domestic wastewaters treatment (PS1, PS2, PS3, PS4 and PS5), reduction of N atmospheric deposition (ATM), and agricultural measures (capriBAU and capriHAS). Average annual values considering the climatology of 2014-2018. Colours represent the contribution of different sources to the total load. (For phosphorus the scenario ATM is the same as Current).





4.7 Scenarios of current and future European agro-food system (GRAFS model)

An original methodology, known as GRAFS (Generalized Representation of Agro-Food Systems, Billen et al., 2014; Le Noë et al., 2017; 2018) was developed to describe nutrient and carbon fluxes involved in the functioning of agro-food systems at territorial regional or national scales and to establish prospective scenarios based on a number of hypotheses.

An informal group of European scientists, co-authoring the present report, has previously applied this methodology to describe the trajectory of the European agro-food system at the country scale over the period 1961-2014, and outlined an agro-ecological scenario for Europe at the 2050 horizon (Billen et al.., 2021). The present report builds on the previous country-scale work but makes the following important improvements:

1. The present study has a substantially finer spatial resolution (i.e., regions at NUTS 0 to NUTS 2 levels), compared to the previous study, which was based on national data extracted from the FAOstat database. A finer spatial resolution is desirable as national data may hide important regional specialization and specificities of significance for the general functioning of agro-food systems and their environmental impacts. Increasing the spatial resolution required use of additional databases, namely Eurostat and specific national data, using several procedures established by Einarsson et al. (2021).
2. The previous study covered the period 1961-2013, while the EuropeAgriDB1.0 of Einarsson et al. (2021) extends from 1961 to 2019. The present study is based on averaged data from 2014-2019. Working with a 5-year average diminishes the effect of year-to-year climatic fluctuations and thus enables to detect structural characteristics of current agri-food systems across Europe.
3. Some flux estimates have been improved to overcome inconsistencies in the FAOstat data and some simplifications in the analysis of the previous study. In particular, in the present study permanent crops are treated separately from arable crops, as they behave quite differently from the latter with respect to water pollution. Further, permanent grassland is more explicitly included in the present study, while in the previous study, the production and N budgets of permanent grasslands were quite imprecisely determined in the analysis of the current situation, and kept constant in the scenarios.
4. Building on the improvements outlined above, the present study evaluates additional scenarios of possible effects of ongoing European policies, compared to both Business as Usual trends and an updated version of the previous Agro-Ecological scenario.

This report briefly describes the GRAFS methodology, its application to the description of the current European agro-food system at regional resolution, and the development of 5 scenarios at the 2050 horizon. The set of scenarios includes a business-as-usual scenario, an ambitious agro-ecological scenario, and a projection of the application of the EC Farm to Fork and Biodiversity Strategies. The scenarios are compared in terms of structure of the agro-food systems, extra-European trade exchanges and environmental impacts such as N2O emissions, NH3 volatilization and nutrient losses to the hydrosystem.

4.7.1 Methodology

4.7.1.1 The GRAFS model

GRAFS consists of a set of functional relationships allowing establish a comprehensive scheme of material flows (N, P, C) between agricultural soils, livestock systems and human consumption in a given territory. The approach begins with the establishment of a full budget of nitrogen fluxes driven by agricultural production in arable cropland, permanent crops, permanent grassland, livestock systems and human nutrition. Nitrogen is considered as the main limiting factor of agricultural production, but the corresponding fluxes of C and P can be estimated as well. The budget is based on input data obtained from available agricultural statistics. The obtained description of the current system is used as a reference which allows calibrating some of the relationships (e.g., the determination of Ymax, and livestock conversion efficiencies, see below), and calculating the missing fluxes such as the required external trade exchanges and environmental nutrient losses. The relationships calibrated on the 2014-2019 reference situation are then used to calculate the operation of the agro-food systems submitted to other constraints or input data, thus representing counter-factual or prospective scenarios. The hypotheses defining these scenarios range from simple business as usual assumptions with only small change in constraints, to changes in farming practices or even to profound structural changes of the system.

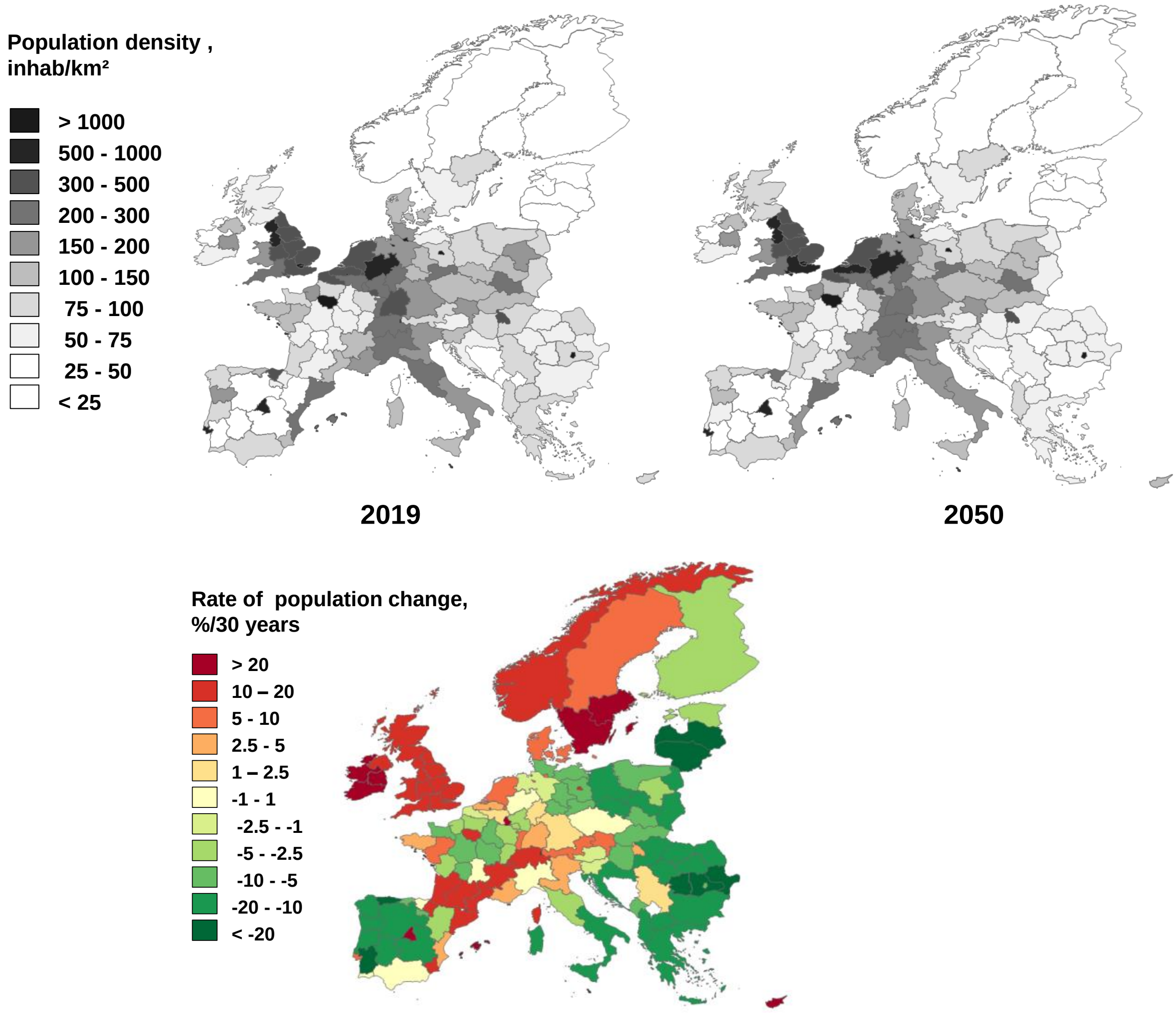
4.7.1.2 Spatial resolution

In this study, we refer to “Europe” as the ensemble of countries located inside the outermost borders of the current European Union thus including 540 million people from the current EU27 plus UK, Norway, Switzerland, Albania, Serbia, Montenegro, and North Macedonia. The spatial resolution for the GRAFS analysis of Europe was guided by the need for representing intra-national specialization in agricultural systems which is often the cause of environmental problems, but also by the recognition that some hypotheses made in the scenario construction, such as regional agricultural autonomy and reconnection of crop and livestock farming, are only meaningful at a reasonable size of the entities considered. The chosen spatial resolution involves 127 geographical units (GU) with similar agricultural area (between 1 and 2.5 Mha) and corresponding to NUTS0, NUTS1 or NUTS2 units according to the countries.

4.7.1.3 Population and diet

The geographical distribution of population between the different geographical units (GU) in 2014-2019 and predicted for 2050, was obtained from Eurostat demographic statistics and prospects (Figure 51).

Figure 51. a. Current and expected population density in the different European GUs. b. rate of change predicted between the two periods (Eurostat demographic statistics and prospects).



National figures for the human diet were considered, as discussed in Westhoek et al. (2015) and Billen et al. (2021). The current average European diet is summarized in Table 18. For one of the prospective scenarios (S1), a different, healthier, diet is considered, differing from the current one by a lower share of animal proteins (30% instead of the current 59% of total apparent consumption), hence a higher share of cereals, grain legumes, fruits and vegetables (Table 18).

The values of Table 18 are apparent consumption figures based on calculated supply to households and restaurants etc. We consider that at present about 30% of the apparent N consumption is not ingested by humans (hence not excreted). In the current situation, most of these wastes are incinerated together with other household solid wastes (Esculier et al., 2018). In the agro-ecological scenarios, it is considered that 50% of the solid food wastes are recycled for feeding pigs and poultry.

Table 18. Current and prospective average European diet (in terms of apparent consumption)

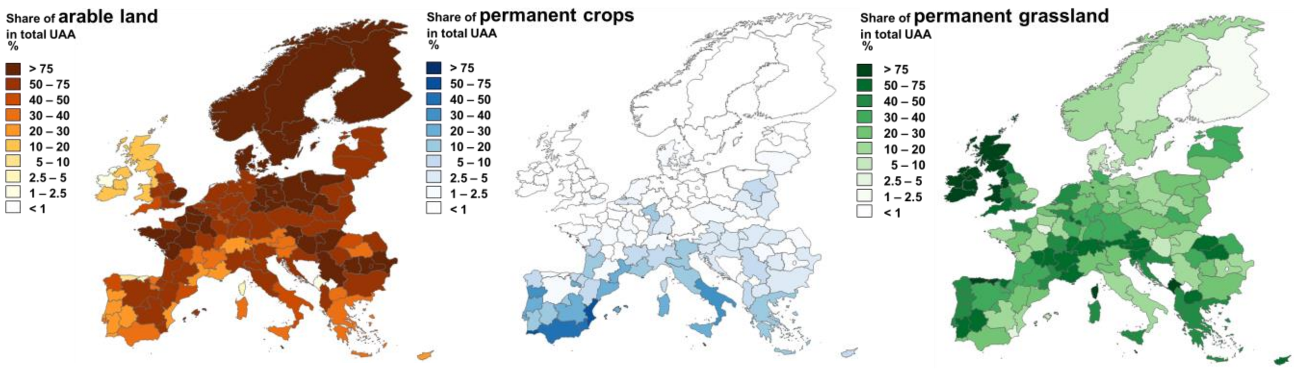
			2014-2019 reference					**Agro-ecological scenario**				
			Apparent consumption					Apparent consumption				
	kcal/g	*%N*	**g/d**	**kcal/d**	gN prot/d	**kgN/cap/y**	*%*	**g/j**	**kcal/d**	gN prot/d	**kgN/cap/y**	*%*
Vegetal products						**2.35**	***41***			9.6	**3.5**	***70***
Cereals	*3.5*	*2*	240	840	4.8	1.75		285	998	5.70	2.08	
Grain legumes	*1*	*3.5*	15	15	0.5	0.19		59	59	2.07	0.75	
Roots and tubers	*0.8*	*0.25*	100	80	0.3	0.09		80	64	0.20	0.07	
Fresh vegetables	*0.3*	*0.3*	150	45	0.5	0.16		300	90	0.90	0.33	
Fruits	*0.3*	*0.15*	140	42	0.2	0.08		200	60	0.30	0.11	
Nuts	*6.5*	*1*	20	130	0.2	0.07		50	325	0.50	0.18	
Animal products						**3.15**	***54***				**1.25**	***25***
Dairy products	*0.8*	*0.7*	565	452	4.0	1.44		224	179	1.57	0.57	
Meat	*1.5*	*3.25*	125	188	4.1	1.48		50	74	1.61	0.59	
Eggs	*1.4*	*2*	30	42	0.6	0.22		12	17	0.24	0.09	
FishSeafood	*1*	*2.9*	27	27	0.8	**0.29**	***5***	23.5	24	0.68	0.25	***5***
non protein food						**kgNeq/cap/y**(1)					**kgNeq/cap/y**(1)	
Added sugar	*3*	*0*	50	150	*0.65*(1)	*0.24*(1)		20	60	*0.26*(1)	*0.09*(1)	
Oil	*7*	*0*	40	280	*3.00*(1)	*1.10*(1)		40	280	*3.00*(1)	*1.10*(1)	
Total			1502	2291	19.5	**5.78**	***100***	1343	2229	17.0	**5.0**	***100***

(1) kgNeq (i.e. kgNequivalent)Nequivalent in corresponding harvested product before extraction of oil or sugar (0.075 equN/kg oil and 0.013 kgN/kg sugar)

4.7.1.4 Crop and grassland production

Three types of agricultural systems are considered separately in the GRAFS analysis: arable cropland, permanent crops and permanent grassland. Figure 52 shows their distribution among the GUs.

Figure 52. Share of arable land, permanent crops and permanent grassland in total UAA in the 127 geographical Units (GU) considered in the GRAFS analysis.



For the 2014-2019 reference situation a full budget of N exported with harvest and N input to the soil as synthetic fertilizers, manure, atmospheric deposition and symbiotic nitrogen fixation was established for each of the 3 land use classes in each GU, based mainly on data available in the Eurostat database, supplemented by the compilation of national databases.

Harvested crop production from arable land and permanent crops was assembled from Eurostat data. Assembling a comprehensive data set of permanent grassland productivities and biomass extraction from grassland required a critical review of the literature, due to the general lack of data on permanent grassland yields and in some cases inconsistencies between data sources which likely to a large extent are due to the wide diversity of natural conditions and management practices collected in the broad category of permanent grassland (see, e.g., Smit et al.., 2008; Eurostat, 2013; Huyghe et al., 2014; Velthof et al., 2014; van den Pol-van Dasselaar et al., 2015; Qi et al., 2017; Einarsson et al., 2020, 2021et al).

Data on total inputs of synthetic N and P fertilizers per GU were collected from Eurostat, FAOSTAT, and a number of national statistical databases. The total quantities of fertilizers were then divided between arable land, permanent crops, and permanent grassland using a procedure similar to that of Einarsson et al. (2021).

The fate of excreted manure by ruminant and monogastric livestock was calculated according to the flow-chart of Figure 53.

Symbiotic N2 fixation by legume crops is calculated from the N yield, using the relationships established by Anglade et al. (2015b) and Lassaletta et al. (2014). For the case of permanent grassland, a new approach was developed taking into account (i) a variable proportion of legumes in the species mix of grassland and (ii) a variable fraction of N derived from atmospheric N2, in response to external N inputs; (iii) a root/shoot ratio of legumes of 2 in permanent grassland.

The principle behind the prediction of crop yields in scenarios in the GRAFS approach is that, for a given cropping system in a given pedo-climatic context, there exists a robust relationship between yield (Y) and fertilization (F), of the form of a single parameter hyperbolic relation:

$$Y = Ymax. F / (F + Ymax) \qquad (1)$$

This is supported empirically by the data of Lassaletta et al. (2014), Anglade et al. (2015a), Billen et al. (2018) who showed that the same relationship (i.e., with the same Ymax parameter) holds for both organic and conventional systems in the same pedo-climatic environment. Therefore, relationship (1), once Ymax has been calibrated on the current situation, allows to calculate Y in future scenarios given the expected amount of fertilizing inputs (F). Based on the N soil budget of the three agricultural systems in each GU, their Ymax parameter can be calculated (Figure 54).

Figure 53. Flows of manure from livestock excretion to application on agricultural areas and losses by ammonia volatilization: a. The processes of ammonia volatilization in relation to synthetic fertilizer application and manure management and application. b. The different pathways from ruminant and monogastric excretion to storage and application to crop- and grassland. Points 1 to 3 (a) and 1 to 5 (b) summarize the methodology for quantitative estimation of the associated N fluxes. Typology of the territorial agro-food systems of Europe.

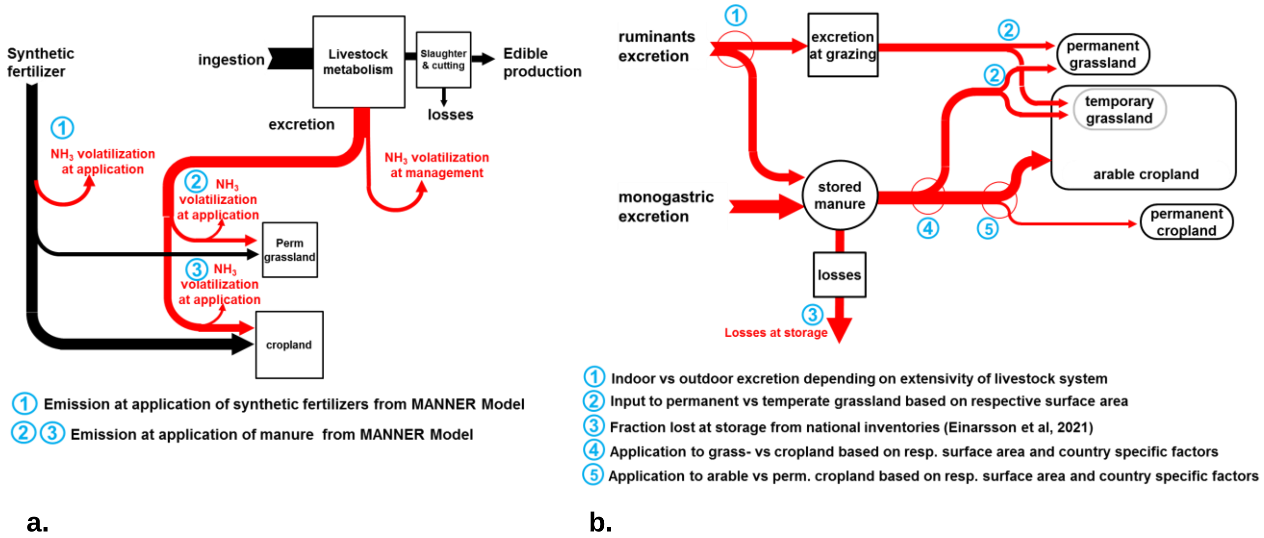
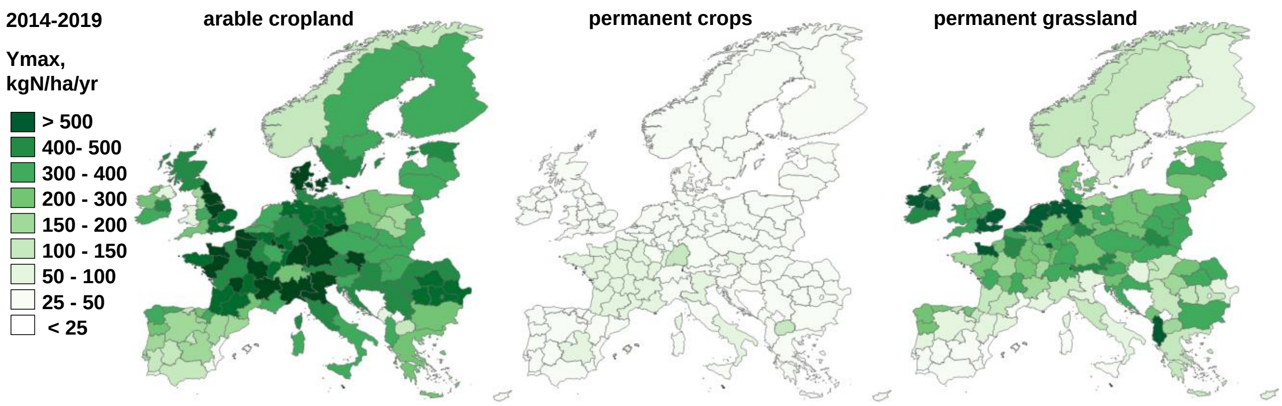


Figure 54. Regional distribution of Ymax of arable cropland, permanent crops and permanent grassland in Europe



4.7.1.5 Livestock number and metabolism.

Livestock populations by categories, and production of meat, milk and eggs were obtained in each GU from Eurostat for the reference situation. Excretion was calculated using national emission coefficients compiled by the Livedate EU project (Eurostat, 2014; Velthof, 2014). We define Livestock Units (LU) as the number of animals excreting 85 kgN/yr (Le Noe et al., 2017, 2018). The calculations of livestock fluxes, carried out separately for ruminants and monogastric, are based on the following mass-balance relationships, relying on two parameters:

1. the conversion efficiency (conveff), defined as the amount of N in edible animal products obtained from the ingestion of one unit of N.
2. the non-edible to edible ratio (nedr) related to the whole animal (with skin, bones and blood)
3. Excretion = 85 kgN/LU/yr
4. Ingestion = Edible production + Non edible production + Excretion
5. Edible production = conveff * Ingestion
6. Non edible production = nedr * edible production

The detailed cutting tables by animal (Le Noë et al., 2017), together with the share of milk vs. meat, and eggs vs. meat, allowed estimating the two parameters in all GU for the 2014-2019 reference situation.

In the scenarios, livestock numbers of ruminants and monogastric were adjusted in each GU to their respective feed resources. For ruminants, these resources consist of the local production of permanent grassland, of forage legumes and other forage crops, plus possible imports of feed from outside the GU (the latter were taken identical to the 2014-2019 reference in the Business as usual scenarios or set to zero in the Agro-ecological ones). For monogastric, potential feed resources consist of 50% of the cereals produced in excess over the requirements of the local human consumption, 80% of the surplus of legume grains, 100% of the surplus of starchy roots, as well as cakes of oilseeds and residues of the sugar industry, together with half the human food waste produced and possible imports of feed from outside the GU. A routine is activated for each GU to define the LU corresponding to local resources, taking into account the fact that these resources partly depend themselves on livestock numbers, because of the fertilization by the manure they produce.

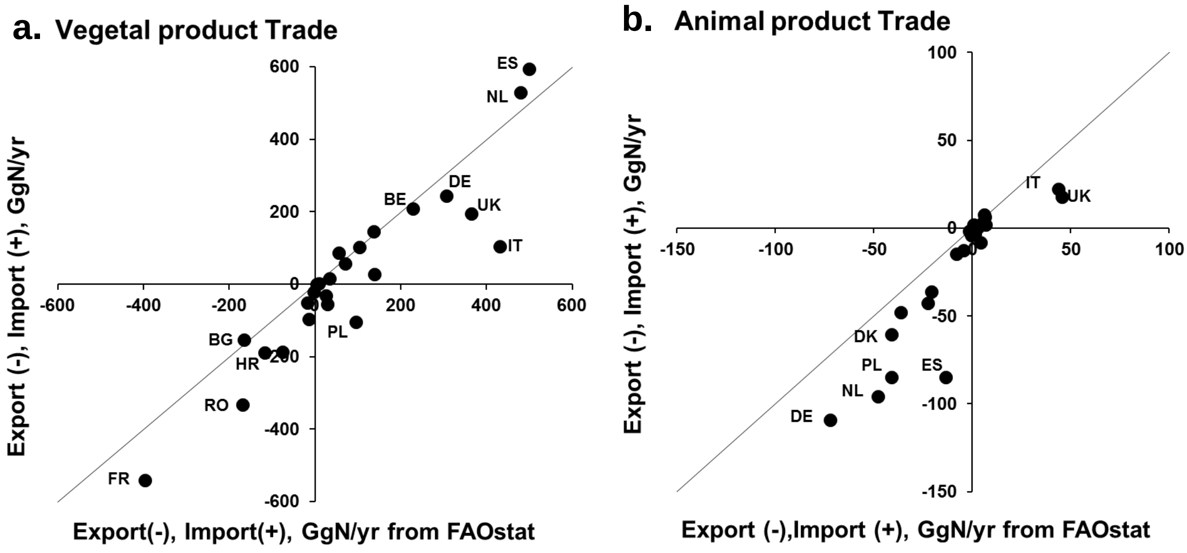
4.7.1.6 Food and feed supply balance

In the absence of empirical data on regional biomass trade, the fluxes of net import/export of vegetal crop products, animal products (meat, milk and eggs) and feed are calculated by the following relationships:

- Net import of crop products = human consumption + animal consumption – local production
- Net import of animal products = human consumption – animal edible production
- Net import of feed = livestock feed requirements not met by local resources

For the reference situation, the net import or export fluxes of vegetable and animal products aggregated by countries from the GRAFS data at GU level (assuming that recourse to intra-national is preferred to international trade) are compared with the data provided by FAOstat for year 2013. Although not perfect, the agreement is reasonable (Figure 55). The model indeed correctly catches the size of import and export trade fluxes of the major countries involved, however better for vegetal (r^2 =0.79) than animal products (r^2=0.69) (Figure 55a, b).

Figure 55. Comparison of Net Import/Export of vegetal (a) and animal products (b) at the national level provided by FAOstat and calculated by the GRAFS model for the reference situation (2015-2019). Line 1:1 is indicated.



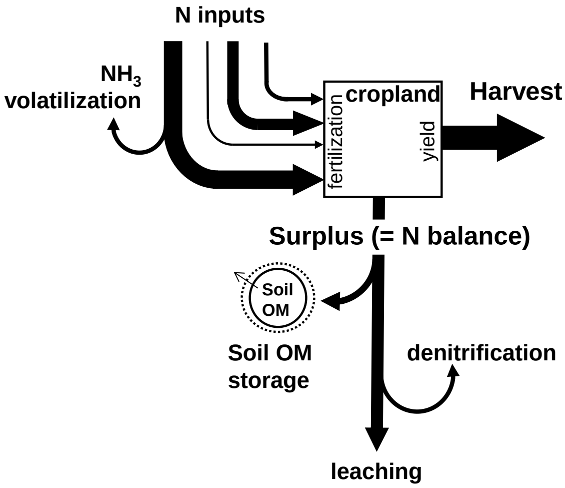
For scenarios, the above relationships are used with the assumption that net feed imports are either equal to the current ones (scenario S2) or zero (scenario S1).

Two indicators are defined: (i) dependence on imports of each GU as the ratio of total net imports (animal+vegetal food and feed) to consumption by humans and livestock, and (ii) export orientation as the share of total food and feed export (if positive) in total food and feed production.

4.7.1.7 Nutrient balance and loss

The gross N soil surplus, defined as the difference between the total effective N soil inputs and the output through harvest, can have different fates, including storage in the soil organic matter pool, denitrification and leaching (Figure 56).

Figure 56. Schematic representation of the surplus and its fate



Denitrification and its associated N2O emissions (EmN2O) are related to total exogenous N inputs (AppNsol, kgN/ha/yr) as fertilizer and manure, annual rainfall (PLU, mm/yr) and mean annual temperature (Temp °C). They can be calculated from the empirical relationship established by Garnier et al. (2019):

Denitrification (kgN/ha/yr) = 4 * EmN2O

EmN2O(kgN/ha/yr) = [0.15 + 0.016 * AppNsol] * (PLU/1000)1.2 * 1.2 Temp°C/10 (4)

Considering that ammonia emissions have been already discounted and the emissions of other gasses are generally low, when the soil organic matter pool is at steady state, leaching can be estimated as the difference between surplus and denitrification.

We defined as net surplus, the N balance (gross surplus) minus the estimated soil denitrification. This can be considered as the best proxy for leaching, at least in the cases where the organic N pool in the soil is in equilibrium. Leaching concentration is estimated considering this annual net N surplus is diluted within the annual leaching water flux, estimated as the average total specific annual water runoff derived from the model LISFLOOD (Burek et al., 2013) for years 2009-2018 (Grizzetti, pers comm).

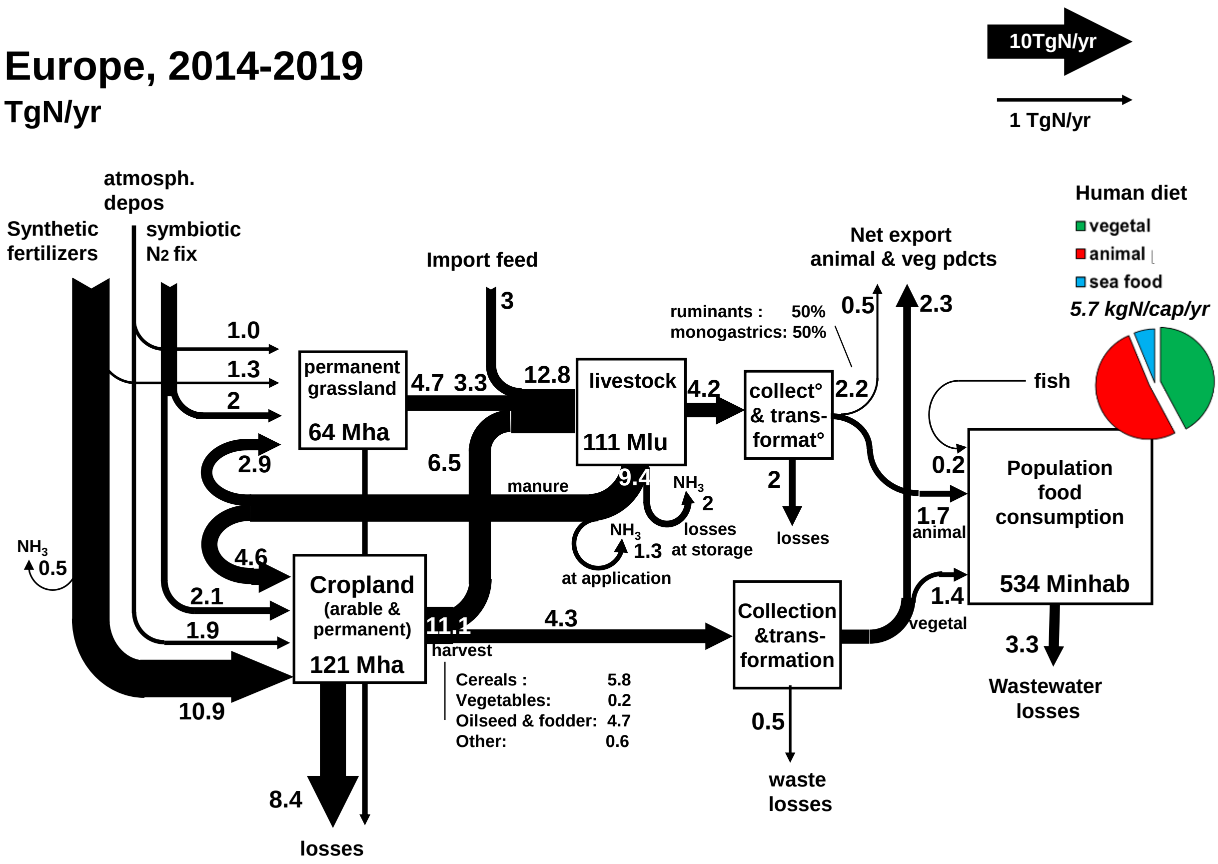
An implicit assumption of our approach is that N is the limiting factor of agricultural production, any other factors being hidden in the Ymax values for each system in each GU. Calculation of the P balance associated to the N-GRAFS representation is however important, because it indicates whether the soil P stock is increasing or decreasing in the absence of erosion. A first estimate of erosion flux (to be made more accurate) shows that this flux is rather low with respect to the P balance, except in mountainous regions with high precipitation.

4.7.2 Reference and Scenarios description

4.7.2.1 The reference European agro-food system

An overview of the flows of nitrogen through the whole agro-food system of Europe as described by the GRAFS approach is provided in Figure 57.

Figure 57. The European agro-food system in 2014-2019, in terms of N flows. All figures shown are obtained as the total of those for each of the 127 territorial units considered.



At the European scale, the reliance on imports of feed (3 TgN/yr), and the capacity of food export (2.3 TgN/yr of vegetal food, 0.5 TgN/yr of animal products) is highlighted.

At the regional scale, a typology of the agro-food systems can be established based on the pattern of major N fluxes between cropland, grassland, livestock and population (Figure 58). This typology, slightly modified from Le Noë et al. (2018), is intended to characterize the degree of coupling between crop and livestock farming as well as between local production and consumption.

A number of indicators of environmental losses of nutrients can be deduced from the GRAFS files. Figure 59 shows the net N surplus of arable cropland soils, permanent cultures and permanent grassland. N_2O emissions from agricultural soils and manure management amount to 366 GgN/yr at the scale of the whole of Europe, with a quite uneven distribution between GUs (Figure 60).

P balance of cropland soil shows contrasted situations in the different GUs (Figure 61), in general agreement with their position in the typology of Figure 58. Most regions of France, Germany and East England which are characterized as specialized stockless cropping systems show negative P balances, in agreement, for France, with the observations and calculations of Le Noë et al. (2018b, 2020). These negative P balances pose no short-term risk of loss of fertility, owing to the very large P legacy accumulated by decades of excessive P fertilization of these soils. By contrast, regions with more important livestock density, often fed with significant import of feed, show positive cropland soil P balance. For grasslands, the contrast between regions is much less marked and few regions display negative values of P balance.

Figure 58. Typology of the current territorial agro-food systems of Europe.

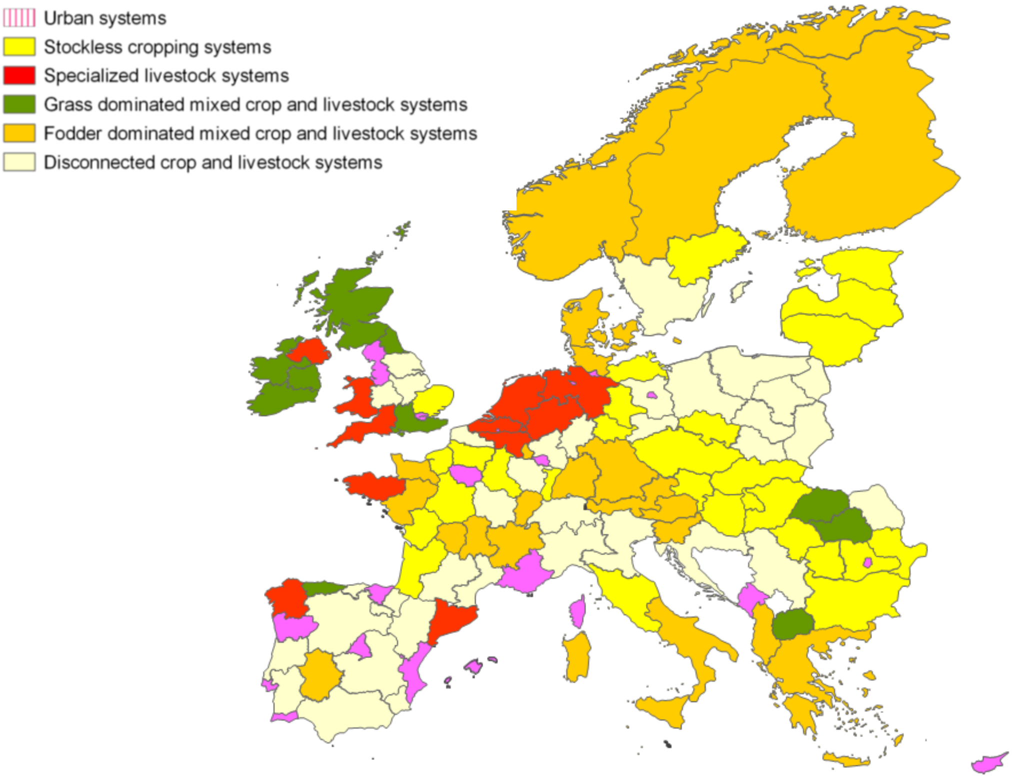


Figure 59. Net N surplus of arable cropland and permanent grassland in the current agro-food systems of Europe.

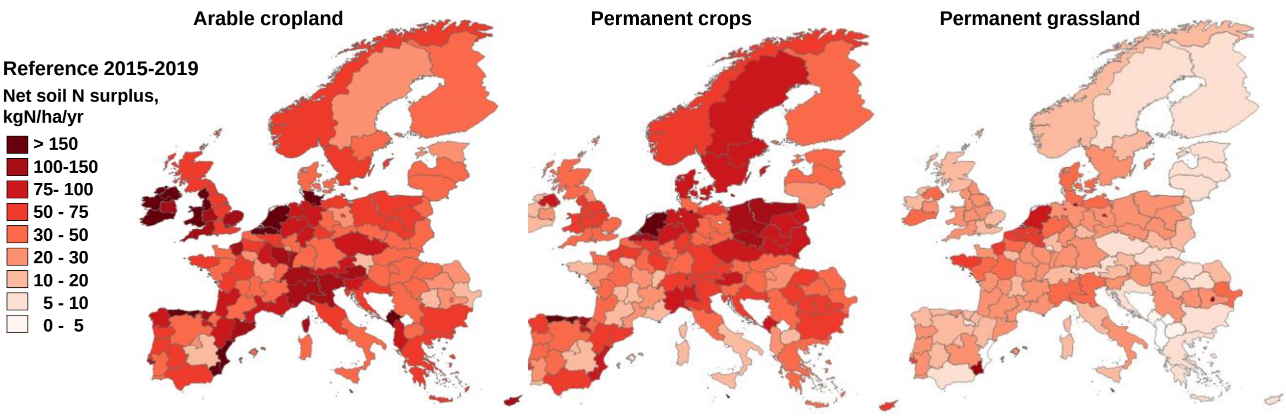


Figure 60. Total N2O emission by agricultural soils and manure management in Europe

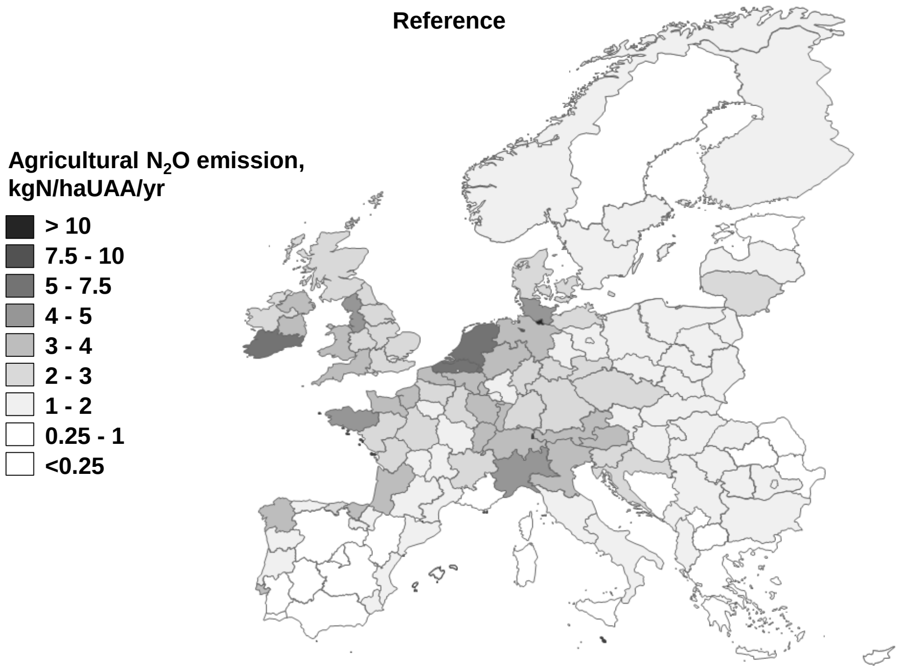
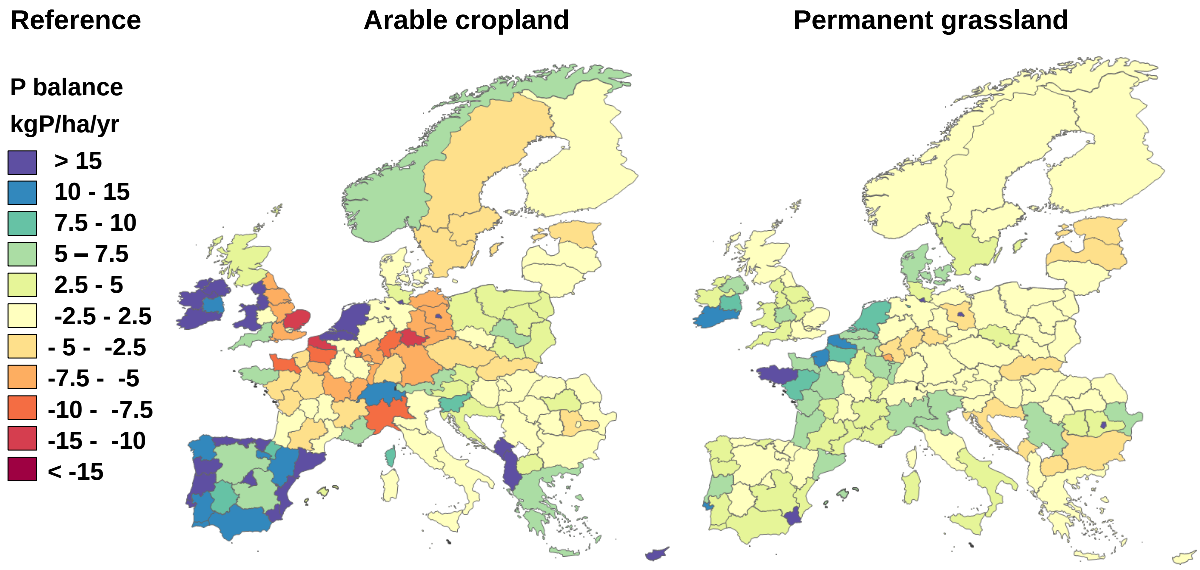


Figure 61. Regional distribution of the cropland (a) and permanent grassland (b) soil P balance in Europe.



4.7.2.2 Prospective scenarios

From the GRAFS description of the current situation, different prospective images of the agro-food system at a certain temporal horizon and under certain constraints can be constructed and their agro-ecological performances assessed. The methodology for the construction of scenarios for future agro-food systems from the GRAFS approach has been published by Billen et al. (2018, 2019, 2021).

The levers that can be operated for exploring alternative agro-food systems concern both functional and structural aspects of the agro-food system, thus allowing to explore a much larger option space than actions limited to improving agricultural practices. These wider functional and structural aspects involve population, diet, land cover, cropping systems and farming practices, and livestock breeding. Note that in the modeling system used in this study, trade exchanges between countries and territorial entities are not per se constrained, but rather calculated as the balance of other calculated fluxes.

In the scope of the INMAP project, three main scenarios have been constructed and explored: a business-as-usual scenario (S2), an agro-ecological scenario (S1), and a scenario aimed at representing as close as possible the results of the EU Farm to Fork and Biodiversity strategies (S3). Below we provide details on how the levers have been operated in the model for simulating these scenarios and the main results obtained. All these scenarios assume the same distribution of arable cropland, permanent crops, and permanent grassland as the present.

In order to validate the calculation procedure, a scenario S0 has been constructed, which is identical by all aspects to the 2014-2019 reference situation regarding the constraints imposed, except that some variables are calculated instead of being provided as input data, namely crop production and livestock numbers and production. The comparison of the results of this scenario S0 with the 2014-2019 reference GRAFS therefore provides an estimate of the margin of errors in the prediction of the scenarios.

4.7.2.2.1 The 2014-2019 reference validation scenario S0

The results of the S0 validation scenario are summarized in table 19.

Table 19. Comparison of the 2014-2019 GRAFS reference (described in previous sections) and the S0 scenario. The S0 scenario uses the same model as the S1-S3 scenarios but is calibrated to reproduce the 2014-2019 reference.

	2014-2019 reference	**S0 scenario**
Crop production, GgN/yr		
Arable crop production	11280	11382
Permanent crop production	173	173
Permanent grassland	4739	5629
Livestock number, MLU	111	137
Edible animal production, GgN/yr	2240	2432
Import (+)/Export(-), GgN/y		
Vegetal food	-2530	-2167
Livestock feed	+2958	+2960
Animal products (food)	-557	-748
Nutrient losses		
Total agric. N_2O emission, GgN/y	366	392
median NO_3^- conc, mgN/l	16.5	17.7
Tot P balance, GgP/y	+378	+485

For variables related to crop production, the agreement with the 2014-2019 GRAFS description is generally within 10%, except for grassland which is overestimated by 18%. Livestock numbers calculated with a calculation routine on the basis of the calculated available resources, overestimate the officially reported values by about 23%. This indicates an overestimation of potential feed resources by about 20% by the scenario construction procedure. This goes together with an overestimation of the same order of the calculated production of animal products, and their net exports.

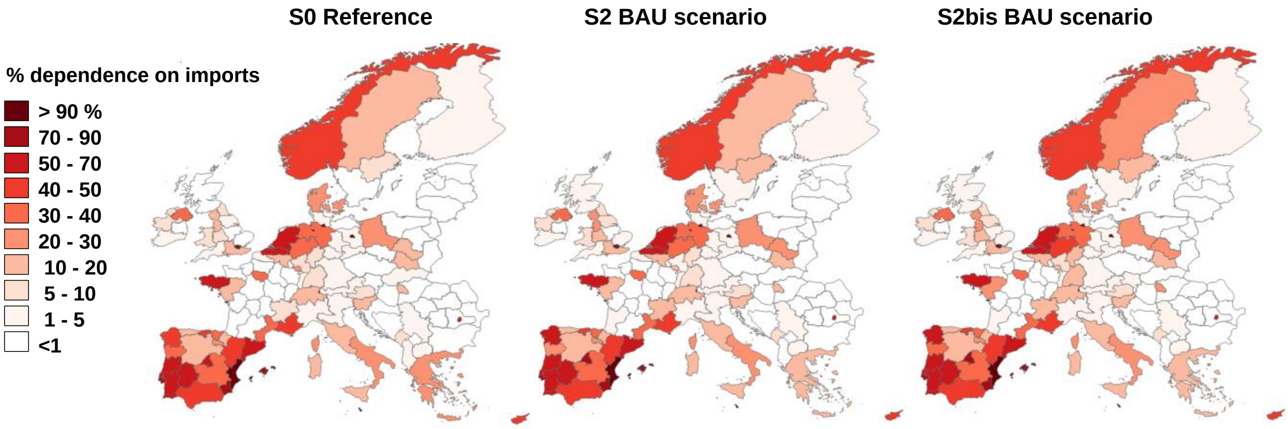
In the following discussion, the calculated S0 scenario will serve as the basis for comparing the scenarios with the current situation.

4.7.2.3 The Business as Usual scenario (S2)

The Business as Usual (BAU) S2 scenario is intended to provide a picture of the agro-food system in 2050 in the absence of significant change in its structure and operating logic, however with the predicted changes in population and associated changes in food demand. Agricultural areas and their division between arable cropland, market gardening, permanent crops and permanent grassland are unchanged with respect to the 2014-2019 reference situation. Human diet is also kept constant at the current values in each country. The rate of synthetic fertilizer application remains identical, and the import of feed from outside each GU is that of the current situation.

We quantify two variants of BAU scenarios to assess the effect of the measures of the F2F and Biodiversity strategy on farming practices that do not involve structural changes: Concerning agricultural N flows, the strongest constraint of the F2F is the objective of reducing by 50% the nutrient losses to the environment. This will clearly require reduced N inputs as well as increased N use efficiency. A figure of 20% reduction in the rate of synthetic fertilizer application is mentioned in the F2F, although the adequacy of such a reduction for reaching the expected goal in terms of nutrient loss is not guaranteed. Here we have assessed the effect on the BAU scenario of a 20% reduction in the application of synthetic N and P fertilizers (S2bis scenario). In both scenarios S2 and S2bis, a slightly increased dependence on food and feed imports is predicted in regions with the highest increased population density (Figure 62).

Figure 62. Dependence on imports of food and feed of the different regions for the reference scenario, the S2 BAU scenario and its variant S2bis with 20% decrease in synthetic fertilizer application.

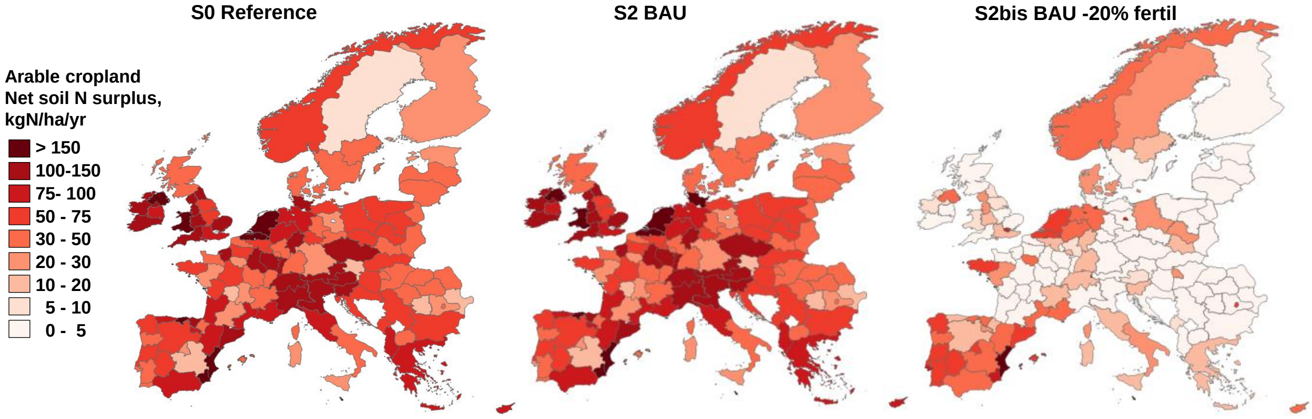


At the European level, while the capacity to export vegetal products (mainly cereals) is not affected by the increase of population (2465 and 2460 GgN/yr for S0 and S2 respectively), the reduction of synthetic fertilization application results in a slight decrease of this capacity (2068 GgN/yr for S2bis). The export of animal products is reduced from 742 and 723 GgN/yr in S0 and S2, to 584 GgN/yr in S2bis.

The effect of reducing N fertilizer input by 20% is substantial in terms of arable cropland N surplus (Figure 63). The median nitrate concentration in leaching water from arable cropland soils is reduced from 17 mgN/L to 14 mgN/l. N_2O emissions are also slightly reduced from 392-396 GgN/yr in the S0 and S2 scenarios to 354 GgN/yr in the S2bis scenario.

The total P balance of agricultural soils is reduced from an excess of 485-477 GgN/yr in the S0 and S2 scenarios to 333 GgN/yr in the S2bis scenario.

Figure 63. Arable crop net N soil surplus in the reference, the S2BAU and the S2bis BAU scenario with 20% reduction of fertilizer application



4.7.2.4 The Agro-ecological scenario (S1)

The agro-ecological scenario operates 3 major levers.

1. The first lever is a change towards a more healthy and equitable human diet, the same in any European regions, including among others a reduction of animal products consumption by more than 50% (see Table 1 above). As this is a very strong hypothesis, a sensitivity analysis to this constraint was carried out, and a variant of the scenario keeping the 2014-2019 reference diet was established (scenario S1bis).

2. The second lever consists of the generalization of agro-ecological farming practices, excluding the use of synthetic fertilizers and pesticides. Symbiotic N fixation by forage and grain legumes is the most important N input to arable cropland soils in this scenario. A review of organic crop rotations in use in the different regions of Europe was established (Billen et al., 2021), based on a thorough compilation of the agronomic literature. On the other hand, Eurostat provides data on the share of cultivated areas and harvested yield of pulses and plants harvested green from organic agriculture for most European countries. The share of grain (λg) and fodder (λf) legumes in the crop rotation (in terms of the fraction of time in the whole rotation cycle) can therefore be defined in each GU, thus describing a typical organic crop rotation for each pedo-climatic context. As the N yield of grain and forage legumes crops (Ylegg and Ylegf) are assumed to be independent on the total soil input, the potential rate of N fixation over a full cycle of crop rotation can be a priori calculated, using the relationship discussed above (Figure 64). Then, knowing the other sources of fertilization (manure and atmospheric deposition) and the Ymax value in each GU, the total crop production Y can be calculated using relation (1) above, and the production of non-legume crops is obtained by difference. In this amount, the proportion of each non-legume crop was taken proportional to its importance in human diet, an assumption consistent with an agriculture primarily oriented toward local human requirements.

3. The third lever of the agro-ecological scenario is the reconnection of livestock and crop farming, which implies that livestock numbers are restricted in each region by the local resources of feed, with no import from outside allowed, and that livestock manure is recycled on agricultural land. The scenario also assumes that 25% of the N content of human excreta is recycled to agriculture.

The S1 scenario implies a profound upheaval of the current agro-food system, as seen in the summarizing representation of the main N fluxes according to the GRAFS approach (Figure 65), to be compared with the similar figure for the reference situation in Figure 57 above.

Figure 64. Rate of symbiotic N fixation by grain and fodder legume crops at the full rotation scale in the agro-ecological scenario S1. The rate is calculated as λf*Ylegf + λg*Ylegg for the typical organic crop rotation in each GU

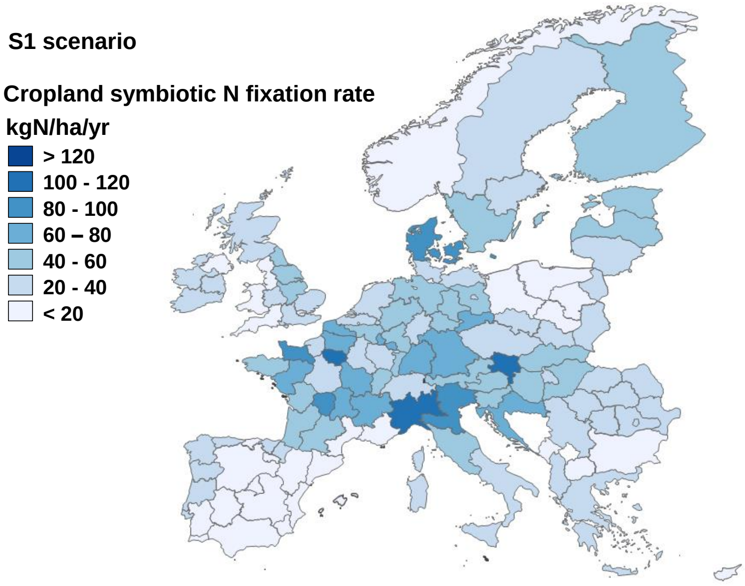
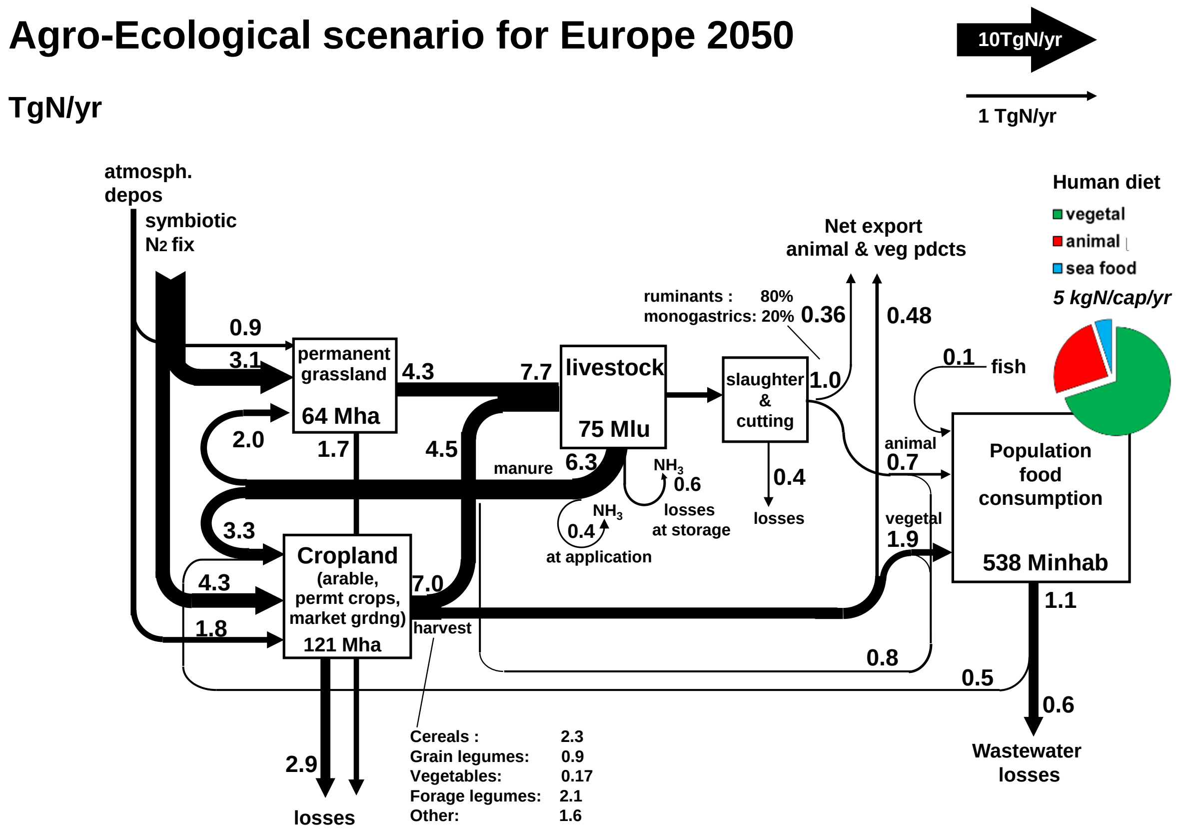
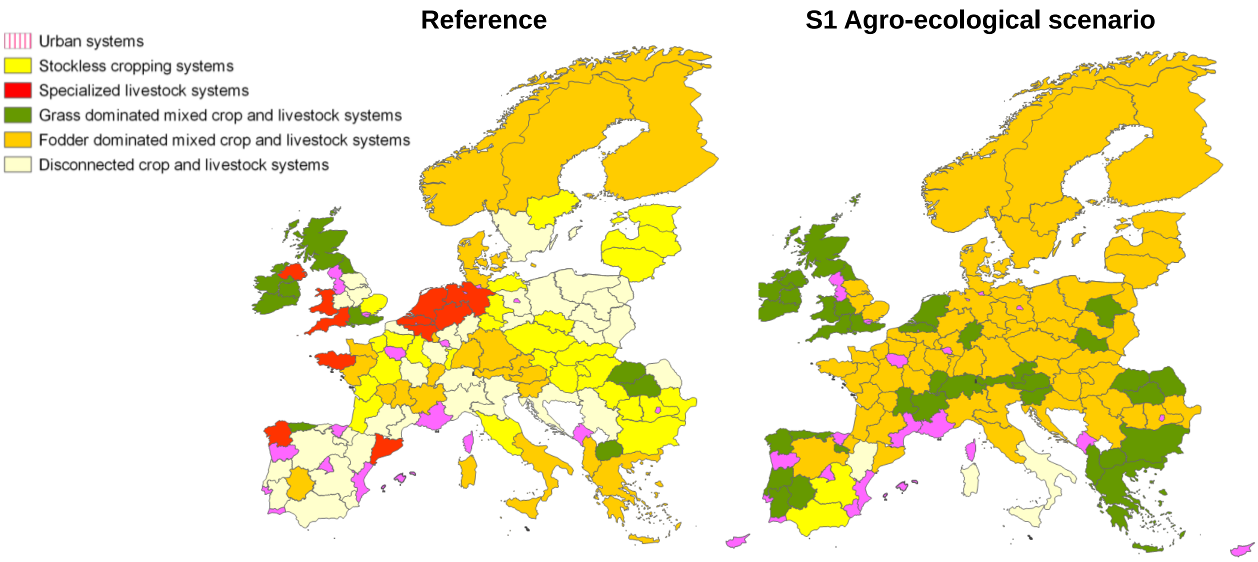


Figure 65. The S1 Agro-ecological European agro-food system in 2050, in terms of N flows.



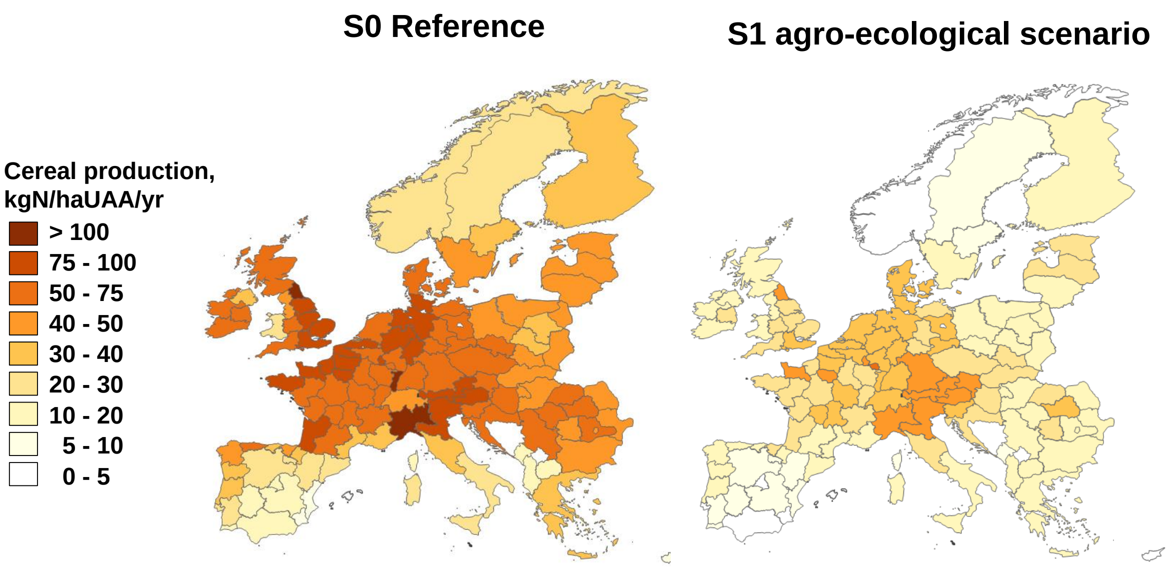
The typology of territorial agro-food systems in the S1 agro-ecological scenario is compared with that of the current situation in Figure 66. As expected, all specialized livestock systems, and most of the specialized stockless cropping systems are replaced by mixed crop and livestock systems.

Figure 66. Typology of the agro-food systems in the reference and the S1 agro-ecological scenarios.



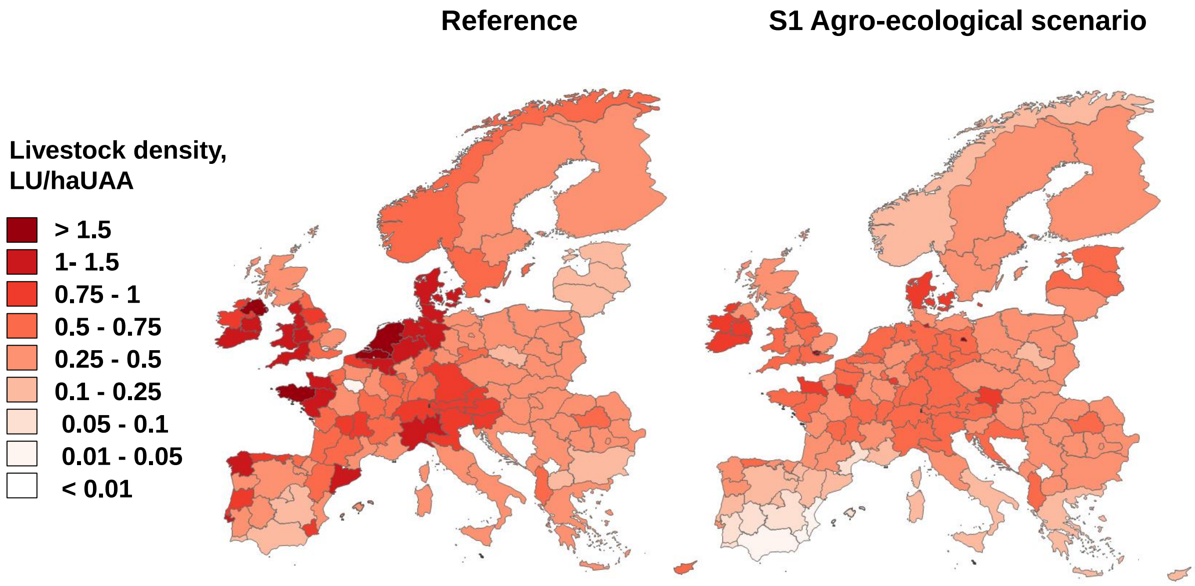
In the agro-ecological scenarios S1, cropland productivity is strongly decreased with respect to the current situation and the BAU scenarios, as shown for example for cereal yields (Figure 67). It must be kept in mind in view of this much lower cereal production, that the use of cereals for livestock feeding is much decreased in the scenario as well, as cereals are by priority reserved for human consumption.

Figure 67. Cereal production per ha of total cropland in the reference current situation (S0) and in the agro-ecological scenario (S1)



Total livestock density amounts 0.40 lu/haUAA in the agro-ecological scenario, compared to 0.74 lu/ha in the S0 reference situation, and is much more evenly distributed (Figure 68).

Figure 68. Livestock density in the 2014-2019 reference (S0) and in the agro-ecological scenario.

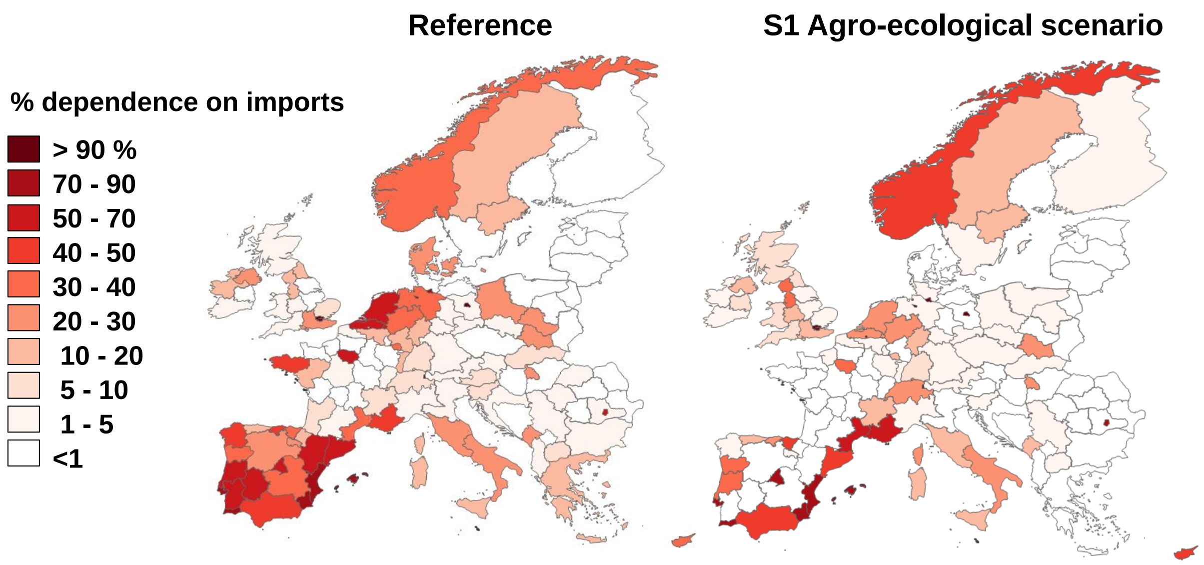


In the agro-ecological scenario, Europe as a whole is not only self-sufficient in cereals (as well as in grain legumes), but can even sustain a small export 370 GgN/yr (compared to 2340 GgN/yr in the S0 current situation).

Also, although no import of feed is allowed in the S1 scenario, Europe is self-sufficient in terms of animal products and can even export meat and milk at about half the current rate (360 GgN/yr compared to 557 and 742 GgN/yr in the 2014-2019 reference and S0 scenario respectively). Obviously, the feed resources generated in the agro-ecological scenarios are such that a higher livestock density can be sustained than required for human nutrition in the hypothesis we made of a largely vegetal based diet. The consequence of this is a large export of animal products. However, the feed resources used for feeding exported livestock could as well be used for other purposes, such as bioenergy or biomaterials.

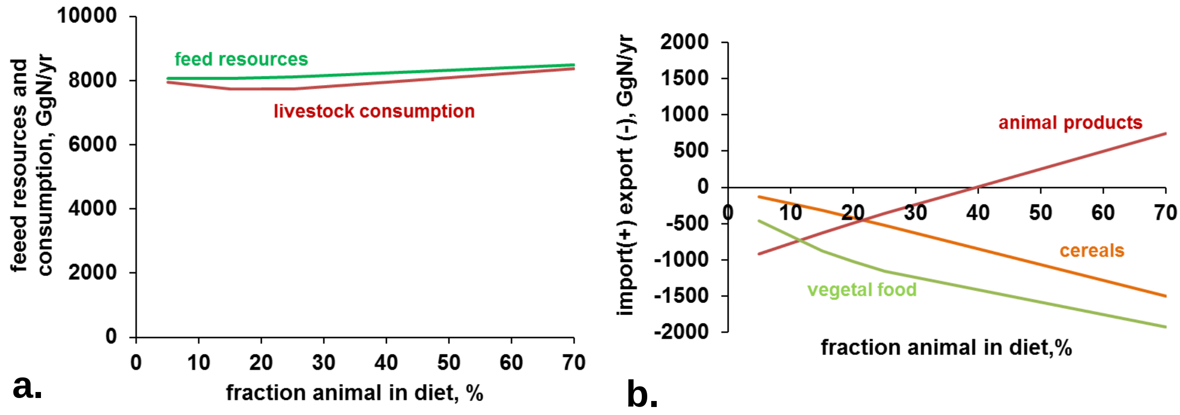
Although the S1 agro-ecological Europe as a whole, is still exporting agricultural products outside its frontiers, all regions are not self-sufficient in the scenario. Figure 69 shows the regions depending on net imports for their food supply. Except for some regions around the Mediterranean Sea, and the highly populated regions with a major city, more regions are self-sufficient in the agro-ecological scenario than in the current situation.

Figure 69. Dependence on food and feed imports (ratio between total animal+vegetal food and feed imports and consumption by human and livestock) of different GUs in the reference situation and in the S1 scenario.



A sensitivity analysis was carried out in order to assess the effect of human diet on the performances of the agro-ecological scenario. The procedure consisted in just varying the proportion of animal proteins in the diet, and re-adjusting livestock numbers accordingly. The results show that changing the diet affects only to a rather limited degree cropland production and the livestock feed resources, hence the livestock density that can be sustained in the absence of feed import. Only the balance between production and human consumption of cereals and animal products is affected, in such a way that the capacity to export or the need to import food is highly and linearly dependent on the diet. Beyond a value of 40% animal products (excluding fish) in the human diet, agro-ecological Europe becomes a net importer of animal products, while its capacity to export cereals is increased (Figure 70).

Figure 70. Sensitivity analysis of the agro-ecological scenario S1 with different shares of animal N products in the human diet.



A variant of the S1 agro-ecological scenario (S1bis) has been established using the same constraints as the S1 scenario, but without change in human diet, i.e., with the same per capita diet as in the 2014-2019 reference situation, but with the population predicted for 2050. In this S1bis scenario, the maximum sustainable livestock number (without feed import) increases only to 80 MLU compared to 75 MLU in the S1 scenario, which is not enough to meet the increased requirement for meat and milk. Compared to the S1 scenario, Europe in S1bis increases its net imports of animal products by 934 GgN/yr, becoming a net importer for 576 GgN/yr of animal products (about 1/3 of the consumption in S1bis) compared to a net export of 358 GgN/yr in the S1 scenario. On the other hand, the capacity to export cereals is increased from 370 GgN/yr in the S1 scenario to 570 GgN/yr in the S1bis scenario. In this context it is relevant to note that each unit of N traded in animal products represents a larger land use and environmental pressure than the same quantity of N in vegetal products. Thus, in terms of externalized land use and environmental pressures, the total increase in import dependence between scenarios S1 and S1bis is very substantial.

The net N surplus of arable land, which represents the most important source of N losses to the hydrosystem is compared in Figure 71 between the reference situation and the S1 agro-ecological scenario. The difference is striking, and indicates much lower environmental losses in the agro-ecological scenario.

The total annual mean N2O emission from agricultural soils and manure management is 55 % lower (157 GgN/yr) in the agro-ecological scenario than in the current reference situation (366-392 GgN/yr in the 2014-2019 reference and the S0 scenario respectively) (Figure 72).

In the agro-ecological scenario, with no input of new P, and in spite of efficient recycling of animal and human manure, more regions are experiencing negative P balances, mostly in croplands, but to a certain extent also in permanent grasslands (Figure 73). Although in many European regions, considerable stocks of legacy P exist in agricultural soils and can sustain production for some time, it is an undeniable fact of mass conservation that some input of P fertilizers would eventually be necessary to sustain production in the long run.

Figure 71. Net N surplus of cropland soils in the 2014-2019 reference and the S1 agro-ecological scenario.

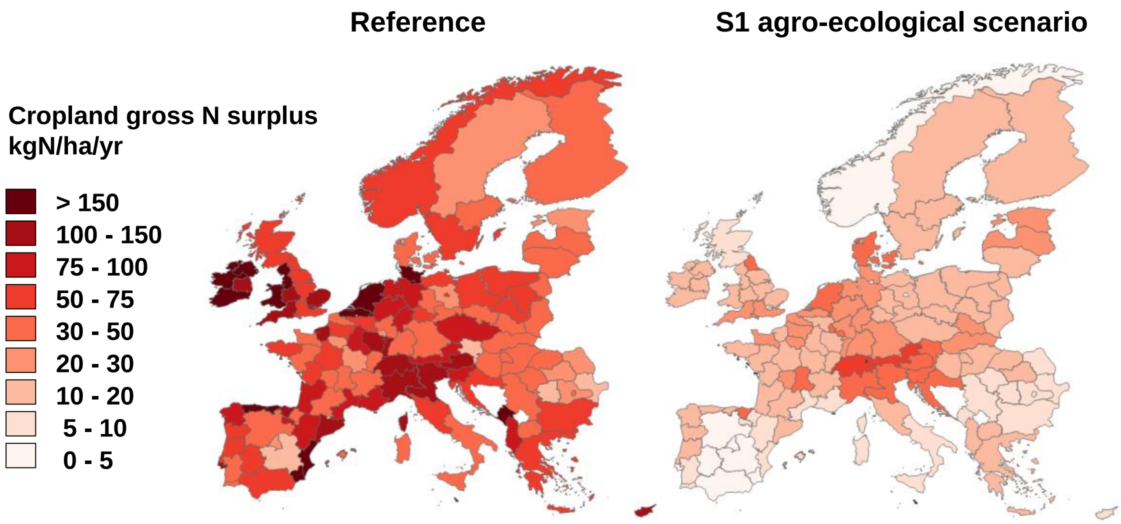


Figure 72. N_2O emissions from agriculture in the 2014-2019 reference (S0) and the agro-ecological scenario.

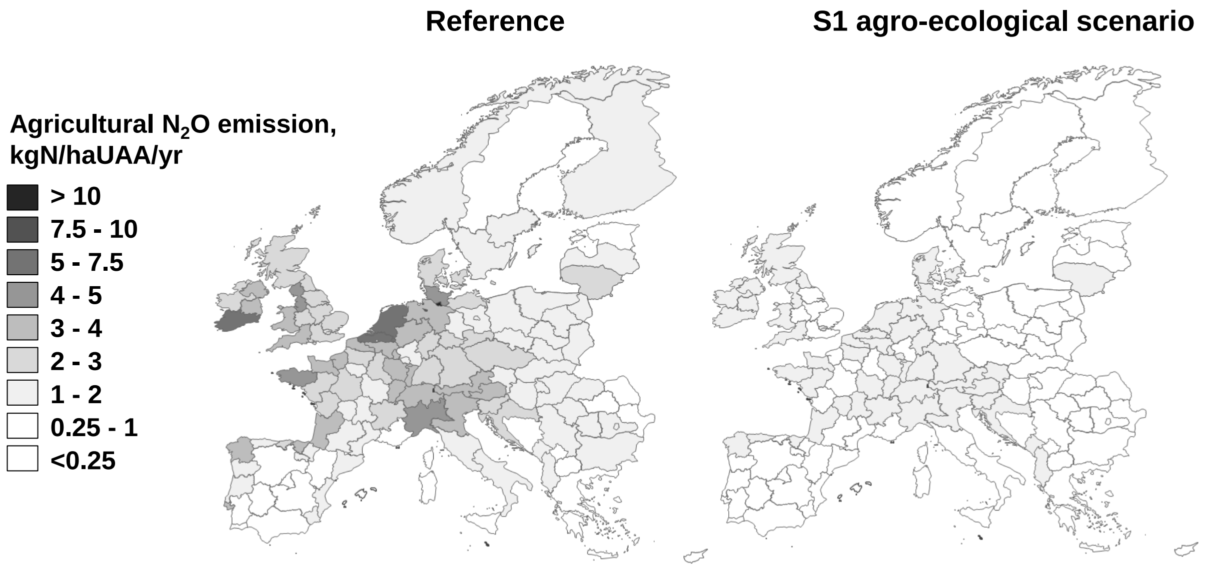
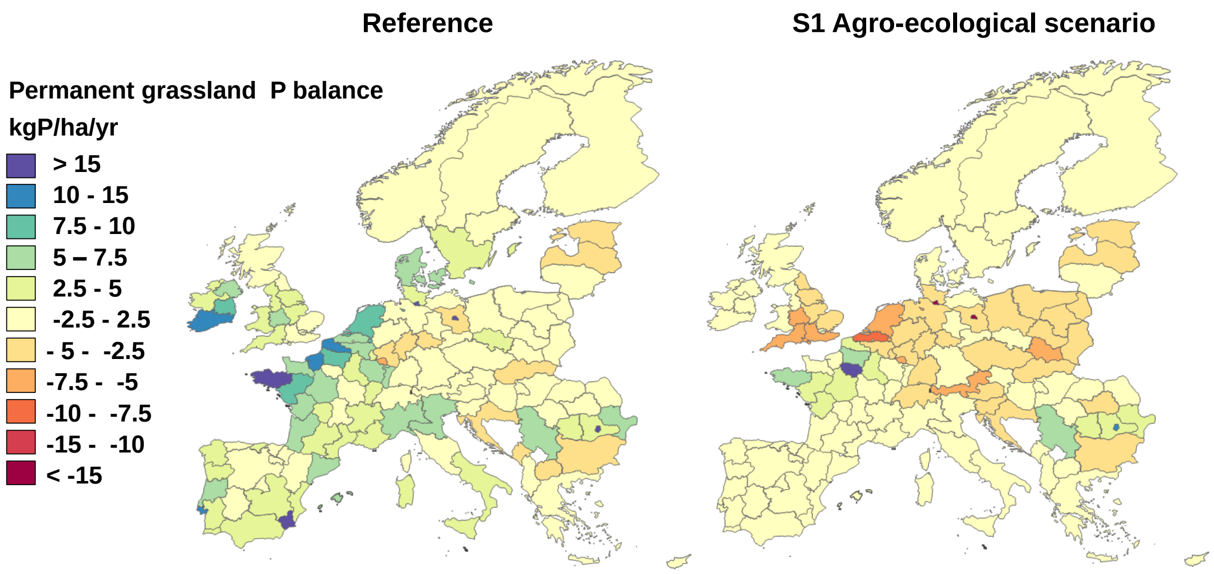


Figure 73. Cropland and permanent grassland P balance in in the reference and the agro-ecological scenario.



4.7.2.5 The F2F scenario (S3)

This scenario is intended to assess the effect of the different measures prescribed in the Farm to Fork and Biodiversity Strategies of the European Commission at the horizon 2030. These measures involve essentially the following:

1. No change in human diet is assumed in this scenario.
2. A reduction by at least 20% of the use of synthetic fertilizer is foreseen in order to reach the objective of halving the nutrient losses to the environment.
3. The share of agricultural area under organic farming management has to reach at least 25%.
4. At least 10% of agricultural area has to be under high-diversity landscape features, e.g., hedgerows or set-aside areas.

The previous scenarios have been established in such a way that the S3 scenario can be constructed as a linear combination of the results of the S1bis and S2bis scenarios. In each GU, the current agricultural area is reallocated as follows: 10% of all surfaces are removed from production and considered as forest. The remaining is allocated to either agro-ecological management (S1bis) or to conventional management with 20% reduction of fertilizer (S2bis). Given that the 2014-2019 reference situation involves already a variable share of organically managed areas, and that the effect of this management is reflected in the current description of the reference and S2 agro-food systems, we consider that measure (iii) implies to allocate to each GU a mix of S1bis (100% organic) and S2bis (GU-dependent share organic) such that the mix has 25% organic area . We calculated this GU-dependent mix of S1bis and S2bis using subnational statistics on 2014-2019 agricultural area shares from Eurostat. In the few GUs where more than 25% of the UAA is already under organic management, all agricultural surfaces were allocated to the BAU S2bis scenario.

The calculation shows an agro-food system considerably modified with respect to the current situation, although less deeply than the agro-ecological scenario, as shown by the resulting typology of territorial agro-food systems (Figure 74; compare to Figure 66 above). A certain degree of reconnection of cropping and livestock systems is apparent by the fact that many specialized stockless cropping regions convert into mixed crop-livestock systems; however, regions of specialized livestock breeding remain.

The de-intensification hypotheses of the S3 F2F scenario result in a drop in crop productivity (Figure 75).

Total livestock density amounts to 0.64 LU/haUAA in the S3 F2F scenario, only slightly decreased from 0.74 LU/haUAA in the S0 reference situation (Figure 76).

Figure 74. Typology of the territorial agro-food systems in the reference and the S3 F2F scenarios.

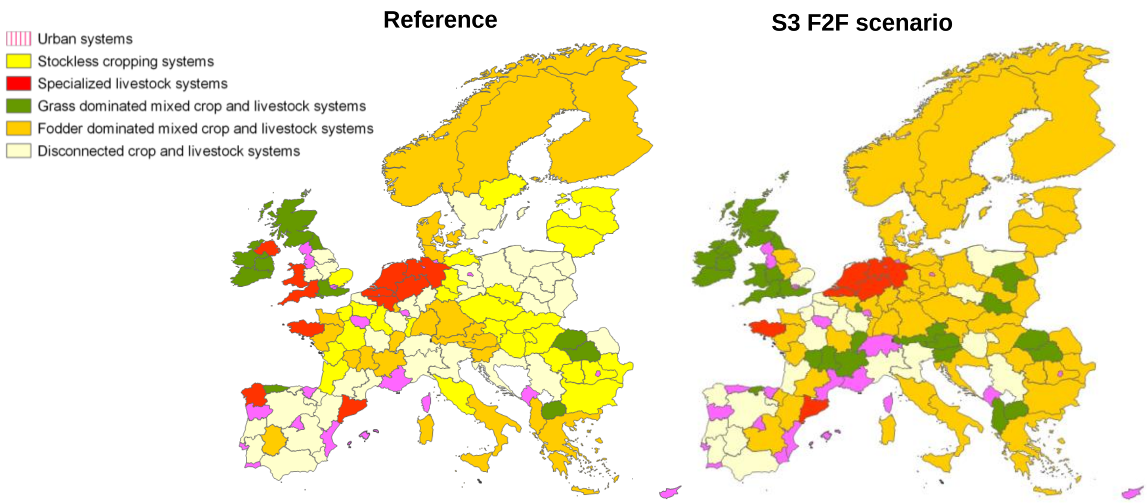


Figure 75. Cereal production per ha of total cropland in the current situation (S0) and in the F2F scenario (S3).

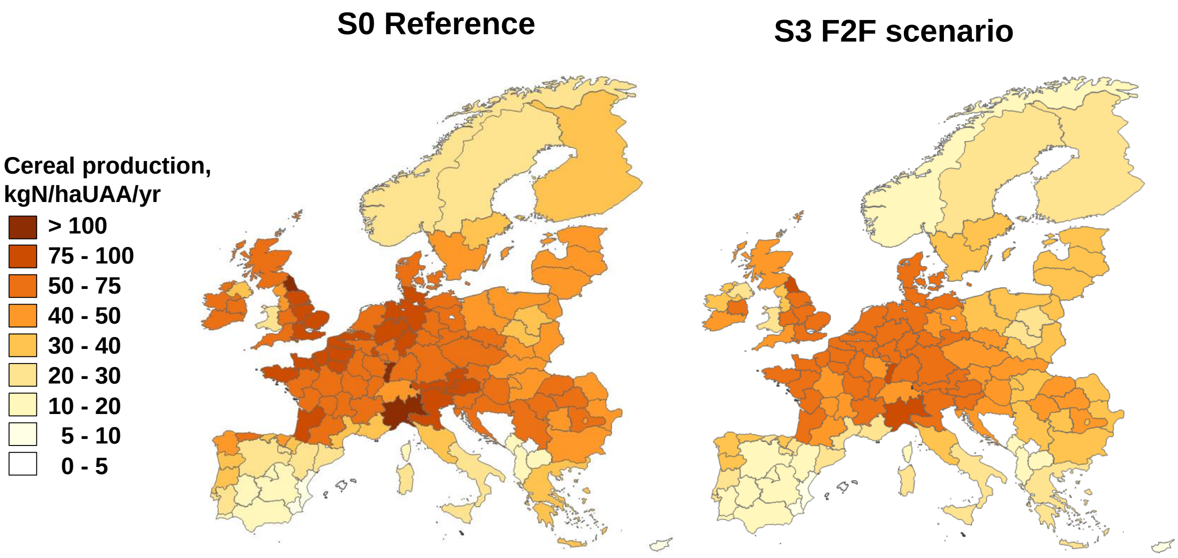
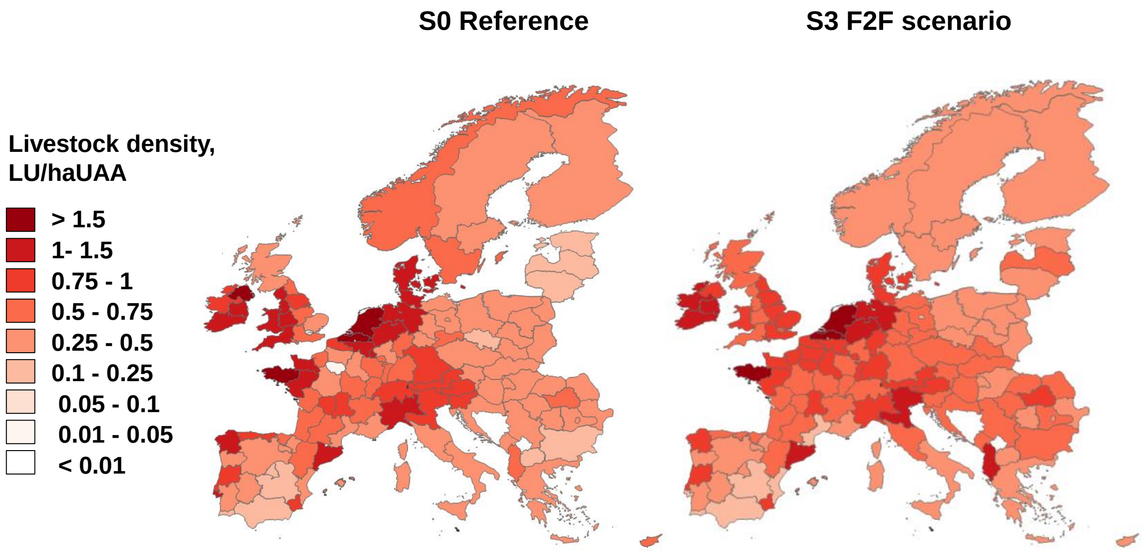


Figure 76. Livestock density in the current situation (S0) and in the F2F scenario (S3).

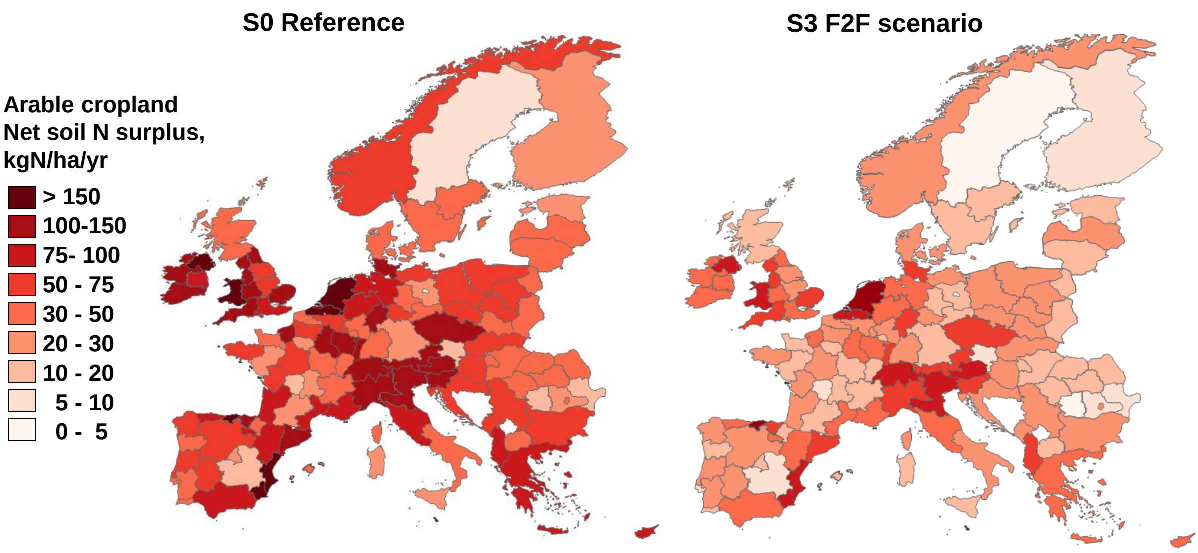


With these characteristics, the F2F scenario is still able to meet the food demand of the European population as well as to export vegetal food at the rate of 1200 GgN/yr (compared to about 2500 GgN/yr in the 2014-2019 situation and in the S2 BAU scenario, and 480 in the agro-ecological scenario). Its capacity to export animal products remains significant although much reduced (133 GgN/yr), compared to the current one (557-742 GgN/yr).

Cropland N surpluses of arable land are reduced considerably (Figure 77). N_2O emissions are decreased to 282 GgN/yr, compared to the current 366-392 GgN/yr.

The total P balance remains positive, but at a much lower value (159 GgP/yr compared to the current 378-485 GgP/yr).

Figure 77. Cropland N surplus in the current situation (S0) and the F2F (S3).



4.7.3 Conclusion: effect of measures

We have established a range of scenarios for the future of the agro-food system of Europe at the 2050 horizon. Some of them only involve adjustment of farming practices without structural change in the system (e.g., a 20% reduction of fertilizer use, scenario S2bis). Others consist of profound changes in cropping systems (e.g., crop production with organic crop rotations, scenario S1 and S1bis), in livestock production (e.g., crop and livestock farming reconnection, scenario S1 and S1bis), and/or in human diets (scenario S1). Whereas the S1/S1bis and S2/S2bis scenarios are relatively extreme developments in different directions, a last scenario combines elements of both directions (scenario S3). As demonstrated above, all scenarios meet European food demand (under different diets), but are associated to different levels of international trade, and to differing environmental impacts. Here, we compare the effects of these scenarios (Table 20), from two points of view: (1) the productive capacity of the European agro-food system, determining its ability to meet the domestic requirements and to export to the international market; and (2) the agricultural N emissions in multiple forms, determining several key environmental impacts of the system on soil, water and atmosphere.

4.7.3.1 Productive capacity and trade balance

For a few decades, Europe has been able to meet the food requirements of its population with a positive net balance of cereals, meat and milk. This is in part due to large imports of feed (Figure 78). This situation would not change as a result of population increase and redistribution (S2), and a 20% reduction of N fertilizers would only slightly affect this balance (S2bis). In contrast, the generalization of agro-ecological practices would completely change the situation as livestock feeding would be restricted to internal feed production and feed imports banished (S1, S1bis), resulting in a reduction of livestock numbers by about 35-45% compared to the current situation (Figure 78, Table 20). Europe would remain a net exporter of cereals (Figure 78), however depending on imports of meat and milk (S1bis) in the absence of a drastic change in the human diet with much less animal proteins. With such a change in diet (S1), otherwise recommended for public health and environmental reasons, Europe would become fully self-sufficient for food and feed and can even export considerable amounts of vegetal and animal food. The Farm to Fork (S3) scenario does not operate the lever of human diet change and involves a limited increase of organic agriculture to 25% of the agricultural area: in that case Europe would halve its capacity to export cereals and animal products and would continue to import feed in substantial amounts.

Table 20. Summary of the main results of the scenarios.

Scenarios	**2014-2019 ref.**	**S0 current**	**S2 BAU** current diet	**S2bis BAU** current diet, -20% fertiliser	**S1 AE** change in diet	**S1bis AE** current diet	**S3 F2F** current diet, -20% fertiliser
Population, M inhab	533	533	538	538	538	538	538
Human consumption, GgN/yr	3240	3240	3270	3270	2690	3279	3270
Vegetal	1350	1350	1355	1355	1880	1367	1355
Animal (excl. fish)	1680	1680	1704	1704	672	1660	1704
Livestock number, MLU	111	137	146	140	81	87	107
Import (+)/Export(-), GgN/y							
Vegetal food	-2530	-2167	-2460	-1906	-607	-1005	-1367
Livestock feed	+2960	+2960	+2960	+2960	0	0	2088
Animal products (food)	-557	-748	-844	-698	-445	494	-231
Farming practices and losses							
N synth. fertilizers, GgN/y	12270	12270	12270	9816	0	0	6957
Symbiotic N fixation, GgN/y	4090	4639	4411	4549	8018	8042	4769
Crop production(1), GgN/y	11280	11382	11482	10785	7265	7543	9085
Tot. gross N surplus(1), GgN/y	8260	7197	9390	7594	2719	2863	5885
NH3 volatilization, GgN/yr	3061	4026	3538	2457	1574	1696	2298
N_2O emission, GgN/y	366	392	384	355	170	169	284
median NO_3^- conc(2), mgN/l	16.5	17.7	17.0	15	4.9	5.5	12.7
P synth fertilizers, GgP/y	1230	1230	1230	985	0	0	700
Tot. P balance, GgP/y	378	485	477	333	-486	-443	159

(1) permanent grassland not included
(2) from arable cropland

Figure 78. Import/Export balance of Europe for vegetal food, meat and milk and feed in the different scenarios established.

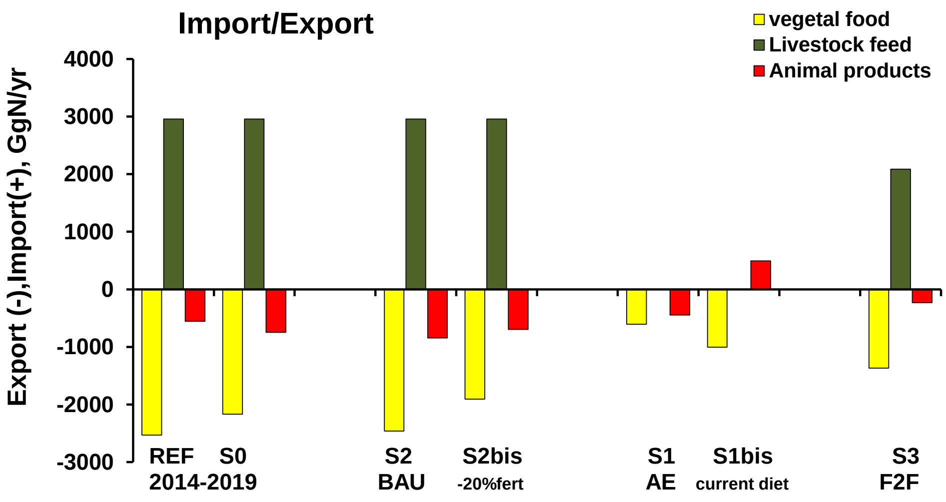
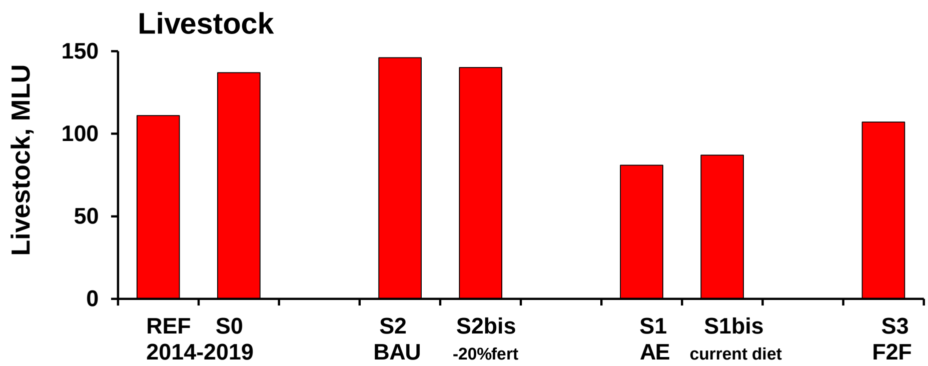


Figure 79. Livestock numbers in the different scenarios



4.7.3.2 Environmental impacts

Decreasing the intensity of agriculture is by far the most effective lever to reduce emissions of reactive N to the atmosphere and the hydrosphere. The agro-ecological scenario (S1) is the only one in which nitrous oxide emissions would be reduced by more than a factor 2 with respect to the current rate (Figure 80). Moreover, only in the agro-ecological scenarios (S1 and S1bis) would the median nitrate leaching concentration from arable cropland drop below the drinking water standard of 11 mgN/L (Figure 81).

Figure 80. Total N2O emissions by agriculture in the different scenarios.

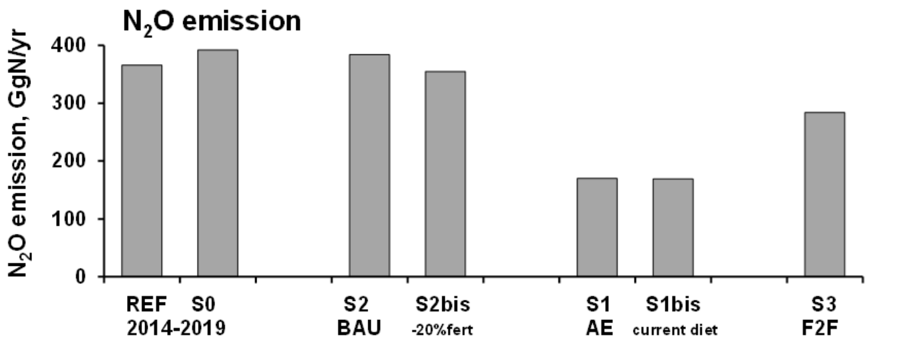
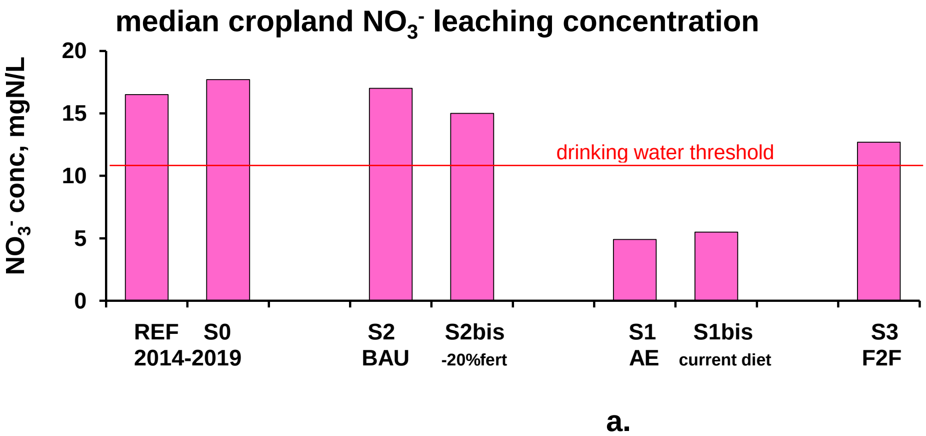
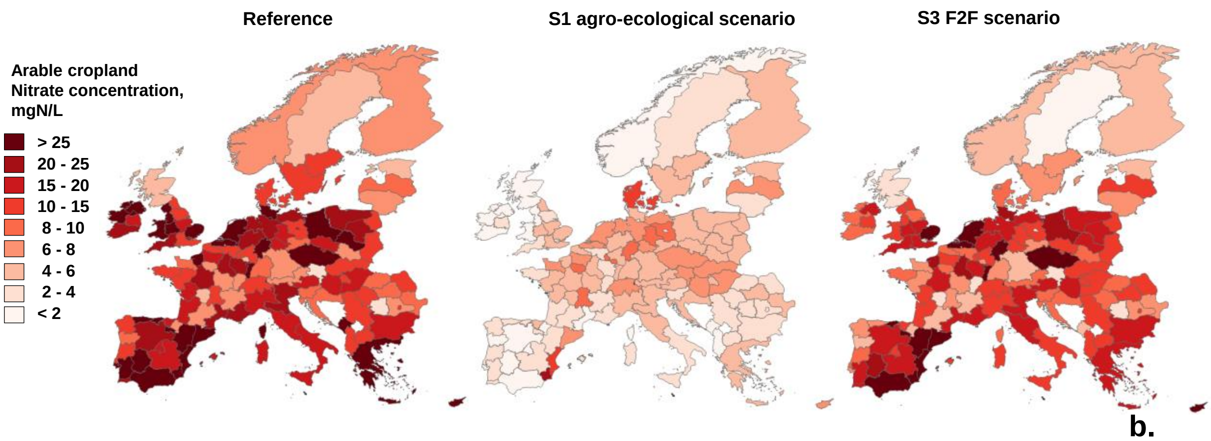


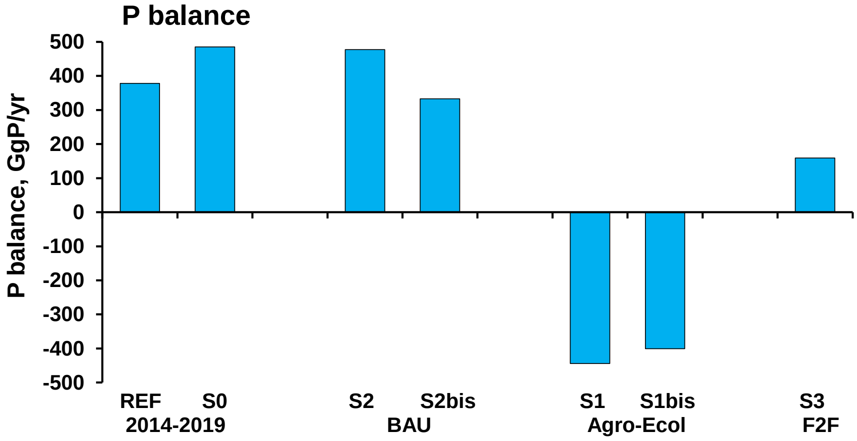
Figure 81. a. Median nitrate concentration in the leaching water from arable cropland in the different scenarios. b. Geographical distribution of nitrate concentration in arable cropland leaching water in the 2014-2019 reference, the S1 and S3 scenarios.





Finally, the effect of the different scenarios on the P soil balance is shown in Figure 82. The balance would remain positive, i.e., resulting in a net accumulation of P in the agricultural soils, in most scenarios, except in the agro-ecological ones. In the latter case, although the legacy from previous excess P fertilization is likely large enough to prevent P deficiencies before several decades, some kind of P fertilization would be needed in the long run.

Figure 82. P balance of European agricultural soils in the different scenarios.



To conclude, the measures advocated by the EC F2F and Biodiversity Strategies, however ambitious, seem insufficient, in view of the present study, to achieve the objective of halving environmental losses of nitrogen to the hydrosphere and the atmosphere. More structural changes, such as those implied in the agro-ecological scenario seem necessary. In order to reach food and feed self-sufficiency of Europe within the environmental constraints set out by the EC, the necessary structural changes would involve transitions at the production level in combination with a transition in the current dietary patterns.

5 Key findings and conclusions

In the Biodiversity Strategy to 2030 (BDS), the Farm to Fork Strategy (F2F) and the Zero Pollution Action Plan (ZPAP) the European Union has set an ambitious and ground-breaking goal to **reduce by 50% nutrient losses** to the environment (air, water, soil) by 2030, while preserving soil fertility. 'This will be achieved by implementing and enforcing the relevant environmental and climate legislation in full, identifying with Member States the nutrient load reductions needed to achieve these goals, applying balanced fertilisation and sustainable nutrient management, and by managing nitrogen and phosphorus better throughout their lifecycle. To this end, the Commission will work with Member States to develop an **Integrated Nutrient Management Action Plan** in 2022' (INMAP).

The '**Knowledge for INMAP**' project, developed by the JRC during the year 2021, aimed to gather scientific knowledge and data available in the EU to support the discussion and preparation of the INMAP. The work focused on three major tasks: 1) the description of the current **flows** of nitrogen (N) and phosphorus (P) in the EU considering all sources and sectors involved (agriculture, industries, urban, food production-consumption, energy and transport) and all environmental losses to air, water, and soil (Chapter 2); 2) the evaluation of the distance to environmental **targets**, considering the EU legislation and strategies (Chapter 3); 3) the analysis of **measures** to reduce nutrient pollution at different intervention points in the nutrient cycle (Chapter 4). In addition, the project developed online **map viewers and dashboards** that allow to browse nutrient fluxes per countries and sources, explore nutrient emissions to the environment identifying regional hotspots of pollution, and examine the regional effects of possible measures to reduce nutrient losses (https://water.jrc.ec.europa.eu).

Many **data sources** were used and combined for the analysis of N and P stocks and flows in the different sectors and environmental compartments. Similarly, for the scenarios analysis different **modelling tools** were considered that are based on specific datasets and assumptions. Therefore, the study cannot ensure a complete coherence of all the datasets adopted. It focused on gathering relevant scientific knowledge available in Europe for the preparation of the INMAP, acknowledging that **uncertainty** in flows estimations is part of the complexity of the N and P cycle analysis.

5.1 FLOWS - How much are current nutrient fluxes in the EU?

Knowledge on the nutrient fluxes between environmental compartments is key for understanding the level of disruption of the natural N and P cycles and for planning measures to reduce nutrient pollution while preserving soil fertility (Chapter 2).

Current **fluxes in N and P cycles** influenced by anthropogenic activities were estimated for EU (Figures 83 and 84) and per country. The annual new input of nitrogen to land is 12 TgN/y leading to around 8 TgN/y emissions to air and 5 TgN/y losses to freshwater. The annual new input of phosphorus is estimated 1 TgP/y, one third of which is lost by soil erosion and emissions to waters. Large part of N and P losses to freshwater ultimately reaches the sea. The amount of nutrients that are applied annually in the agricultural system as manure, 6 TgN/y and 2 TgP/y, shows the prominent role of the livestock sector in the environmental impacts. The comparison of agricultural fluxes according to different data sources and modelling assessments shows a lower **variability** in estimating mineral fertilizer input (CV[28]<10%) than manure application (CV around 20-30%) and N biological fixation (CV almost 70%). Major **knowledge gaps** in the quantification of N and P cycles concern the legacy and buildup of N in groundwater and of P in soil (Section 2.1).

According to the material flow analysis of nutrients in the **EU food system** (year 2015), about 50% of N and 40% of P from annual agricultural production entering the food processing system end up in waste (3.7 TgN/y and 0.4 TgP/y, Figures 83-84), of which food waste represents 13% and 10% for N and P, respectively (Section 2.2).

Information on N and P losses to the environment were also calculated at different spatial resolution, depending on data availability, including administrative (country, NUTS2) and hydrological units (river basin), with the intention to show regional impacts and **pollution hotspots**, and support several planning levels (Section 2.3). Web maps and dashboard applications were developed to browse interactively information on nutrient fluxes and sources contribution (https://water.jrc.ec.europa.eu Integrated nutrient management page) (Section 2.3).

[28] Coefficient of variation (CV)

Figure 83. Major nitrogen fluxes in EU27 (TgN/y) across air, land and water compartments (data sources and values from Table 1 and Table 2, riverine load to sea from Section 4.6). EU27 as in January 2021, values refer to 2015 or closest year, only major fluxes influenced by antrophogenic activities are depicted, therefore the overall budget is not closed as reported here. Manure is represented as an internal flux within land. *Net import of food 0.09 TgN/y.

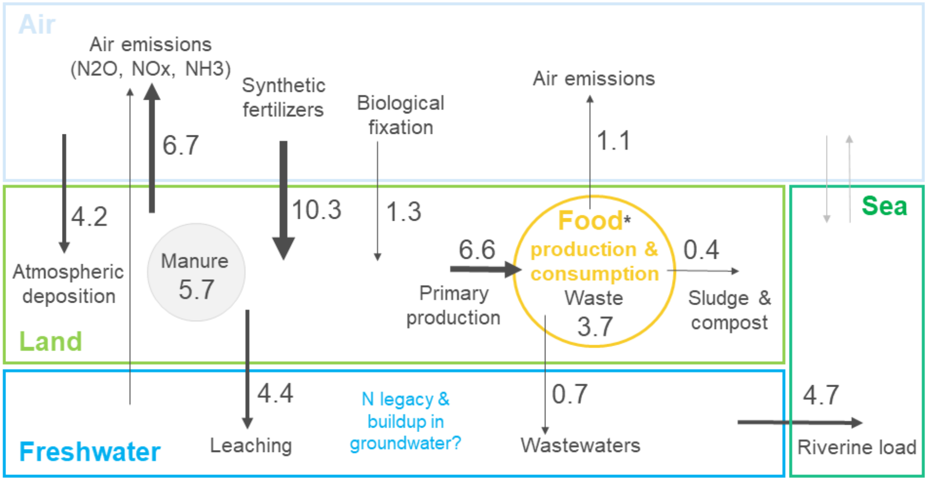
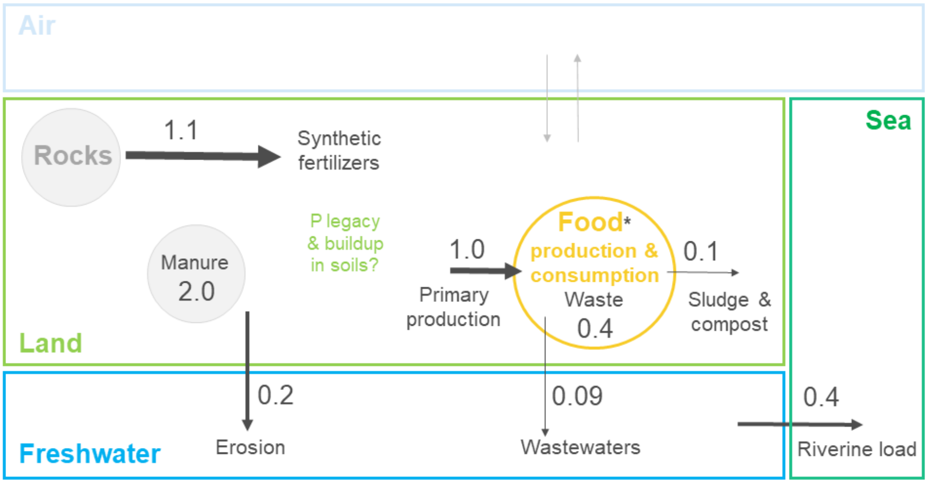


Figure 84. Major phosphorus fluxes in EU27 (TgP/y) across air, land and water compartments (data sources and values from Table 1 and Table 3, riverine load to sea from Section 4.6). EU27 as in January 2021, values refer to 2015 or closest year, only major fluxes influenced by antrophogenic activities are depicted, therefore the overall budget is not closed as reported here. Manure is represented as an internal flux within land. *Net import of food 0.01 TgP/y.



5.2 TARGETS - How much should the EU reduce nutrient fluxes?

The target of reducing nutrient losses to the environment of 50% by 2030, while preserving soil fertility, set in the EU Green Deal strategies, is an overarching goal that will be achieved by the implementation and enforcing of environmental and climate EU legislation and new actions. The study examined the current EU legislation dealing with nutrient emissions to the environment and nutrient management and recycling in waste, highlighting existing environmental goals (and where possible showing the distance to the policy targets; Figure 85). The EU target of halving nutrient losses was also considered in the perspective of planetary boundaries (Chapter 3, Figure 86 and 87).

The **National Emission reduction Commitments Directive** (NECD, 2016/2284/EU) regulates the concentration of pollutants in the air to achieve levels of air quality that do not produce significant negative impacts (acidification, eutrophication and ground-level ozone pollution) and risks to human health and the environment. NECD sets emission reduction commitments for the periods 2020 – 2029 as well as 2030 and beyond for five main air pollutants (NOx, NMVOCs, SO2, NH3 and PM2.5). The

Industrial Emissions Directive (IED, 2010/75/EU) is the main EU instrument regulating pollutant emissions to air, water and land (including NH_3, NO_X, N_2O, total N, total P) from industrial installations (listed in Annex I). The IED regulates the emissions through the establishment of sector-specific Best Available Techniques REFerence documents (BREFs) containing information about the sector and the latest emission control techniques used (Section 3.1).

The **Water Framework Directive** (WFD, 2000/60/EC) sets the environmental objective of achieving good chemical and ecological status for all water bodies: rivers, lakes, groundwater, transitional and coastal waters (by 2027). As per the WFD, Member States analyse the environmental impact of human activities on waters and develop River Basin Management Plans (RBMPs) every 6 years, including a Programme of Measures to achieve the environmental objective of good status. The measures include among others the implementation of the EU legislation for the protection of water from nutrient pollution from point sources, **Urban Waste Water Treatment Directive** (UWWTD, 91/271/EEC) and IED, and diffuse agricultural sources, **Nitrates Directive** (ND, 91/676/EEC, concerning the protection of waters against pollution caused by nitrates from agricultural sources). Measures include also the protection of water bodies for abstraction of drinking water, to avoid their deterioration and reduce the level of purification treatment, **Drinking Water Directive** (Directive EU 2020/2184). According to the second RBMPs, 26% of surface water bodies and 17% of ground water bodies area reported impact of nutrient pollution. Regarding the distance to the WFD environmental targets: 74% of the groundwater bodies are in good chemical status, and 40% good ecological status or potential. The compliance rates for the UWWTD are 88% for secondary treatment of waste water and 86% for more stringent removal of phosphorus and nitrogen. The last implementation report of the ND (reporting period 2016-2019) indicates that 14.1% of groundwater stations exceeded the environmental target of annual average 50 mg nitrates/L, and 81% of marine waters and at least one third of rivers, lakes, transitional and coastal waters are reported as eutrophic. The **Marine Strategy Framework Directive** (MSFD, Directive 2008/56/EC) establishes a framework for achieving good environmental status (GES) in the marine environment. The GES of marine waters is characterised by 11 qualitative descriptors. Most of them are influenced by nutrient pollution, including biodiversity, presence of non-indigenous species, fish population, reproduction, eutrophication and sea floor integrity (Descriptors 1-6). The last implementation report of the MSFD indicates that eutrophication and nutrient conditions are a problem in large part of coastal waters in the Baltic Sea, in southern North Sea, along the north-western coast of France and close to river outflows in the Mediterranean Sea, and that phytoplankton conditions pose a problem in the Black Sea (Section 3.2).

Agriculture is a major driver of nutrient pollution of air, soil and water. The new **Common Agricultural Policy** (CAP, Regulation EU 2021/2116) focuses on ten key objectives, linked to common EU goals for social, environmental, and economic sustainability in agriculture and rural areas. Three out of the ten objectives concern the environment and climate, i.e. contribute to climate change mitigation and adaptation, foster sustainable development and efficient management of natural resources (water, soil and air), and contribute to the protection of biodiversity. Member States will draw up a "CAP strategic plan", analysing the situation on their territory in respect of the ten key objectives. The strategic plan will set quantified targets against the objectives and design actions for achieving them, on the basis of an EU-level menu. Year-by-year progress against the targets will be monitored and the plan will be adjusted as necessary. In the plan, Member States will have to show how, in pursuing the CAP's objectives, they will also make a contribution to achieving the objectives of various EU environmental and climate legislation (on biodiversity, water and air quality, greenhouse gas emissions, energy and pesticides) and of the Green Deal, including the reduction of 50% of the nutrient losses by 2030 (Section 3.3).

Nutrients in waste streams are mainly present in sludges from municipal and industrial waste water treatment plants and municipal bio-waste. Also suboptimal management of animal by-products, including (excess) manure, can be an important source of nutrient dissipation. In addition, NH_3 and NO_x emissions can be trapped from N-rich off-gases from specific facilities and industries (e.g. livestock stables, incineration plants) and end up as residues that are disposed of. Waste management systems can help to achieve a circular economy and ensure that waste materials containing nutrients can safely re-enter the biosphere. The **Waste Framework Directive** (2008/98/EC) promotes the prevention of waste, and regulates the collection and management of food and animal waste as well as other nutrient-rich waste to promote safe nutrient re-use and recycling. The EU has set the target of re-use and recycling of municipal waste to at least by 60% by 2030, and required the diversion of biodegradable municipal waste from landfills (Landfill Directive, 1999/31/EC). It has also established the reuse of sludge arising from waste water treatment whenever appropriate (UWWTD, and Sludge Directive 86/278/EEC), and the conditions for the incineration of waste (IED). In addition, the EU committed to the goal of halving the per capita food waste by 2030 at retail and consumer levels (F2F and SDG Target 12.3) (Section 3.4).

The EU legislation addressing nutrient losses to the environment or their impacts on ecosystems has established indicators to **monitor progress** towards the policy objectives (Table 11). Data on these indicators are collected by Member States but are reported with spatial and temporal resolutions that depend on the respective policy and reporting cycle (Section 3.5).

The concept of **planetary boundaries** has been introduced by Rockstorm et al. (2009) to illustrate the impact of human activities on the Earth System functioning. The planetary boundaries framework proposes levels of anthropogenic perturbations below which the risk of generalized ecosystem destabilisation remains low (Steffen et al. 2015). Rescaling the global planetary boundary of Steffen et al. (2015) for EU indicates a boundary of 4.4 TgN/y of N mineral fertiliser and intentional biological fixation and 0.4 TgP/y of P fertiliser input. Similar values are obtained when upscaling the nutrient per capita planetary boundary of O'Neill et al. (2018) for EU (4.0 TgN/y and 0.4 TgP/y). According to these boundaries and the values of new input of N and P estimated in this study (Section 2.1), the EU should reduce its annual mineral fertiliser input of about 60% (Figures 86 and 87). Planetary and regional boundaries are useful concepts to understand whether the current N and P flows in EU are within the 'safe operating space', which is not the case, and how much possible intervention measures to reduce nutrient losses can help achieving this goal (Section 3.6).

Figure 85. Distance to targets of EU legislation (references in the text).

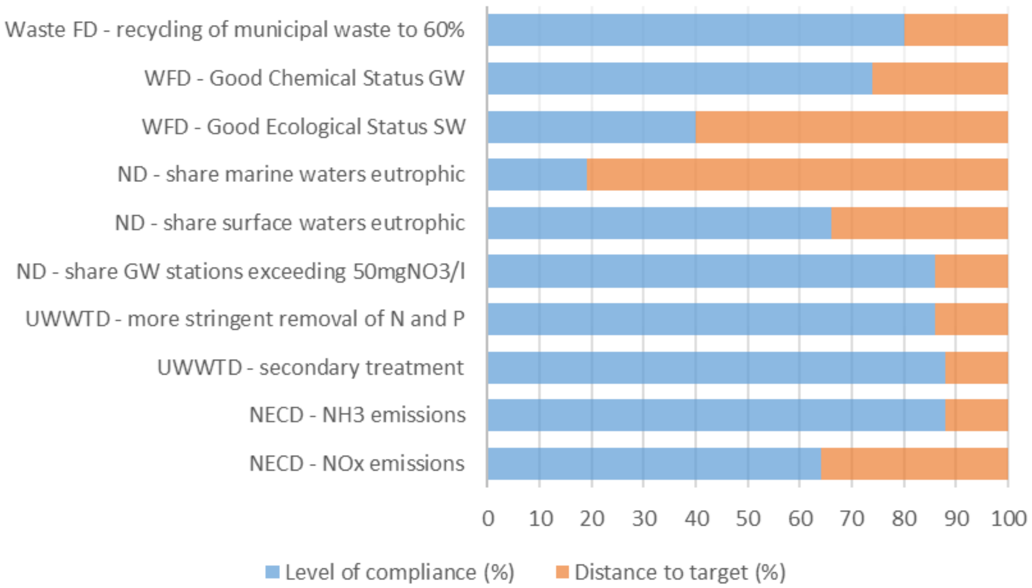


Figure 86. Major nitrogen fluxes in EU27 (TgN/y) across air, land and water compartments (data sources and values from Table 1 and Table 2, riverine load to sea from Section 4.6) concerned by EU legislation, the Biodiversity Strategy (BDS) target and planetary boundaries. EU27 as in January 2021, values refer to 2015 or closest year, only major fluxes influenced by antrophogenic activities are depicted.

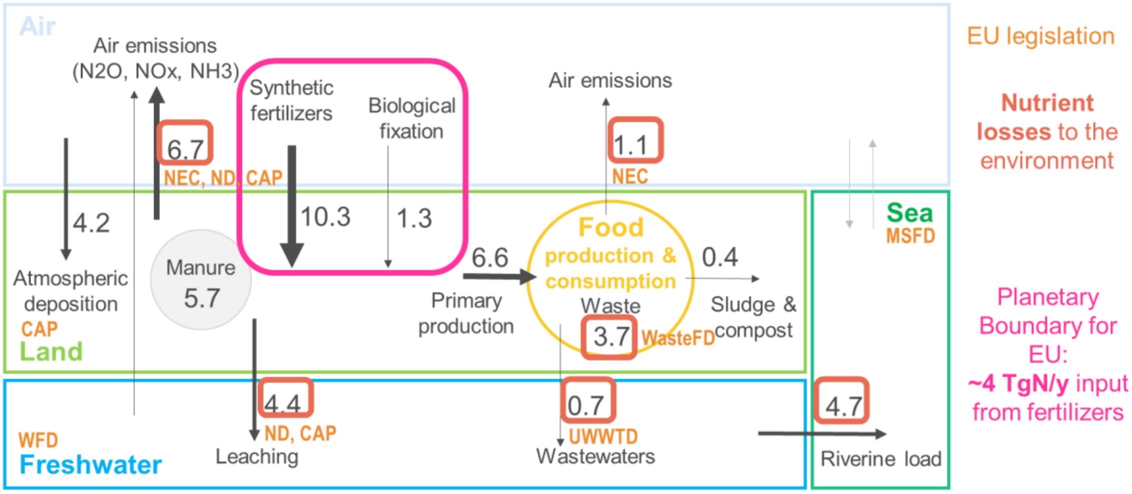
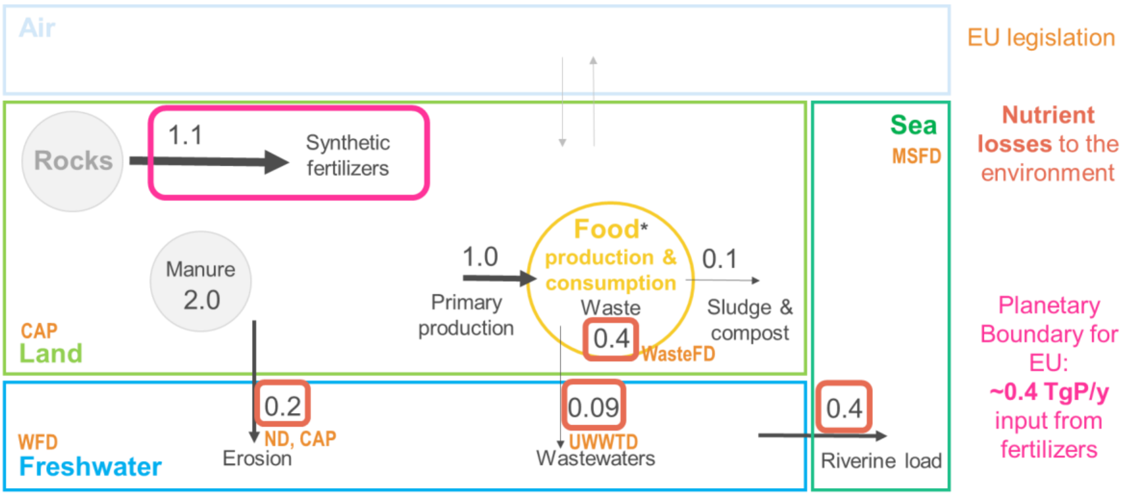


Figure 87. Major phosphorus fluxes in EU27 (TgP/y) across air, land and water compartments (data sources and values from Table 1 and Table 3, riverine load to sea from Section 4.6) concerned by EU legislation, the Biodiversity Strategy (BDS) target and planetary boundaries. EU27 as in January 2021, values refer to 2015 or closest year, only major fluxes influenced by antrophogenic activities are depicted.



5.3 MEASURES - How much measures could reduce nutrient fluxes in the EU?

Measures to reduce nutrient losses to the environment can be adopted at different intervention point of the N and P cycles. They range from technical measures for recovering and recycling nutrients in waste streams and improving nutrient use efficiency in agriculture, to policy measures at the EU level and to broad societal changes, such as changes in the human diet and the agricultural system (food production-consumption system). This study presents a (non-exhaustive) analysis of possible measures, considering both evidence from the literature and results of new modelling assessments. The latter were carried out adopting different modelling approaches, with an ensemble modelling perspective rather than a full integration, i.e. gathering evidence from different modelling assessments based on independent assumptions (Chapter 4).

The analysis of techniques for **nutrient recovery from manure, sewage sludge and bio-waste** indicates that novel recycling techniques can capture and transform N and P from organic waste into nutrient-dense concentrated and safe (mineral) fertilisers, and may enable to transfer nutrients from nutrient-excess to nutrient-demanding EU regions. The maximum potential of such actions is to substitute about 10% and 25% of the N and P mineral fertilisers, respectively. In addition, increased efforts to collect and re-use current discarded biogenic materials may contribute marginally to make available supplementary amounts of organic fertilisers (Section 4.1).

A systematic analysis of the impacts of **agricultural farming practices** on environment and climate has been developed by the Commission in the project IMAP (https://webgate.ec.europa.eu/fpfis/wikis/display/IMAP/Home currently restricted access), based on published scientific meta-analysis. For each farming practice qualitative (positive, negative, no effects, uncertain) impacts on environmental and climate issues are provided, including N2O and NH3 emissions, N leaching/run-off, N plant nutrient uptake, N use efficiency and soil N content. In addition, when available, quantitative data extracted from the reviewed meta-analyses are provided (Section 4.2).

Several **modelling assessments** were carried out to estimate the effects of measures (scenarios of measures) at the EU level. Reduction of major nutrient flows, according to the different assessments, are presented in Figures 88 and 89 for N and P, respectively. Modelling tools include EMEP, DayCent, Model P, GREEN and GRAFS.

The effect of air emissions reduction on **N atmospheric deposition** were analysed by the model EMEP. The scenarios considered the measures to reduce air emissions adopted in the Commission FitFor55 package (SO2 -57.7%, PM2.5 -54.5% and NOx -40.3%) and in the National Emission Reduction Commitments (NEC Directive, NH3 -10% and NMVOCs -24%). The total N atmospheric deposition on land estimated by the modelling in this scenario shows a decrease of 69%, compared to the values of 2015. The reductions vary from country to country (Section 4.3).

A scenario of **balanced mineral N fertilization** was tested by the model DayCent, considering a reduction of mineral N fertiliser by 20% in agricultural areas with high N surplus (N surplus > 50 kgN/ha), and an increase of mineral N fertiliser by 20 % in agricultural mining areas (N surplus < 50 kgN/ha). The scenario analysis indicates that a more balanced mineral N fertilization might allow a 7% saving of mineral N fertiliser compared to the current application. The expected reduction of N leaching is about 6% (compared to current losses). Central Europe plus Ireland show higher benefit, while very marginal effects are predicted in Eastern countries. The N2O emissions patterns follow those of change in mineral N fertilization, with a total decrease of 4%. Overall, the scenario foresees a slightly decrease of soil organic content (SOC -14 Mt C, which represents 0.1% of the current topsoil SOC stock) with regional increase in areas currently affected by N mining and decrease in surplus area. Small SOC deficit can be recovered by appropriate best practices (cover crop, residue management, agroforestry etc.) (Section 4.4).

A new assessment of **P budget and erosion** in European agricultural soils was performed (P model). Summing up the inputs from inorganic fertilizers, the manure input and atmospheric deposition, the mean annual input of P in soils was estimated at 15 kgP/ha/y (EU+UK). In most of the North-western European regions, the rates of P removal are higher than 20 kgP/ha/y, while rates are lower than 10 kgP/ha/y in Mediterranean regions and South-East EU countries. The P lost to river-basins from agricultural lands due to soil erosion is about 60,000 tP/y. P budget is still a surplus for the whole EU and UK area with about 0.7 kgP/ha/y. However, there is a high spatial variation of P budget across countries and regions and reduction of P should be focused region specific. Coupling soil erosion with P stock allows to estimate the total P displacement due to water erosion. Only 15-20% of displaced P ends to the river basins and furthermore to the sea outlets (Section 4.5).

The effect of different nutrient reduction measures on **N and P losses to freshwater and sea** were analysed with the model GREEN. The scenario analysis included measures to reduce: 1) nutrient discharges from domestic wastewaters (according to the Impact Assessment for the revision of the UWWTD); 2) nutrient emissions from agricultural sources (scenarios based on the CAPRI model, considering the new CAP, F2F and BDS targets and New Generation EU Funds); 3) nitrogen input from atmospheric deposition (developed by the model EMEP, including the measures adopted in the FitFor55 package, Section 4.3); and 4) a scenarios (INMAX) combining all measures together (i.e. reducing domestic emissions, agricultural sources and atmospheric deposition). Improvement of domestic wastewaters treatment could decrease the nutrient export to the European seas by 8% for N and 13% for P. Reduction of N atmospheric emissions could lower the N export to the sea by 11%. Measures under the new CAP and to achieve BDS and F2F strategy targets could lead to a decrease of N and P load to the seas of 13% and 3%, respectively. Adopting all the measures together (INMAX scenario) could reduce the nutrients load to the European seas by 32% for N and 17% for P. The regional effect of measures on the N:P ratio in the aquatic environment needs to be considered for the potential impact on coastal and marine ecosystems (Section 4.6).

Several scenarios for the **future of the agro-food system** of Europe at the 2050 horizon were analysed by the model GRAFS, including the scenarios: Agro-Ecological (S1 AE), Business As Usual (S2 BAU) and Farm to Fork (S3 F2F). All scenarios meet European food demand (under different diets), but are associated to different levels of international trade, and to differing environmental impacts. A 20% reduction of N fertilizers would only slightly affect the current condition. In contrast, the generalization of agro-ecological practices would completely change the situation as livestock feeding would be restricted to internal feed production and feed imports banished (S1), resulting in a reduction of livestock numbers by about 35-45% compared to the current situation. Europe would remain a net exporter of cereals, however depending on imports of meat and milk (S1bis) in the absence of a drastic change in the human diet with less animal proteins. With such a change in diet (S1), otherwise recommended for public health and environmental reasons, Europe would become fully self-sufficient for food and feed and could even export considerable amounts of vegetal and animal food. The Farm to Fork scenario (S3) does not operate the lever of human diet change and involves a limited increase of organic agriculture to 25% of the agricultural area: in that case Europe would halve its capacity to export cereals and animal products and would continue to import feed in substantial amounts. Decreasing the intensity of agriculture is by far the most effective lever to reduce emissions of reactive N to the atmosphere and the hydrosphere. The agro-ecological scenario (S1) is the only one in which nitrous oxide emissions would be reduced by more than a factor 2 with respect to the current rate. Moreover, only in the agro-ecological scenarios (S1 and S1bis) would the median nitrate leaching concentration from arable cropland drop below the drinking water standard of 11 mgN/L. The P balance would remain positive, i.e., resulting in a net accumulation of P in the agricultural soils, in most scenarios, except in the agro-ecological ones. In the latter case, although the legacy from previous excess P fertilization is likely large enough to prevent P deficiencies before several decades, some kind of P fertilization would be needed in the long run (Section 4.7).

Current measures for cutting air emissions under the FitFor55 and NECD will substantially reduce NOx (-40%) and NH3 (-10%) with a consequent reduction of N atmospheric deposition, but further reduction of NH3 and N2O emissions will strongly depend on the measures adopted in agriculture under the new CAP. Balanced mineral fertilization alone would lead only to limited reduction of N losses air (N2O emissions -4%) and water (NO3 leaching -6%). Improvement of domestic wastewaters treatment, reduction of N atmospheric emissions and measures under the new CAP and to achieve BDS and F2F strategy targets could reduce the nutrients load to the European seas by 32% for N and 17% for P. These reductions are substantial but still below the BDS target of -50% losses, although this target is intended for the initial nutrient losses to freshwater, such as nutrient leaching to groundwater and runoff to surface water. The scenario analysis of the agro-food system (GRAFS model) concluded that the measures of F2F, however ambitious, seem insufficient, to achieve the objective of halving environmental losses of nitrogen to the hydrosphere and the atmosphere, and that in order to reach food and feed self-sufficiency of Europe within the environmental constraints, the necessary structural changes would involve transitions at the production level in combination with a transition in the current dietary patterns.

The results of this study are preliminary and not exhaustive, additional measures can be tested. What emerges is that a combination of measures and societal changes addressing different fluxes in the nutrient cycles will be necessary to achieve the BDS target, and impacts on all environmental compartments and feedbacks should be considered. The uncertainty in data and modelling assumptions also highlights the added value of adopting several modelling tools and approaches. Finally, regional variability might offer specific opportunities for nutrient reduction.

Figure 88. Reduction of major nitrogen fluxes in EU* across air, land and water compartments estimated by several scenarios modelling assessments in the study (Chapter 4). Values in TgN/y, reductions in %. *Fluxes and % of change refer to the extent and time period of reference of each specific assessment (EMEP Section 4.1, DayCent Section 4.2, GREEN and CAPRI Section 4.6, and GRAFS Section 4.7)

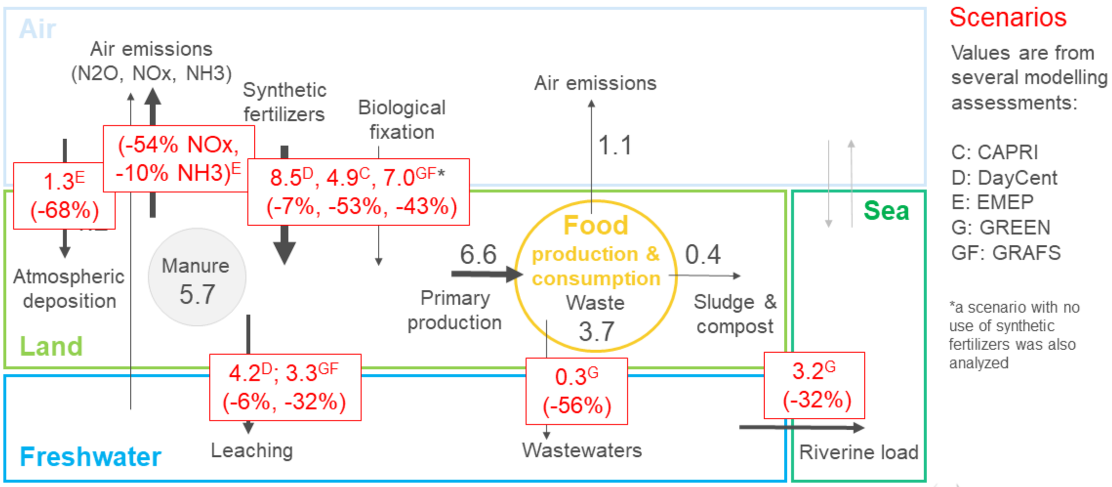
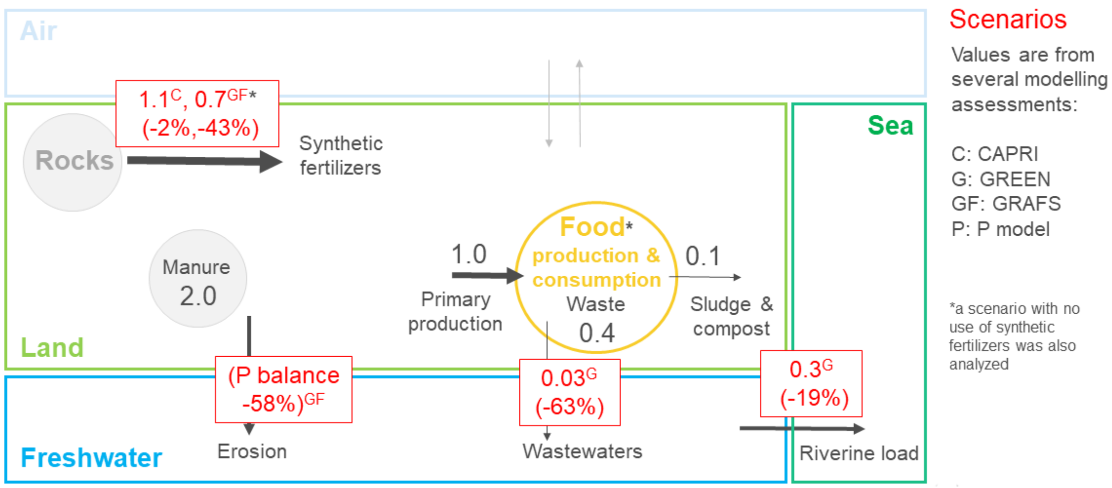


Figure 89. Reduction of major phosphorus fluxes in EU* across air, land and water compartments estimated by several scenarios modelling assessments in the study (Chapter 4). Values in TgP/y, reductions in %. *Fluxes and % of change refer to the extent and time period of reference of each specific assessment (GREEN and CAPRI Section 4.6, and GRAFS Section 4.7)



References

3M, Capturing Ammonia in Flue Gas Condensate Treatment for Biomass Power Stations with Liqui-Cel ® Membrane Contactors. Fact-sheet, 2021.

Abu-Orf, M., and Goss, T, Comparing Thermal Hydrolysis Processes (CAMBITM and EXELYSTM) For Solids Pretreatmet Prior To Anaerobic Digestion. *Proceedings of the Water Environment Federation*, 2, 2012, 1024–1036. https://doi.org/10.2175/193864712811693272

Adam, C., Peplinski, B., Michaelis, M., Kley, G., and Simon, F. G., Thermochemical treatment of sewage sludge ashes for phosphorus recovery. *Waste Management*, 29(3), 2009, 1122–1128. https://doi.org/10.1016/j.wasman.2008.09.011

Adamczyk, Z., Cempa, M., and Białecka, B., Phosphorus-rich ash from poultry manure combustion in a fluidized bed reactor. *Minerals*, 11, 2021, https://doi.org/10.3390/min11070785

Agar, D. A., Kwapinska, M., and Leahy, J. J., Pyrolysis of wastewater sludge and composted organic fines from municipal solid waste: laboratory reactor characterisation and product distribution. *Environmental Science and Pollution Research*, 25(36), 2018, 35874–35882. https://doi.org/10.1007/s11356-018-1463-y

Alewell, C., Ringeval, B., Ballabio, C., Robinson, D.A., Panagos, P., and Borrelli, P., Global phosphorus shortage will be aggravated by soil erosion. *Nat Commun*, 11, 2020, 4546. https://doi.org/10.1038/s41467-020-18326-7

Amann, A., Zoboli, O., Krampe, J., Rechberger, H., Zessner, M., and Egle, L., Environmental impacts of phosphorus recovery from municipal wastewater. *Resources, Conservation and Recycling*, 130, 2018, 127–139. https://doi.org/10.1016/j.resconrec.2017.11.002

AMEC, *Collection and Analysis of Data for the Control of Emissions from the Spreading of Manure* - Final Report for European Commission (DG Environment), 2014, https://ec.europa.eu/environment/air/pdf/Final Report.pdf

Andersen, J.K., Boldrin, A., Christensen, T.H., and Scheutz, C., Mass balances and life cycle inventory of home composting of organic waste. *Waste Manag*., 31 (9–10), 2011, 1934–1942.https://doi.org/10.1016/j.wasman.2011.05.004.

Anglade, J., Billen, G., Garnier, J., Makridis, T., Puech, T., and Tittel, C., Nitrogen soil surface balance of organic vs conventional cash crop farming in the Seine watershed. *Agricultural Systems*., 139, 2015a, 82–92

Anglade, J., Billen, G., Garnier, J., Relationships for estimating N2 fixation in legumes: incidence for N balance of legume-based cropping systems in Europe. *Ecosphere*, 6, 2015b, 37, 1-24.

Araújo, R, Somma, F, Aigars, J, Axe, P, Bartolo, A, De Cauwer, K, Devreker, D, Fleming-Lehtinen, V, García-Denche, LM, Knudsen-Leerbeck, H, Kraśniewski, W, Lazar, L, Lefebvre, A, Lips, I, Magaletti, E, Moncheva, S, PagouK, Precali, R, Ruiter, H, Wilkes, R. Eutrophication in marine waters: harmonization of MSFD methodological standards at EU level, EUR 29854 EN, Publications Office of the European Union, 2019, ISBN 978-92-76-11224-2, doi:10.2760/437291, JRC117109

Araujo, R., Boschetti, S.T., Marine Strategy Framework Directive, Review and analysis of EU Member States' 2018 reports, Descriptor 5: Eutrophication, Assessment (Art.8) and Good Environmental Status (Art. 9), EUR 30677 EN, Publications Office of the European Union, Luxemburg, 2021, ISBN 978-92-76-36246-3, doi:10.2760/180642, JRC124915.

Asman, W.A.H., et al. 2011. A simplified model of nitrogen flows from manure management IIASA Interim Report IR-11-030, International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria.

AV Braunschweig, Projekt KlärWert - Nährstoffrückgewinnung und thermische Desintegration. Presentation, 2019.

Ballabio, C., Lugato, E., Fernández-Ugalde, O., Orgiazzi, A., Jones, A., Borrelli, P., Montanarella, L. and Panagos, P., Mapping LUCAS topsoil chemical properties at European scale using Gaussian process regression. *Geoderma*, 355, 2019, 113912.

Ballabio, C., Panagos, P., and Monatanarella, L., Mapping topsoil physical properties at European scale using the LUCAS database. *Geoderma*, 261, 2016, 110–123.

Barreiro-Hurle, J., Bogonos, M., Himics, M., Hristov, J., Pérez-Domiguez, I., Sahoo, A., Salputra, G., Weiss, F., Baldoni, E., Elleby, C. Modelling environmental and climate ambition in the agricultural sector with the CAPRI model. Exploring the potential effects of selected Farm to Fork and Biodiversity strategies targets in the framework of the 2030 Climate targets and the post 2020 Common Agricultural Policy, EUR 30317 EN, Publications Office of the European Union, Luxembourg, 2021, ISBN 978-92-76-20889-1, doi:10.2760/98160, JRC121368.

Barth, J., *Marketing Strategies and Quality Requirements in Compost Applications*. ISWA Beacon Conference, BTW, 2006, Perugia. Available at: https://www.iswa.org/uploads/tx_iswaknowledgebase/btw26.pdf

Battista, F., Masala, C., Zamboni, A., Varanini, Z., and Bolzonella, D., Valorisation of Agricultural Digestate for the Ammonium Sulfate Recovery and Soil Improvers Production. *Waste and Biomass Valorization*, 12(12), 2021, 6903–6916. https://doi.org/10.1007/s12649-021-01486-y

Beckinghausen, A., Odlare, M., Thorin, E., and Schwede, S., From removal to recovery: An evaluation of nitrogen recovery techniques from wastewater. *Applied Energy*, 263, 2020, 114616, https://doi.org/10.1016/j.apenergy.2020.114616

Biavetti, I., Karetsos, S., Ceglar, A., et al., European meteorological data: contribution to research, development, and policy support. In: *Second international conference on remote sensing and geoinformation of the environment (RSCy2014)*. International Society for Optics and Photonics, 2014, p 922907

Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, L., and Sanz-Cobena, A., Reshaping the European agro-food system and closing its nitrogen cycle: the potential of combining dietary change, agroecology, and circularity, *One Earth*, 4, 2021, 839-850

Billen, G., Lassaletta, L., Garnier, J., Le Noë, J., Aguilera, E., and Sanz-Cobena, A., Opening to distant markets or local reconnection of agro-food systems? Environmental consequences at regional and global scales. Section VI, chapter 2. In: *Sustainable farming systems » Diversity within and among agro-ecosystems. A key to reconcile contemporary agriculture and environment quality?* Eds: Lemaire, G., Carvalho, P., Kronberg, S., and Recous, S., Elsevier, 2019. 452pp + Index

Billen, G., Lasseletta, L., and Garnier, J., A biogeochemical view of the global agro-food system: nitrogen flows associated with protein production, consumption and trade. *Glob. Food Sec.*, 3, 2014, 209–219. https://doi.org/10.1016/j.gfs.2014.08.003i.

Billen, G., Le Noë, J., and Garnier, J., Two contrasted future scenarios for the French agro-food system. *Science of The Total Environment*, 637–638, 2018, 695-705, https://doi.org/10.1016/j.scitotenv.2018.05.043.

Billen, P., Costa, J., Van Caneghem, J., and Vandecasteele, C., Electricity from Poultry Manure: A CleaCaneghem, Vann Alternative to Direct Land Application. *4th International Workshop Advanced Cleaner Production*. Sao Paulo, 2013. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.660.7521&rep=rep1&type=pdf

Binkowski F, and Shankar, U., The Regional Particulate Matter Model .1. Model description and preliminary results, *J. Geophys. Res*., 100, 1995, 26191–26209.

Birk S., Wendy Bonne, Angel Borja, Sandra Brucet, Anne Courrat, Sandra Poikane, Angelo Solimini, Wouter van de Bund, Nikolaos Zampoukas, Daniel Hering. 2012. Three hundred ways to assess Europe's surface waters: An almost complete overview of biological methods to implement the Water Framework Directive. Ecological Indicators 18, 31–41.

Böhler, M. A., Rückgewinnung und Wiederverwendung von Stickstoff aus Abwasser. Eawag-Infotag. Abwasser Als Ressource – Zukunftsweisende Technologien Zur Rückgewinnung von Wertstoffen. Eawag-Infotag. 11. September 2018, 2018.

Bonomo, Trattamento Delle Acque Reflue. Mc-Graw Hill, 2008.

Borrelli, P., Paustian, K., Panagos, P., Jones, A., Schütt, B., and Lugato, E., Effect of Good Agricultural and Environmental Conditions on erosion and soil organic carbon balance: A national case study. *Land Use Policy*, 50, 2016, 408-421. DOI: 10.1016/j.landusepol.2015.09.033

Borrelli, P., Van Oost, K., Meusburger, K., Alewell, C., Lugato, E., and Panagos, P., A step towards a holistic assessment of soil degradation in Europe: coupling on-site erosion with sediment transfer and carbon fluxes. *Environ. Res*., 161, 2018, 291–298

Brink, C., et al. 2011. Costs and benefits of nitrogen in the environment, pp. 513 – 540. In: The European Nitrogen Assessment, ed. M.A. Sutton et al. Cambridge University Press 2011.

Buckwell, A., and Nadeu, E., Nutrient Recovery and Reuse (NRR) in European agriculture. A review of the issues, opportunities, and actions. In RISE Foundation, Brussels (Issue April), 2016. www.risefoundation.eu

Burek, P., Van Der Knijff, J., and De Roo, A., *LISFLOOD - Distributed Water Balance and Flood Simulation Model - Revised User Manual 2013*. EUR 26162. Luxembourg (Luxembourg): Publications Office of the European Union, 2013, JRC78917

Caldeira, C., De Laurentiis, V., Ghose, A., Corrado, S., and Sala, S. Grown and thrown: Exploring approaches to estimate food waste in EU countries. *Resources, Conservation and Recycling*, 168, 2021, 105426. https://doi.org/10.1016/j.resconrec.2021.105426

CE Delft, 2017. Handboek Milieuprijzen 2017. S.M. de Bruyn, S. Ahdour, M, Bijleveld, L. de Graaff, A. Schroten. Delft.

Celik, I., Ortas, I., and Kilic, S., Effects of compost, mycorrhiza, manure and fertilizer on some physical properties of a Chromoxerert soil. *Soil and Tillage Research*, 78, 2004, 59–67, https://doi.org/10.1016/j.still.2004.02.012

Chen, T. L., Chen, L. H., Lin, Y. J., Yu, C. P., Ma, H. Wen, and Chiang, P. C., Advanced ammonia nitrogen removal and recovery technology using electrokinetic and stripping process towards a sustainable nitrogen cycle: A review. *Journal of Cleaner Production*, 309, 2021, 127369. https://doi.org/10.1016/j.jclepro.2021.127369

Corrado, S., Caldeira, C., Carmona-Garcia, G., Körner, I., Leip, A., and Sala, S. Unveiling the potential for an efficient use of nitrogen along the food supply and consumption chain. *Global Food Security*, 25, 2020, 100368. https://doi.org/10.1016/j.gfs.2020.100368

Crippa, M., Oreggioni, G., Guizzardi, D., Muntean, M., Schaaf, E., Lo Vullo, E., Solazzo, E., Monforti-Ferrario, F., Olivier, J. and Vignati, E., *Fossil CO2 and GHG emissions of all world countries*, EUR 29849 EN, Publications Office of the European Union, Luxembourg, 2019. ISBN 978-92-76-11100-9 (online),978-92-76-11025-5 (print), doi:10.2760/687800

Crippa, M., Solazzo, E., Huang, G., Guizzardi, D., Koffi, E., Muntean, M., Schieberle, C., Friedrich, R., and Janssens-Maenhout, G., High resolution temporal profiles in the Emissions Database for Global Atmospheric Research. *Sci Data*, 7, 2020, 121.

De Graaff, L., Odegard, I., and Nusselder, S., *LCA of thermal conversion of poultry litter at BMC Moerdijk*. CE Delft, Delft, The Netherlands, 2017.

De Laurentiis, V., Caldeira, C., and Sala, S., Building a balancing system for food waste accounting at National Level, EUR 30685 EN, Publications Office of the European Union, Luxembourg, 2021, ISBN 978-92-76-37275-2, doi:10.2760/316306, JRC124446

De Vries, W., Schulte-Uebbing, L., Kros, H., Voogd, J.C., Louwagie, G., 2021. Spatially explicit boundaries for agricultural nitrogen inputs in the European Union to meet air and water quality targets. Sci. Total Environ. 786, 147283. https://doi.org/https://doi.org/10.1016/j.scitotenv.2021.147283

DG Agriculture and Rural Development, 2019. The post-2020 Common Agricultural Policy: Environmental benefits and simplification. https://ec.europa.eu/info/sites/default/files/food-farming-fisheries/key_policies/documents/cap-post-2020-environ-benefits-simplification_en.pdf

Diacono, M., and Montemurro, F., Long-term effects of organic amendments on soil fertility. A review. *Agronomy for Sustainable Development*, 30, 2010, 401–422. https://doi.org/10.1051/agro/2009040

DIONE - an integrated EO-based toolbox for modernising CAP area-based compliance checks and assessing respective environmental impact. D2.1: DIONE stakeholders, personas and co-designed scenarios, April 2020. https://dione-project.eu/wp-content/uploads/2020/10/DIONE_D2.1_V1.0_DIONE-stakeholder-list-personas-and-co-designed-scenarios.pdf

Djandja, O. S., Duan, P. G., Yin, L. X., Wang, Z. C., and Duo, J., A novel machine learning-based approach for prediction of nitrogen content in hydrochar from hydrothermal carbonization of sewage sludge. *Energy*, 232, 2021, 121010, https://doi.org/10.1016/j.energy.2021.121010

Doka, G., *Life Cycle Inventories of Waste Treatment Services*. Ecoinvent report No. 13, Swiss Centre for Life Cycle Inventories, Dübendorf, 2003.

Easymining. *Ash2Phos*, 2021, https://www.easymining.se/de/technologies/ash2phos/

ECE - Economic Commission for Europe. 2014. Guidance document on preventing and abating ammonia emissions from agricultural sources. ECE/EB.AIR/120

Egle, L., Rechberger, H., Krampe, J., and Zessner, M., Phosphorus recovery from municipal wastewater: An integrated comparative technological, environmental and economic assessment of P recovery technologies. *Science of the Total Environment*, 571, 2016, 522–542. https://doi.org/10.1016/j.scitotenv.2016.07.019

Ehlert, P., Agronomic Effectivity of Hydrated Poultry Litter Ash. In *Biorefinery of Inorganics: Recovering Mineral Nutrients from Biomass and Organic Waste*, 2020, https://doi.org/https://doi.org/10.1002/9781118921487.ch4-4

Ehlert, P., Sigurnjak, I., Meers, E., Verbeke, M., Adani, F., Zilio, M., Tambone, F., and Schoumans, O., *Nitrogen fertilising products based on manure and organic residues Supporting literature of the SYSTEMIC factsheets*. Wageningen, Wageningen Environmental Research, Report. 8, 2019, https://systemicproject.eu/wp-content/uploads/Factsheets-supporting-literature_nitrogen-fertilising-products.pdf

Einarsson, R., Pitulia, D., and Cederberg, C., Subnational nutrient budgets to monitor environmental risks in EU agriculture: Calculating phosphorus budgets for 243 EU28 regions using public data. *Nutrient Cycling in Agroecosystems*, 117, 2020, 199–213. https://doi.org/10.1007/s10705-020-10064-y

Einarsson, R., Sanz-Cobena, A., Aguilera, E., Billen, G., Garnier, J., van Grinsven, H.J.M., and Lassaletta, L., Crop production and nitrogen use in European cropland and grassland 1961–2019. *Scientific Data* (SDATA-21-00221) 8, 2021, 288 | https://doi.org/10.1038/s41597-021-01061-z

Elbl, J., Plošek, L., Kintl, A., Přichystalová, J., Záhora, J., and Friedel, J. K., The Effect of Increased Doses of Compost on Leaching of Mineral Nitrogen from Arable Land. *Polish Journal of Environmental Studies* 23, 2014, 697–703.

Ellersdorfer, M., Stocker, K., Lubensky, J., and Pesendorfer, S., *ReNOx2.0 - Rückgewinnung von Stickstoff und Phosphor aus Abwässern*, 2018, https://nachhaltigwirtschaften.at/resources/nw_pdf/events/20181207_stakeholderdialog/ellersdorfer_renox.pdf?m=1544113439&

Enders, A., Hanley, K., Whitman, T., Joseph, S., and Lehmann, J., Characterization of biochars to evaluate recalcitrance and agronomic performance. *Biores. Technol*., 114, 2012, 644–653.

Esculier, F., Le Noë, J., Barles, S., Billen, G., Créno, B., Garnier, J., Lesavre, J., Petit, L., Tabuchi, J.-P., The biogeochemical imprint of human metabolism in Paris Megacity: a regionalized analysis of a water-agro-food system. *J. Hydrol*. 2018, DOI 10.1016/j.jhydrol.2018.02.043

European Commission (EC), Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 Amending Directive 2008/98/EC on Waste, 2018

European Commission (EC), *Emissions Database for Global Atmospheric Research (EDGAR)*, 2021, https://data.europa.eu/doi/10.2904/JRC_DATASET_EDGAR

European Commission (EC), Expert Group for Technical Advice on Organic Production (EGTOP) Final Report on Organic Fertilizers And Soil Conditioners (II), 2016. http://ec.europa.eu/agriculture/organic/home_en

European Commission (EC), Green paper on the management of bio-waste in the European Union. SEC(2008) 2936, 2008

European Environment Agency (EEA), 2020a. European Environment Agency/Federal Office for the Environment FOEN. Is Europe living within the limits of our planet? An assessment of Europe's environmental footprints in relation to planetary boundaries. 2020a — 61 pp. ISBN 978-92-9480-216-3 doi:10.2800/537187

European Environment Agency (EEA), *Bio-waste in Europe — turning challenges into opportunities.* European Environment Agency, 2020b, https://www.eea.europa.eu/publications/bio-waste-in-europe

European Environment Agency (EEA), *Managing municipal solid waste - a review of achievements in 32 European countries*, European Environment Agency, 2013, https://doi.org/10.2800/71424

European Environment Agency (EEA), *Urban waste water treatment*. https://www.eea.europa.eu/data-and-maps/indicators/urban-waste-watertreatment/urban-waste-water-treatment-assessment4, 2019, Accessed in June 2021.

European Environment Agency (EEA), *National Emission reduction Commitments Directive reporting status 2021*. European Environment Agency Briefing no. 06/2021, 2021, https://www.eea.europa.eu/publications/national-emission-reduction-commitments-directive-2021

European Parliament and Council, (EC), Regulation (EC) No 178/2002 of the European Parliament and of the Council of 28 January 2002 laying down the general principles and requirements of food law, establishing the European Food Safety Authority and laying down procedures in matters of food safety, 2002

EUROSTAT, *Agri-environmental indicator - mineral fertiliser consumption*, 2020a, https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_mineral_fertiliser_consumption

EUROSTAT, *Annual crop statistics*, 2020b, https://ec. europa.eu/eurostat/databrowser/view/apro_cpsh1/default/table?lang=en. Accessed 15 July 2021

EUROSTAT, *Generation of waste by waste category, hazardousness and NACE Rev. 2 activity* (env_wasgen), 2021a, https://ec.europa.eu/ urostat/data/database Accessed in June 2021.

EUROSTAT, *Gross nitrogen balance on agricultural land, 2004-2015*, kg N per ha UAA, 2018, https://ec.europa.eu/eurostat/statistics-explained/index.php?title=File:Gross_nitrogen_balance_on_agricultural_land,_2004-2015,_kg_N_per_ha_UAA.png

EUROSTAT, *Municipal waste generation*, 2021c. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics

EUROSTAT, *Nutrient Budgets – Methodology and Handbook*. Version 1.02. Eurostat and OECD, 2013.

EUROSTAT, Sewage sludge production and disposal from urban wastewater (in dry substance). Retrieval of the EUROSTAT, 2017.

EUROSTAT, *Sewage sludge production and disposal from urban wastewater (in dry substance)*, 2021d, . https://ec.europa.eu/eurostat/databrowser/view/ten00030/default/table?lang=en

EUROSTAT, *Treatment of waste by waste category, hazardousness and waste management operations* (env_wastrt), 2021b, https://ec.europa.eu/eurostat/data/database. Accessed in June 2021.

Fertilizers Europe, *Fertilizer consumption by nutrient 2018*, 2019, https://www.fertilizerseurope.com/fertilizers-in-europe/facts-figures/

Foletto, E. L., Barreto dos Santos, W. R., Jahn, S. L., Santos, W. R. B. dos, Jahn, S. L., Maria, U. F. de S., and Bassaco, M. M., Organic pollutants removal and recovery from animal wastewater by mesoporous struvite precipitation. *Desalination and Water Treatment*, 51, 2013, 2776–2780. https://doi.org/10.1080/19443994.2012.749378

Food and Agriculture Organization of the United Nations (FAO). *FAO/INFOODS Global Food Composition Database for Fish and Shellfish* Version 1.0 – uFiSh1.0. Rome, Italy, 2016.

Frey, R., Method and plant for recovering ammonia in the separation of nitrogen oxides from flue gases. Patent EP0264041, 2018.

Friedmann, H., and McMichael, P., The rise and decline of national agriculture, 1870 to the present. *Sociologia*, 1989.

García-Condado, S., López-Lozano, R., Panarello, L., et al., Assessing lignocellulosic biomass production from crop residues in the European Union: modelling, analysis of the current scenario and drivers of interannual variability. *GCB Bioenergy*, 11, 2019, 809–831

Garnier, J., Le Noë, J., Marescaux, A., Sanz-Cobena, A., Lassaletta, L., Silvestre, M., Thieu, V., and Billen, G., Long-term changes in greenhouse gas emissions from French agriculture and livestock (1852–2014): from traditional agriculture to conventional intensive systems. *Science of the Total Environment*, 660, 2019b: 1486-1501. doi.org/10.1016/j.scitotenv.2019.01.048

Ghosh, S., Lobanov, S., and Lo, V. K., An overview of technologies to recover phosphorus as struvite from wastewater: advantages and shortcomings. *Environmental Science and Pollution Research*, 26, 2019, 19063–19077. https://doi.org/10.1007/s11356-019-05378-6

Giner Santonja G., Georgitzikis K., Scalet B.M., Montobbio P., Roudier S. & Delgado Sancho L. (2017) Best available techniques (BAT) reference document for the intensive rearing of poultry or pigs - Industrial Emissions Directive 2010/75/EU. Seville.

GMB BioEnergie. *Nitrogen recovery from wastewater sludge. Ammonium sulphate from process-air*, 2021. https://www.nutrientplatform.org/en/success-stories/nitrogen-recovery-from-wastewater-sludge/

Grizzetti, B., Vigiak, O., Udias, A., Aloe, A., Zanni, M., Bouraoui, F., Pistocchi, A., Dorati, C., Friedland, R., DeRoo, A., Benitez Sanz, C., Leip, A., and M. Bielza., How EU policies could reduce nutrient pollution in European inland and coastal waters, Global Environmental Change 69, 2021, 102281, doi:10.1016/j.gloenvcha.2021.102281

Grizzetti, B., Vigiak, O., Udias, A., Bisselink, B., Pistocchi, A., Bouraoui, F., Malagó, A., Aloe, A., Zanni, M., Weiss, F., Hristov, J., Wilson, J., Pisoni, E., de Meij, A., Macias, M., Stips, A., A European assessment of freshwater availability and nutrient pollution - Historical analysis and scenarios developed in the project Blue2.2 , Publications Office of the European Union, Luxembourg, 2022, doi:10.2760/924432, JRC130025.

Gysin, A., Lycke, D., and Wirtel, S., The Pearl® and WASSTRIP® processes (Canada). In C. Schaum (Ed.), *Phosphorus: Polluter and Resource of the Future - Removal and Recovery from Wastewater*, 2018, https://doi.org/10.2166/9781780408361_359

Hekman J., Fernandez Milan B., Willcocks J., Scalet B., Merti M., 2021. Assignment #7: Updating of available information for undertaking the assessment of impacts for a possible modification of the IED with regard to aspects of intensive agriculture. Final Report for DG Environment, European Commission. file:///C:/Users/grizzbr/Downloads/Intensive%20Agriculture%20and%20IED%20Final%20Report%202021%20V2%20updated.pdf

Herzel, H., Stemann, J., Simon, S., and Adam, C., Comparison of thermochemical treatment of sewage sludge ash with sodium sulphate in laboratory-scale and pilot-scale experiments. *International Journal of Environmental Science and Technology*, 1, 2021, https://doi.org/10.1007/s13762-021-03252-y

Hijbeek, R., van Ittersum, M. K., ten Berge, H. F. M., Gort, G., Spiegel, H., and Whitmore, A. P., Do organic inputs matter – a meta-analysis of additional yield effects for arable crops in Europe. *Plant and Soil*, 411, 2017, 293–303. https://doi.org/10.1007/s11104-016-3031-x

Himics, M., Fellmann, T., Barreiro-Hurlé, J. et al., Does the current trade liberalization agenda contribute to greenhouse gas emission mitigation in agriculture? *Food Policy*, 76, 2018, 120–129

Huygens D. & Saveyn H.G.M. (2018) Agronomic efficiency of selected phosphorus fertilisers derived from secondary raw materials for European agriculture. A meta-analysis. Agron. Sustain. Dev. 38.

Huygens D., Orveillon G., Lugato E., Tavazzi S., Comero S., Jones A., . . . Saveyn H. (2020) Technical proposals for the safe use of processed manure above the threshold established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC), EUR 30363 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-76-21539-4, doi:10.2760/373351, JRC121636.

Huygens, D., and Saveyn, H. G. M., Agronomic efficiency of selected phosphorus fertilisers derived from secondary raw materials for European agriculture. A meta-analysis. *Agronomy for Sustainable Development*, Springer Verlag/EDP Sciences/INRA, 38, 2018, 52, https://doi.org/10.1007/s13593-018-0527-1

Huygens, D., Orveillon, G., Lugato, E., and Tavazzi, S., Technical proposals for the safe use of processed manure above the threshold established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC), 2020, https://doi.org/10.2760/373351

Huygens, D., Saveyn, H. G. M., Tonini, D., Eder, P., and Delgado Sancho, L., Technical proposals for selected new fertilising materials under the Fertilising Products Regulation (Regulation (EU) 2019/1009) - Process and quality criteria, and assessment of environmental and market impacts for precipitated phosphate salts & derivate. In EUR 29841 EN, Publications Office of the EU, Luxembourg, 2019, https://doi.org/10.2760/186684

Huyghe, C., De Vliegher, A., van Gils, B., and Peeters, A., (Eds), Grasslands and herbivore production in Europe and effects of common policies, 2014.

ICL-Fertilizers, *ICL Amsterdam opens phosphate recycling unit*, 2019, https://icl-sf.com/global-en/news/icl-amsterdam-opens-phosphate-recycling-unit/

Janssens-Maenhout, G., et al., EDGAR v4.3.2 Global Atlas of the three major greenhouse gas emissions for the period 1970–2012 *Earth Syst. Sci. Data*, 11, 2019, 959–1002.

Jones, D. B., Factors for converting percentages of nitrogen in foods and feeds into percentages of protein. US Department of Agriculture-circ. 183.Washington, DC, 1941.

Kabbe, C., Inventory of phosphorus "recovery and /or recycling" facilities operating or under construction at wastewater treatment installations (Status 2021), 2021, https://www.phosphorusplatform.eu/images/download/Kabbe_P-recorecy_tech_implementation Table_2021_10_31.pdf

Köninger, J., Lugato, E., Panagos, P., et al., Manure management and soil biodiversity: towards more sustainable food systems in the EU*. Agric. Syst.*, 194, 2021, 103251

Körner, I., Stegmann, R., and Hrsg. Stickstoffhaushalt bei der Kompostierung: Bilanzen, Gehalte, Umsetzungs- und Austragsprozesse. Habilitationsschrift, Technische Universität Hamburg-Harburg. In: *Hamburger Berichte* 33. Verlag Abfall aktuell, Stuttgart, Deutschland ISBN: 978-3-9812867-0-0, 320 Seiten, 2009. https://d-nb.info/104845438X/34.

Krishnamurthy, R., and Andrecovich, M. J., Separation of carbon dioxide and nitrogen from combustion exhaust gas with nitrogen and argon by-product recovery. Patent EP0469781B1, 1991.

Langeveld, J.W.A., Verhagen, A., Neeteson, J.J., et al., Evaluating farm performance using agri-environmental indicators: recent experiences for nitrogen management in The Netherlands. *J Environ Manag*, 82, 2007, 363–376

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., and Garnier, J., 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. *Environ. Res. Lett.*, 9, 2014, doi:10.1088/1748-9326/9/10/105011

Le Noë, J., Billen, G., and Garnier, J., How the structure of agro-food systems shapes nitrogen, phosphorus, and carbon fluxes: the Generalized Representation of Agro-Food System applied at the regional scale in France. *Science of the Total Environment*, 586, 2017, 42–55, http://dx.doi.org/10.1016/j.scitotenv.2017.02.040

Le Noë, J., Billen, G., Esculier, F., and Garnier, J., Long-term socioecological trajectories of agro-food systems revealed by N and P flows in French regions from 1852 to 2014. *Agr Ecosyst Env*., 265, 2018, 132-143. https://doi.org/10.1016/j.jenvman.2017.09.039

Le Noë, J., Roux, N., Billen, G., Gingrich, S., Erb, K., Krausmann, F., Thieu, V., Silvestre, M., Garnier, J., The phosphorus legacy offers opportunities for agro ecological transition (France 1850-2075). *Environ. Res. Lett*., 15, 2020, 064022 https ://doi.org/10.1088/1748-9326/ab82cc

Lehman, J., and Joseph, S., Biochar for environmental management - science, technology and implementation. 2nd edition. Routledge, New York, 2015.

Leng, L., Yang, L., Leng, S., Zhang, W., Zhou, Y., Peng, H., Li, H., Hu, Y., Jiang, S., and Li, H., A review on nitrogen transformation in hydrochar during hydrothermal carbonization of biomass containing nitrogen. *Science of the Total Environment*, 756, 2021, 143679. https://doi.org/10.1016/j.scitotenv.2020.143679

Lugato, E., and Jones, A., Modelling Soil Organic Carbon Changes Under Different Maize Cropping Scenarios for Cellulosic Ethanol in Europe. *Bioenergy Research*, 8, 2015, 537-545. DOI: 10.1007/s12155-014-9529-2

Lugato, E., Bampa, F., Panagos, P., Montanarella, L., and Jones, A., Potential carbon sequestration of European arable soils estimated by modelling a comprehensive set of management practices. *Global Change Biology*, 20, 2014a, 3557-3567. DOI: 10.1111/gcb.12551

Lugato, E., Cescatti, A., Jones, A., Ceccherini, G., Duveiller, G., Maximising climate mitigation potential by carbon and radiative agricultural land management with cover crops. *Environmental Research Letters*, 15, 2020, 094075, DOI: 10.1088/1748-9326/aba137

Lugato, E., Leip, A., and Jones, A. Mitigation potential of soil carbon management overestimated by neglecting N2O emissions. *Nature Clim Change*, 8, 2018a, 219–223, https://doi.org/10.1038/s41558-018-0087-z

Lugato, E., Panagos, P., Bampa, F., Jones, A., and Montanarella, L., A new baseline of organic carbon stock in European agricultural soils using a modelling approach. *Global Change Biology*, 20, 2014b, 313-326. DOI: 10.1111/gcb.12292

Lugato, E., Paniagua, L., Jones, A., de Vries, W., and Leip, A., Complementing the topsoil information of the Land Use/Land Cover Area Frame Survey (LUCAS) with modelled N2O emissions. *PLoS ONE*, 12, 2017, e0176111. https://doi.org/10.1371/journal.pone.0176111

Lugato, E., Paustian, K., Panagos, P., Jones, A., and Borrelli, P., Quantifying the erosion effect on current carbon budget of European agricultural soils at high spatial resolution. *Global Change Biology*, 22, 2016, 1976-1984. DOI: 10.1111/gcb.13198

Lugato, E., Smith, P., Borrelli, P., Panagos, P., Ballabio, C., Orgiazzi, A., Fernandez-Ugalde, O., Montanarella, L., and Jones, A., Soil erosion is unlikely to drive a future carbon sink in Europe. *Science Advances*, 4, 2018b, aau3523, DOI: 10.1126/sciadv.aau3523

MacDonald, G.K., Bennett, E.M., Potter, P.A., and Ramankutty, N., Agronomic phosphorus imbalances across the world's croplands. *Proc Natl Acad Sci.*, 108, 2011, 3086–3091

Makowski, D., Bosco, S., Chen, M., Montero-Castaño, A., Pérez-Soba, M., Schievano, A., & Terres, J., Systematic review of meta-analyses to assess the impacts of farming practices - A methodological framework. (2021, June 1), https://doi.org/10.31219/osf.io/byuw9

Martin, T.M.P., Esculier, F., Levavasseur, F., and Houot, S., Human urine-based fertilizers: a review. *Crit. Rev. Environ. Sci. Technol.,* 52, 2020, 890-936, https://doi.org/10.1080/10643389.2020.1838214

Massé, D. I., Saady, N. M. C., and Gilbert, Y., Potential of biological processes to eliminate antibiotics in livestock manure: An overview. *Animals*, 4, 2014, 146–163. https://doi.org/https://doi.org/10.3390/ani4020146

Mc Carty, P.L., What is the best biological process for nitrogen removal: when and why? *Environ. Sci. Technol.* 52, 2018, 3855-3841.

Meesuk, S., Cao, J. P., Sato, K., Hoshino, A., Utsumi, K., and Takarada, T., Nitrogen conversion of pig compost during pyrolysis. *Journal of Chemical Engineering of Japan*, 46, 2013, 556–561. https://doi.org/10.1252/jcej.12we262

Menkveld, W., and Broeders, E., Recovery of ammonia from digestate as fertilizer. *Water Practice and Technology*, 13, 2018, 382–387. ttps://doi.org/10.2166/wpt.2018.049%0A

Monforti, F., Lugato, E., Motola, V., Bodis, K., Scarlat, N., and Dallemand, J.-F., Optimal energy use of agricultural crop residues preserving soil organic carbon stocks in Europe. *Renewable and Sustainable Energy Reviews*, 44, 2015, 519-529. DOI: 10.1016/j.rser.2014.12.033

Muys, M., Phukan, R., Brader, G., Samad, A., Moretti, M., Haiden, B., Pluchon, S., Roest, K., Vlaeminck, S. E., and Spiller, M., A systematic comparison of commercially produced struvite: Quantities, qualities and soil-maize phosphorus availability. *Science of the Total Environment*, 756, 2021, 143726, https://doi.org/10.1016/j.scitotenv.2020.143726

Neuwahl, F., Cusano, G., Benavides, J. G., Holbrook, S., and Serge, R., Best Available Techniques (BAT) Reference Document for Waste Incineration. In Industrial Emissions Directive 2010/75/EU (Integrated Pollution Prevention and Control) Frederik, 2019.

Nicholas, D. M., Bushinsky, J. P., Kumar, R., and Kratz, W. C., *Adsorptive process for recovering nitrogen from flue gas*. Patent US 7-244429, 1991.

O'Neill, D.W., Fanning, A.L., Lamb, W.F., Steinberger, J.K., 2018. A good life for all within planetary boundaries. Nat. Sustain. 1, 88–95. https://doi.org/10.1038/s41893-018-0021-4

OEWAV, Critical resource phosphorus - restoration of interrupted phosphorus cycles by using the available phosphorus sources: municipal wastewater and animal by-products. Austrian Water and Waste Management Association, 2018.

Oreggioni, G.D., Mahiques, O., Schaaf, E., Duerr, M., Monforti-Ferrario, F., Muntean, M., Guizzardi, D., Vignati, E., and Crippa, M., The impacts of technological changes and regulatory frameworks on global air pollutant emissions from energy industry and road transport, *Energy Policy*, submitted, 2021.

Oreggioni, G.D., Monforti-Ferraio, F., Crippa, M., Muntean, M., Schaaf, E., Guizzardi, D., Solazzo, E., Duerr, M., Perry, M., and Vignati, E., Climate change in a changing world: Socio-economic and technological transitions, regulatory frameworks and trends on global greenhouse gas emissions from EDGAR v.5.0, *Global Environmental Change*, 70, 2021, 102350, https://doi.org/10.1016/j.gloenvcha.2021.102350.

Palakodeti, A., Azman, S., Rossi, B., Dewil, R., and Appels, L., A critical review of ammonia recovery from anaerobic digestate of organic wastes via stripping. *Renewable and Sustainable Energy Reviews*, 143, 2021, https://doi.org/10.1016/j.rser.2021.110903

Panagos, P., Ballabio, C., Himics, M., et al., Projections of soil loss by water erosion in Europe by 2050. *Environ Sci. Policy*, 124, 2021, 380–392

Panagos, P., Muntwyler, A., Liakos, L., Borrelli, P., Biavetti, I., Bogonos, M. and Lugato, E., Phosphorus plant removal from European agricultural land. *Journal of Consumer Protection and Food Safety*, 2022, DOI: 10.1007/s00003-022-01363-3

Panagos, P., Köninger, J., Ballabio, C., Liakos, L., Muntwyler, A., Borrelli, P. and Lugato, E., 2022b. Improving the phosphorus budget of European agricultural soils. Science of the Total Environment, 853, 158706.

Patel, A., Mungray, A.A., and Mungray, A.K., Technologies for the recovery of nutrients, water and energy from human urine: a review. *Chemosphere*, 259, 2020, 127372.

Poikane S., Richard K. Johnson, Leonard Sandin, Ann Kristin Schartau, Angelo G. Solimini, Gorazd Urbanič, Kęstutis Arbačiauskas, Jukka Aroviita, Wim Gabriels, Oliver Miler, Martin T. Pusch, Henn Timm, Jürgen Böhmer, 2016. Benthic macroinvertebrates in lake ecological assessment: A review of methods, intercalibration and practicalrecommendations. Science of the Total Environment 543, 123–134.

Poikane S., Sebastian Birk, Jürgen Böhmer, Laurence Carvalho, Caridad de Hoyos, Hubert Gassner, Seppo Hellsten, Martyn Kelly, Anne Lyche Solheim, Mikko Olin, Karin Pall, Geoff Phillips, Rob Portielje, David Ritterbusch, Leonard Sandin, Ann-Kristin Schartau, Angelo G. Solimini, Marcel van den Berg, Georg Wolfram, Wouter van de Bund, 2015. A hitchhiker's guide to European lake ecological assessment and intercalibration. Ecological Indicators 52, 533–544.

Poikane, S., Kelly, M.G., Salas Herrero, F., Pitt, J.-A., Jarvie, H.P., Claussen, U., Leujak, W., Lyche Solheim, A., Teixeira, H., Phillips, G., 2019a. Nutrient criteria for surface waters under the European Water Framework Directive: Current state-of-theart, challenges and future outlook. Sci. Total Environ. 695, 133888 https://doi.org/10.1016/j.scitotenv.2019.133888.

P-Rex, Sustainable sewage sludge management fostering phosphorus recovery and energy efficiency - Final Report. In ENV.2012.6.5-2 Collaborative project, 2015

Qi, A., Holland, R.A., Taylor, G.T., Richter, G.M., Grassland futures in Great Britain – Productivity assessment and scenarios for land use change opportunities. *Science of the Total Environment*, 634, 2017, 1108-1118.

Quemada, M., Lassaletta, L., Jensen, L.S., Godinot, O., Brentrup, F., Buckley, C., Foray, S., Hvid, S.K., Oenema, J., Richards, K.G., Oenema, O., Exploring nitrogen indicators of farm performance among farm types across several European case studies, *Agricultural Systems*, 177, 2020b, 102689, https://doi.org/10.1016/j.agsy.2019.102689

Quemada, M., Lassaletta, L., Leip, A., Jones, A., and Lugato, E. Integrated management for sustainable cropping systems: Looking beyond the greenhouse balance at the field scale. *Global Change Biology*, 2020a, 14989, https://doi.org/10.1111/gcb.14989

Rak, A., Das REMONDIS TetraPhos-Verfahren Phosphorrückgewinnung Das REMONDIS TetraPhos ®-Verfahren. *VIVIS*, 491–494, 2018. https://www.vivis.de/wp-content/uploads/VvK/2018_VvK_491-494_Rak

Rapf, M., and Raupenstrauch, H., RecoPhos and other thermo-chemical processes for the recovery of phosphorus from sewage sludge. Depotech, 2012.

Reza, T. M., Freitas, A., Yang, X., Hiibel, S., Lin, H., and Coronella, C. J., Hydrothermal carbonization (HTC) of cow manure: Carbon and nitrogen distributions in HTC products. *American Institute of Chemical Engineers Environ Prog*., 35, 2016, 1002–1011. https://doi.org/10.1002/ep.12312

Ricardo Energy & Environment, *Review of National Air Pollutant Projections and Assessment of National Air Pollution Control Programmes*, 2020, https://ec.europa.eu/environment/air/pdf/reduction_napcp/Horizontal%20review_final%2010Jun20.pdf

Ringeval, B., et al., Phosphorus in agricultural soils: drivers of its distribution at the global scale. *Global Change Biol.,* 23, 2017, 1–15, https://doi.org/10.1111/gcb.13618

Risberg, K., Cederlund, H., Pell, M., Arthurson, V., and Schnürer, A., Comparative characterization of digestate versus pig slurry and cow manure – Chemical composition and effects on soil microbial activity. *Waste Management*, 61, 2017, 529–538. https://doi.org/10.1016/j.wasman.2016.12.016

Rockström J. et al., 2009. Planetary boundaries: Exploring the safe operating space for humanity. Ecol. Soc. 14, 32 (2009). http://www.ecologyandsociety.org/vol14/iss2/art32/

Rogoff, M. J., and Screve, F., Waste to Energy. Chapter 8 - Permitting issues. In *Waste-to-Energy*, 2011, 89–116, https://doi.org/10.1016/b978-1-4377-7871-7.10008-5

Sarkar, D. K., Thermal Power Plant. Design and Operaton. *Chapter 14 - Air pollution control*. In Air pollution control, 2015, https://doi.org/10.1016/B978-0-12-801575-9.00014-7

Saud, A., Havukainen, J., Peltola, P., and Horttanainen, M., Integrating pyrolysis or combustion with scrubbing to maximize the nutrient and energy recovery from municipal sewage sludge. *Recycling*, 6, 2021, 52, https://doi.org/10.3390/recycling6030052

Saveyn, H., and Eder, P., *End-of-waste criteria for biodegradable waste subjected to biological treatment (compost and digestate)*. In Technical proposals . EUR 26425. Luxembourg (Luxembourg): Publications Office of the European Union; 2013. JRC87124. https://doi.org/10.37307/j.1863-9763.2014.04.07

Scarlat, N., Fahl, F., Lugato, E., Monforti-Ferrario, F., and Dallemand, J.-F., Integrated and spatially explicit assessment of sustainable crop residues potential in Europe. *Biomass and Bioenergy*, 122, 2019, 257-269. DOI: 10.1016/j.biombioe.2019.01.021

Scarlat, N., Martinov, M., and Dallemand, J.-F., Assessment of the availability of agricultural crop residues in the European Union: potential and limitations for bioenergy use. *Waste Manag*., 30, 2010, 1889–1897

Schipper, W. J., Klapwijk, A., Potjer, B., Bulkens, W. H., Temmink, B. G., Kiestra, F. D., and Lijmbach, A. C., Phosphate recycling in the phosphorus industry. *Environ. Technol*., 22, 2001, 1337–1345. https://doi.org/doi: 10.1080/09593330.2001.9619173

Schlumberger, S., Das Phos4Life Verfahren als Lösungsansatz für die Phosphorrückgewinnung aus Klärschlammasche, 2017.

Schoumans, O. F., Rulkens, W. H., Oenema, O., and Ehlert, P., *Phosphorus recovery from animal manure - Technical opportunities and agro-economical perspectives*. In Alterra report 2158, 2010, https://edepot.wur.nl/163684

SERAPLANT, Phosphorrecycling im Industriemaßstab - Dünger aus Sekundärrohstoffen, 2021, https://seraplant.com/

Shen, Y., Haibin, Z., Meng, H., and Guo, R., Generation and Emission of Ammonia During the Full-Scale Composting of Sewage Sludge. *Waste and Biomass Valorization*, 11, 2020, 4757–4766.

Sigurnjak, I., Brienza, C., Snauwaert, E., De Dobbelaere, A., De Mey, J., Vaneeckhaute, C., Michels, E., Schoumans, O., Adani, F., and Meers, E., Production and performance of bio-based mineral fertilizers from agricultural waste using ammonia (stripping-)scrubbing technology. *Waste Management*, 89, 2019, 265–274. https://doi.org/10.1016/j.wasman.2019.03.043

Simpson, D., Benedictow, A., Berge, H., Bergström, R., Emberson, L.D., Fagerli, H., Flechard, C.R., Hayman, G.D., Gauss, M., Jonson, J.E., Jenkin, M.E., Nyíri, A., Richter, C., Semeena, V.S., Tsyro, S., Tuovinen, J.-P., Valdebenito, Á., and Wind, P., The EMEP MSC-W chemical transport model – technical description, *Atmos. Chem. Phys*., 12, 2012, 7825–7865, https://doi.org/10.5194/acp-12-7825-2012.

Smit, H.J., Metzger, M.J., and Ewert, F., Spatial distribution of grassland productivity and land use in Europe. *Agricultural Systems*, 98, 2008, 208-219. https://doi.or/10.1016/j.agsy.2008.07.004

Sommer, S. G., Christensen, M. L., Schmidt, T., and Jensen, L. S., *Animal manure recycling. Treatment and management.* John Wiley & Sons Ltd, 2013.

Steffen W. et al. 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347, 1259855. DOI: 10.1126/science.1259855

Sutton, M.A., Howard, C.H., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H., Grizzetti, B., The European Nitrogen Assessment, Cambridge University Press, 664 pp, ISBN: 9781107006126, 2011

SYSTEMIC. 2020. Living Lab conversations: N stripping-scrubbing technologies. H2020 EU-project SYSTEMIC

Tasca, A. L., Puccini, M., Gori, R., Corsi, I., Galletti, A. M. R., and Vitolo, S., Hydrothermal carbonization of sewage sludge: A critical analysis of process severity, hydrochar properties and environmental implications. *Waste Management*, 93, 2019, 1–13. https://doi.org/10.1016/j.wasman.2019.05.027

Thunis, P., Crippa, M., Cuvelier, C., Guizzardi, D., de Meij, A., Oreggioni, G., and Pisoni, E., Sensitivity of air quality modelling to different emission inventories: A case study over Europe*, Atmospheric Environment: X*, 10, 2021, 100111, https://doi.org/10.1016/j.aeaoa.2021.100111.

Tonini, D., Saveyn, H., and Huygens, D., Environmental and health co-benefits for advanced phosphorus recovery. *Nature Sustainability*, 2, 2019, 1051-1061.

USDA, *Food Composition Database*, v.3.9.5.1_2018-09-23, 2018. Available online. https://ndb.nal.usda.gov/ndb/ Accessed May 2019.

van den Pol-van Dasselaar, A., Aarts, H. F. M., De Vliegher, A., Elgersma, A., Reheul, D., Reijneveld, J. A., Verloop, J., and Hopkins, A. (Eds.), *Grassland and forages in high output dairy farming systems* (Vol. 20). European Grassland Federation, 2015.

van Dijk, K. C., Lesschen, J. P., and Oenema, O., Phosphorus flows and balances of the European Union Member States. *Science of the Total Environment*, 542, 2016, 1078–1093. https://doi.org/10.1016/j.scitotenv.2015.08.048

Van Eekert, M., Weijma, J., Verdoes, N., De Buisonjé, F., Reitsma, B., Van den Bulk, J., and Van Gastel, J., Explorative research on innovative nitrogen recovery. *STOWA* 2012-51, 2012, ISBN 978.90.5773.585.1. http://www.stowa.nl/Upload/publicaties/STOWA%0A2012%0A51_LR.pdf

Vehlow, J., Transformation of waste combustion facilities from major polluters to pollution sinks. In Waste to Energy Conversion Technology. Woodhead Publishing Limited, 2013, https://doi.org/10.1533/9780857096364.3.179

Velthof, G. L., Lesschen, J. P., Schils, R. L. M., Smit, A., Elbersen, B. S., Hazeu, G. W., Mucher, C. A., and Oenema, O., *Grassland areas, production and use. Lot 2. Methodological studies in the field of Agro-Environmental Indicators*. Alterra Wageningen, 2014, https://ec.europa.eu/eurostat/documents/2393397/8259002/Grassland_2014_Final+report.pdf/58aca1dd-de6f-4880-a48e-1331cafae297

Velthof, G. L., Mineral concentrate from processed manure as fertiliser. Alterra Report 2650, 36, 2015.

Velthof, G.L., *Task 1 of Methodological studies in the field of Agro-Environmental Indicators*. Lot 1 excretion factors. Final draft. Alterra, Wageningen, 2014. http://ec.europa.eu/eurostat/documents/2393397/8259002/LiveDate_2014_Task1.pdf/e1ac8f30-3c76-4a61-b607-de99f98fc7cd

Verbeke, M., Hermann, L., Schoumans, O., and Regelink, I., SYSTEMIC: Systemic large scale eco-innovation to advance circular economy and mineral recovery from organic waste in Europe. WP3, D3.4, Market Research in Europe, 2021.

Vigiak, O., Grizzetti, B., Zanni, M. et al., Domestic waste emissions to European waters in the 2010s*. Sci. Data*, 7, 2020, 33. https://doi.org/10.1038/s41597-020-0367-0

Vigiak, O., Grizzetti, B., Zanni, M., Aloe, A., Dorati, C., Bouraoui, F., and Pistocchi, A., *Domestic waste emissions to European freshwaters in the 2010s* (v. 1.0). European Commission, Joint Research Centre (JRC), 2019, [Dataset] PID: http://data.europa.eu/89h/0ae64ac2-64da-4c5e-8bab-ce928897c1fb

Vigiak, O., Udías, A., Grizzetti, B., Zanni, M., Aloe, A., Weiss, F., Hristov, J., Bisselink, B., de Roo, A., Pistocchi, A., Recent regional changes in nutrient fluxes of European surface waters*, Science of the Total Environment*, 858, 2023, 160063, http://dx.doi.org/10.1016/j.scitotenv.2022.160063

Westhoek H., et al., *Nitrogen on the Table : the influence of food choices on nitrogen emissions, greenhouse gas emissions and land use in Europe*. ENA special report on Nitrogen and Food. CEH, UK, 2015

Witter, E., and Lopez-Real, J., Nitrogen losses during the composting of sewage sludge, and the effectiveness of clay soil, zeolite, and compost in adsorbing the volatilized ammonia. *Biological Wastes*, 23, 2013, 279–294.

Wong, J., Wang, X., and Selvam, A., Improving Compost Quality by Controlling Nitrogen Loss During Composting. In *Current Developments in Biotechnology and Bioengineering*, 2017, pp. 59–82.

World Bank Commodity Price Data (pink sheet) (2 March). 2022.

World-Bank, *Fertilizer prices*, 2021, https://data.worldbank.org/

Yang, X., Liu, E., Zhu, X., Wang, H., Liu, H., Liu, X., and Dong, W., Impact of Composting Methods on Nitrogen Retention and Losses during Dairy Manure Composting. *Int J Environ Res Public Health*, 16, 2019, 3324. https://doi.org/10.3390/ijerph16183324

Zarebska, A., Romero Nieto, D., Christensen, K., Fjerbaek Sotoft, L., and Norddahl, B., Ammonium Fertilizers Production from Manure: A Critical Review. *Critical Reviews in Environmental Science and Technology*, 45, 2015.

Zimmermann, J., and Dott, W., Sequenced bioleaching and bioaccumulation of phosphorus from sludge combustion - A new way of resource reclaiming. *Advanced Materials Research*, 71–73, 2009, 625–628. https://doi.org/10.4028/www.scientific.net/AMR.71-73.625

List of boxes

List of figures

List of tables

Annexes

Annex 1. Nitrogen and phosphorus fluxes in EU27 Member States (reference year 2015)

Table A1.1. Nitrogen fluxes from and to AIR (t N/y)

Member State	From LAND to AIR(1)			From WASTE to AIR		From AIR to LAND		
	N2O	NOx	NH3	N2	Nr(2)	Atmospheric deposition	Mineral Fert	N plant fixation
AT	7968	49464	81131	19497	1048	121105	109347	20801
BE	19963	66514	106160	44319	24300	67260	183537	9372
BG	7818	30892	65881	8659	13753	87609	291991	9163
CY	548	5327	5173	1621	735	3473	4000	95
CZ	14458	72430	90803	22988	18505	114915	362006	24339
DE	81186	333619	674532	236121	150685	683191	1678089	116607
DK	11265	32419	129408	32910	17021	74441	195313	35412
EE	1980	8920	13540	3254	3188	28847	51000	7920
EL	8996	68463	65818	18603	18809	82771	186018	48232
ES	45372	220747	444459	119185	103514	344957	1037041	77079
FI	12780	51562	45012	15746	7836	104966	145998	5932
FR	83042	227067	703430	202922	81402	633458	2206820	409828
HR	5672	15933	38674	12643	2371	62114	129995	10595
HU	12774	36885	89223	21151	14672	107226	329001	28190
IE	17175	23873	147931	17427	2987	70704	345989	57277
IT	34412	240469	441705	111651	115502	432475	594013	186801
LT	11334	13686	49938	7748	6452	64129	154008	30130
LU	602	7781	4451	1670	1027	4636	7523	840
LV	3397	9741	23547	4425	2917	51334	57007	15174
MT	87	2076	1057	1036	511	190	1252	21
NL	25299	81671	194498	61162	46847	110755	195001	19012
PL	56632	191316	397177	77562	59187	438013	1171999	30462
PT	7423	47165	61744	26686	18469	48105	79012	6382
RO	23448	68724	185131	26833	30576	220229	519988	61961
SE	18839	57450	67459	24533	14974	165639	163003	26496
SI	1596	9298	16447	2822	0	28746	25101	1777
SK	6667	22331	34983	7853	3763	58407	119996	16483

(1) Except waste sector
(2) Less reactive N emissions to water

Table A1.2. Nitrogen fluxes from LAND (t N/y)

Member State	To LAND	To FOOD	To WASTE	To WATER				To OUTSIDE
	Manure Fert	Food production	Agric Waste/losses	NO3 leaching	OrgN leaching	Losses in soil erosion	Uncollected dom emissions to water	NET exp Agric Products([1])
AT	112867	133313	2171	27846	1033	31323	6	-33593
BE	173813	274065	3403	74140	1257	8539	3806	-213221
BG	65681	122343	3956	23037	616	23825	1930	119795
CY	10383	8004	206	2287	42	1026	1017	-9862
CZ	81139	152782	4045	44598	762	19321	3930	67229
DE	900501	1349510	25411	739166	13097	88389	2434	-272701
DK	203401	145122	9039	185190	3535	4907	0	51893
EE	35967	24031	1318	12944	161	1667	2	17031
EL	129885	119097	4259	65688	1089	13370	2117	-38176
ES	614185	909516	23388	285394	4066	84371	1571	-388908
FI	65450	64261	2360	28425	348	2223	2367	6950
FR	1085467	1047065	39638	602042	15235	146256	13639	578942
HR	31771	45215	2298	24356	500	17056	4087	20532
HU	95244	160108	5306	33691	812	21282	2433	146337
IE	359951	83939	3238	201665	6583	19154	1239	-15523
IT	407860	793807	14705	220478	6081	153285	6841	-321864
LT	49712	55911	2613	57522	678	6771	1449	66039
LU	10215	4646	69	4013	56	1196	11	-1049
LV	30367	22753	2061	23499	298	4864	970	46299
MT	1960	2978	109	2	0	6	0	-3094
NL	402167	410440	8978	158166	3231	2543	251	-337670
PL	366800	501660	15002	524673	6423	42611	17865	105387
PT	115926	191190	3391	41337	1044	10527	385	-125494
RO	199157	232785	7153	47380	1459	78583	13848	213422
SE	105340	100873	3601	90088	889	10700	1	22893
SI	16738	17225	357	10454	281	11951	1738	-2165
SK	34913	43251	1894	14943	252	16329	3286	41207

(1) difference between export and import, negative values indicate net import.

Table A1.3. Nitrogen fluxes from FOOD; WASTE and WATER (t N/y)

Member State	From FOOD			From WASTE			From WATER	
	To LAND	To OUTSIDE	To WASTE	To LAND		To water	To SEA	To OUTSIDE
	Feed and biorefineries	NET Export Food(1)	Waste	Sludge/ composting	Uncollected Domestic emissions	Sewer collected emissions	Load to sea	Stream network net export(2)
AT	37366	7927	53567	12566	9	19095	0	90549
BE	79007	6179	94099	8996	5709	10125	55855	14127
BG	59498	-1415	38142	396	2894	8778	9303	65722
CY	767	-952	5473	500	1526	696	3692	0
CZ	45163	3628	67271	7329	5895	8103	0	66162
DE	375843	31662	521863	31911	3651	96638	257067	207703
DK	20237	6866	64937	3108	0	11200	73034	-615
EE	6927	504	9350	1424	3	1378	20385	-8177
EL	38574	-3995	57414	3553	3176	9477	224100	-88360
ES	255146	4045	391767	50379	2357	96487	244155	67848
FI	10831	-2297	40187	4531	3550	7940	36998	8213
FR	264140	8689	426418	31893	20459	74053	647735	52007
HR	9687	-2441	30763	683	6131	6035	26874	34178
HU	73131	-4159	62998	6890	3649	12316	0	60164
IE	14351	902	40134	6034	1858	10011	120697	-779
IT	215643	-8573	415734	80752	10261	96085	403657	-7101
LT	14804	-860	24752	3794	2173	2939	48791	-17271
LU	662	-957	4115	659	16	703	0	3008
LV	4589	-1308	13391	1706	1455	1734	46083	-25813
MT	697	-1080	2563	0	0	768	4301	0
NL	144718	14240	140553	14371	376	17284	397801	-291964
PL	84274	27187	250222	38478	26798	39361	224235	-17209
PT	57804	-1492	86834	11918	577	24783	132356	-68156
RO	61644	-3676	122308	3445	20772	30338	3402	136263
SE	20695	-8298	63310	10057	1	13695	72378	-962
SI	3071	559	9259	1289	2606	2331	295	24231
SK	14343	-3102	23089	2999	4929	2498	0	30624

(1) Difference between export and import, negative values indicate net import.
(2) In this case OUTSIDE indicate other countries (EU27 or outside EU27)

Table A1.4. Main phosphorus fluxes to and from LAND (t P/y)

Member State	From ROCK	From LAND					
	To LAND	To LAND	To FOOD	To WASTE	To WATER		To OUTSIDE
	Mineral Fert	Manure Fert	Food Production	Agric waste	Losses in soil erosion	Uncollected dom emissions to water	NET exp Agric Products(1)
AT	18321.9	43489.7	20634.4	408.2	5280.7	0.8	-7392.0
BE	10562.3	62175.2	42370.4	562.6	932.2	521.8	-49762.9
BG	23548.4	16955.6	18684.9	759.3	7162.2	300.7	32775.4
CY	872.0	2869.7	1145.8	27.1	14.0	153.4	-2902.6
CZ	20934.9	31457.2	23455.3	728.1	2486.6	571.3	19814.1
DE	130626.5	332257.4	205348.8	4244.8	5732.3	339.2	-23104.2
DK	5694.6	59840.5	21431.6	928.6	950.3	0	11758.7
EE	5275.9	6635.6	3525.2	169.4	101.4	0.2	4686.4
EL	24009.3	32972.0	17286.7	564.5	15449.7	305.1	-8652.6
ES	186329.7	209582.5	132536.8	2981.2	43860.8	215.7	-94893.7
FI	11103.1	21722.0	9434.7	280.3	652.8	316.8	3820.4
FR	220603.1	388414.8	157506.5	6857.4	20134.8	1912.6	194943.5
HR	14388.6	10340.2	6796.4	324.3	8253.2	599.6	4997.6
HU	28119.4	34028.7	23529.1	999.5	1938.5	360.0	42439.6
IE	36362.4	108811.3	12805.8	353.1	9979.8	173.5	-5746.5
IT	75769.7	178977.4	115935.9	2314.1	39939.2	940.2	-78031.8
LT	17878.3	16492.1	8686.2	393.1	494.5	197.4	18479.5
LU	21.3	4049.8	645.6	12.8	109.2	1.5	-313.2
LV	9157.6	9039.8	3234.1	252.3	392.6	134.0	12666.0
MT	139.9	529.6	411.6	12.5	0	0	-776.0
NL	6072.4	116315.9	62509.3	1134.4	683.4	32.3	-83691.4
PL	171780.7	163464.8	75250.2	2401.8	2949.2	2571.4	30109.5
PT	14388.5	36020.5	26447.6	397.3	3860.0	52.1	-28221.8
RO	66283.9	74148.2	35177.6	1236.4	7414.9	2031.3	58814.8
SE	11769.5	32038.8	14412.2	437.1	782.2	0.1	9045.7
SI	4797.0	9468.9	2520.3	66.7	2340.2	254.3	-607.0
SK	5229.0	9889.4	6362.6	360.1	1091.3	514.6	11274.7

(1) Difference between export and import, negative values indicate net import.

Table A1.5. Phosphorus fluxes from FOOD; WASTE and WATER (t P/y)

Member State	From FOOD			From WASTE			From WATER	
	To LAND	To OUTSIDE	To WASTE	To LAND		To water	To SEA	To OUTSIDE
	Feed and biorefineries	NET Export Food(1)	Waste	Sludge/ composting	Uncollected Domestic emissions	Sewer collected emissions	Load to sea	Stream network net export(2)
AT	5527.1	966.0	6879.4	4905.1	1.2	1279.9	0	5057.6
BE	11527.7	396.6	12672.4	3878.4	730.5	871.5	2374.9	567.6
BG	8535.4	-192.2	5051.5	110.9	421.0	1062.9	407.7	5369.0
CY	110.4	-143.9	576.1	154.7	214.7	59.5	695.0	0
CZ	6716.9	878.5	8733.1	3737.7	799.8	757.1	0	2650.1
DE	55286.3	4106.6	67029.2	12357.8	474.9	7032.8	7209.4	7556.2
DK	2830.9	763.4	6693.0	797.9	0	725.3	1930.9	-12.0
EE	953.6	60.7	1189.0	635.7	0.3	97.5	1168.1	-631.3
EL	5541.5	-537.5	6680.2	695.0	427.1	2092.0	37528.9	-15655.1
ES	36186.2	-110.2	48067.6	14722.4	301.9	10186.3	21040.7	5504.5
FI	1562.9	-294.1	4926.8	1459.8	443.6	344.3	3347.8	618.4
FR	39062.4	908.1	48251.2	9464.3	2677.6	7538.5	37700.2	1617.8
HR	1440.3	-452.7	4144.8	123.9	839.4	810.1	1560.4	1199.6
HU	10641.1	-431.5	7282.0	2761.4	504.1	1316.4	0	2529.8
IE	1911.6	-495.3	5255.7	1989.6	242.9	1098.0	8022.2	-42.4
IT	31633.4	-766.0	51000.5	20500.2	1316.3	11603.8	28546.8	-498.9
LT	2086.8	-163.2	3068.0	1621.4	276.4	207.8	2334.3	-1053.3
LU	101.5	-129.2	477.5	256.5	2.1	50.9	0	100.4
LV	620.2	-213.4	1615.3	727.8	187.7	158.6	2439.6	-1443.1
MT	100.3	-145.0	281.0	0	0	202.5	371.2	0
NL	20833.5	1625.6	18650.1	4353.3	45.3	1158.2	13720.8	-10852.4
PL	12373.7	2854.3	30455.0	13062.6	3599.9	3548.6	11770.4	-1293.2
PT	8128.1	-321.1	9755.1	3135.3	73.0	2996.3	12197.3	-5526.2
RO	8927.9	-668.0	15618.2	745.2	2843.8	3775.8	204.6	7714.5
SE	3071.5	-1178.4	7318.2	3858.5	0.1	729.1	5079.1	72.9
SI	446.2	-96.5	1146.0	423.5	356.0	281.1	19.1	1999.3
SK	2137.4	-546.6	3000.3	1286.5	720.5	246.6	0	1408.6

(1) Difference between export and import, negative values indicate net import.
(2) In this case OUTSIDE indicate other countries (EU27 or outside EU27)

Annex 2. Methodology for Nitrogen & Phosphorous material flow analysis in the EU food system

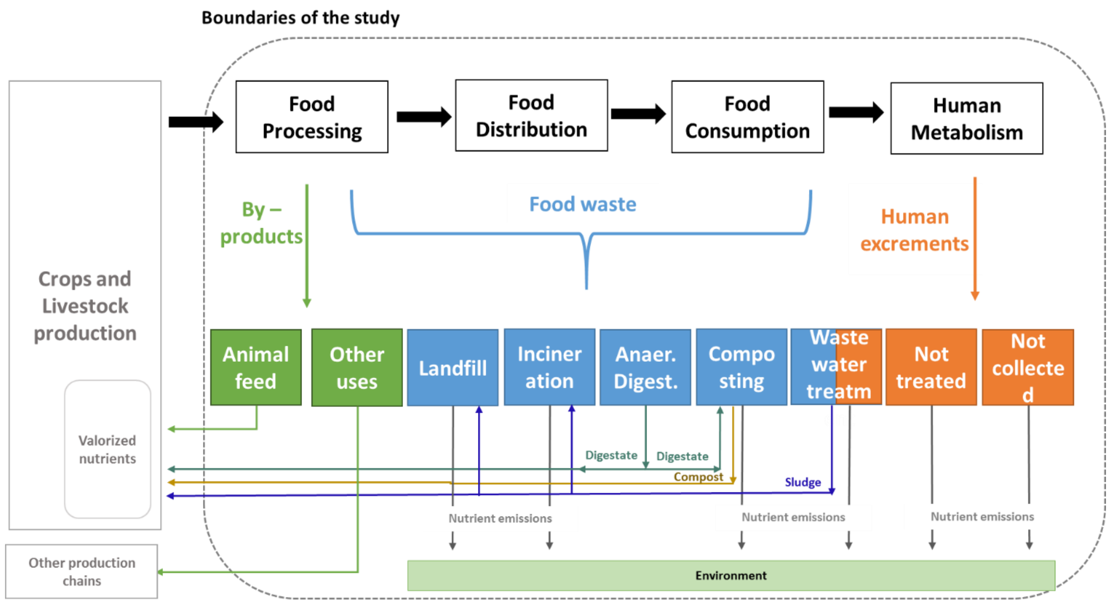
Having as a starting point the model developed by Corrado et al. (2020) who quantified nitrogen flows along the European food chain from food processing up to waste treatment, we have further developed the model to 1) account as well for P flows and 2) downscale the model to quantify both N and P flows in the food system at Member State (MS) level. Based on data availability, the model can be run for all years between 2002 and 2017, and results can be obtained for the EU27 Member States. In this report results are presented for 2015.

As illustrated in figure A2.1, we have quantified the nutrient content in food products, food waste, by-products, human excrements, and losses to the environment originated in the destinations of these flows. Food waste is defined according to EU legislation as: "all food as defined in Article 2 of Regulation (EC) No 178/2002 of the European Parliament and of the Council (European Parliament and Council, 2002) that has become waste" (European Parliament and of the Council, 2018). The definition of 'food' laid down in Regulation (EC) No 178/2002 encompasses food as a whole, along the entire food supply chain from production until consumption. Food also includes inedible parts, where those were not separated from the edible parts when the food was produced, such as bones attached to meat destined for human consumption. Hence, food waste can comprise items that include parts of food intended to be ingested and parts of food not intended to be ingested. 'Waste' means any substance or object which the holder discards or intends or is required to discard (European Parliament and Council, 2008). By-products are defined as surplus food used as animal feed and for non-food uses such as the production of biobased materials. To be noted that the estimation of by-products was done by assuming that a share of the food waste is recycled mainly based on expert judgement Kemna et al. (2017) and literature, and may not be fully representative of the EU MSs. Therefore, there might be the case that a higher amount of what is considered here waste should instead be considered by-product, and vice-versa. We are currently looking to refine these coefficients.

The food groups considered are sugar beet, cereals, fruit, vegetables, potatoes, oilseeds, meat, fish, eggs, and dairy. The study system boundaries include post-farm gate stages of the food chain, i.e. processing, distribution, and consumption, and human metabolism (intended as the processes of human digestion of food and excretion of residues) as well as waste management.

In the following sections, we describe briefly how we calculated the nutrient content in food products, by-products, food waste, and human excrements (section 1) and the nutrient losses to the environment from the different destinations (section 2). For more details on the model the reader is referred to the paper Corrado et al. (2020).

Figure A2.90. Boundaries of the study, flows and destinations considered. Adapted from Corrado et al. (2020).



Source: Corrado et al. (2020).

PART 1. Nutrient content in food products, food waste, and human excrements

Quantities of consumed food, food waste, and by-products at each stage of the EU food chain for each MS were obtained from the food waste model described in Caldeira et al. (2021). For the purpose of this project the outcome of the food waste model for animal food waste and by-product at processing only considers the meal fraction of the flows, being nutrients (N and P) almost missing in the remaining fats. Meal and fat fractions per animal and waste categories were derived from Ferronato et al. (2021). The calculation of the nutrient content in food products and food waste is described below separately for N and P. Regarding human excrement it was considered that 97% of N and P in ingested food is present in human excrements, the remaining 3% is excreted via nails, hair, etc. (Vigiak et al, 2020).

A2.1.1 Nitrogen content in food products and food waste

The calculation of the nitrogen content in food products was done according to Corrado et al. (2020) using the following data:

i. Crude protein content in food products and food waste and by-products

For the food products the protein content (kgCRPR/kg product, CRPR – Crude Protein) was derived from the CAPRI model (Britz & Witzke, 2014), with exception for oilcrop cakes, which was obtained from Feedipedia (2013), and fish that was obtained from FAO/INFOODS Global food composition database – version 1.0 (uFiSh1.0) – 2016) (FAO, 2016).

Regarding the food waste and by-products, as its composition is in some cases different from that of the food products (food waste is typically composed of a higher share of inedible parts of food, e.g. peels), additional sources were consulted to obtain the protein content of the inedible parts of food. For instance, the USDA food composition database was used to retrieve protein content in fruit peels (USDA, 2018). Moreover, as waste composition varies across different food supply chain stages, the waste composition presented in table A2.1 was assumed.

Table A2.1. Food waste composition assumed per food group along the stages of the food chain.

Food group	**Stage of the food chain**			
	Production	**Processing**	**Distribution**	**Consumption**
Meat	Non-consumptive parts from slaughterhouses	Non-consumptive parts of rendering processes	Slaughterhouse products used for human consumption	
Fish	We assume same N content in all stages			
Dairy	Produced cow and sheep milk	Milk derivatives used for feed and in industrial processes	Losses of milk derivatives in the market	Milk derivatives used for human consumption
Eggs	Whole egg	Shell	26% egg without shell + 74% whole egg	Shell
Cereals	Raw cereal	Bran	Cereal after processing	
Fruits	Whole fruit	Peels	80% whole fruit and 20% juice (without peel)	Peels
Vegetables	We assume same N content in all stages			
Potatoes	We assume same N content in all stages			
Sugarbeet	Sugarbeet	Molasse	Processed sugar	Processed sugar
Oilcrops	Olives, sunflower, rapeseed, soya and other oil crops	Sunflower, rapeseed and soya cakes	Olive, sunflower, rapeseed and soya oils	Olive, sunflower, rapeseed and soya oils

ii. Conversion factors protein content / nitrogen content - Jones' factors

The Jones' factors (in kgCRPR/kgN) allow the conversion of nitrogen content to protein content and are available for a series of foodstuffs, including a standard default conversion factor (Jones, 1941). The values used are presented in Table A2.2.

Table A2.2. Jones' factors per food group.

Food group	Jones' factor (kgCRPR/kgN)
Meat	6.25
Fish	6.25
Dairy	6.38
Eggs	6.25
Cereals	5.83
Fruits	6.25
Vegetables	6.25
Potatoes	6.25
Sugarbeet	6.25
Oilcrops	6.25
Default value	6.25

Source: Jones (1941)

iii. Food consumption amounts

Amounts of food consumption was derived from CAPRI model (Britz & Witzke, 2014) when available: "Production" (balanced with net trade), "household consumption" and "lost in distribution" were used to properly represent the different stage of the food supply chain.

The nitrogen content is calculated as illustrated in figure A2.2 for vegetables at primary production. First, we calculate the N content for each food item as the ratio between protein contents and Jones' factors. And then, we calculate the N content for the food category as weighted average of protein contents per food items using consumption as weight.

Figure A2.2. Exemplification of the calculation of N content in vegetables.

Food item	Protein (kg/kg prod)	Jones' factor (kg protein/kg N)
Pulses	0.26	6.25
Tomato	0.009	6.25
Other vegetables	0.02	6.25

$$\frac{Protein\ content}{Jones'factor}$$

Food item	N content (kg N/kg prod)	Production (1000 tonnes)
Pulses	0.0416	750.04
Tomato	0.0014	17342.67
Other vegetables	0.0032	51640.36

$$\frac{\sum_{items} N\ content\ * Production}{\sum_{items} Production}$$

Food category	N content (kg N/kg prod)
Vegetables	0.0032

The N content in food products and food waste calculated is presented in table A2.3.

Table A2.3. N content in food products, food waste, and by-products based in Corrado et al. PP: Primary Production; P&M: Processing and Manufacturing; R&D: Retail and Distribution; Cons: Consumption (household and food service).

N content in food waste and by-products (g N/kg waste or by-product)										
	Meat	**Fish**	**Dairy**	**Eggs**	**Cereal**	**Fruits**	**Vegetable**	**Potato**	**Sugar**	**Oilcrop**
PP	23.8	33.1	5.2	19.2	18.8	1.2	3.2	3.4	2.3	37.5
P&M	36.7	33.1	5.6	20.9	27.4	2.0	3.3	3.4	9.1	52.2
R&D	32.7	33.1	5.5	19.2	10.2	1.2	3.6	5.0	2.2	34.1
Cons	32.7	33.1	10.2	20.9	10.2	1.7	3.5	5.0	1.6	3.6
N content in food product (g N/kg product)										
PP	25.9	33.1	5.2	19.2	18.8	1.2	3.2	3.4	2.3	37.5
P&M	32.7	33.1	10.2	19.2	10.2	0.9	3.5	5.0	1.6	3.6
R&D	32.7	33.1	10.2	19.2	10.2	0.9	3.5	5.0	1.6	3.6
Cons	32.7	33.1	10.2	19.1	10.2	0.8	3.5	5.0	1.6	3.6

An alternative source of information on the N content in food products, the FAO INFOOD (FAO, 2016), was considered to assess the values obtained. The N content in food products based in FAO INFOOD were calculated as:

$$N\ content\ FAO\ INFOOD\ [kg\frac{N}{kg}prod] = \frac{Protein\ [g]}{Jones'\ Factor * 100}$$

A factor of 100 was applied as all values reported in FAO INFOOD refers to 100g of edible portion of fresh weight. The average of all values available for the same food categories was considered. Correspondent classes for Sugarbeet and oilcrops were missing, their protein content was retrieved from (DietGrail, 2021a)

The values calculated from FAO INFOOD for food products are presented in table A2.4. Differences across the supply chain stages and between products and waste cannot be captured using this source. We therefore use it to assess the values calculated for food products at consumption.

Table A2.4. N content in Food Product based on FAO INFOOD (g N/kg product)calculated from FAO INFOOD.

	Meat	**Fish**	**Dairy**	**Eggs**	**Cereal**	**Fruits**	**Vegetable**	**Potato**	**Sugar**	**Oilcrop**
Cons	34.3	30.5	5.0	20.2	10.7	3.2	1.3	4.0	0.3	2.6

Source: FAO INFOOD

A2.1.2 Phosphorus content in food products and food waste

The phosphorus content in food products, food waste, and by-products was calculated following a similar approach as for nitrogen, in order to ensure consistency with nitrogen flows analysis and reflect the variability in product and waste composition at the different supply chain stages. The crude protein content in food products and food waste (i) and the food consumption amounts (iii) were the same amounts used in the N calculations. Instead, the conversion factors protein content/ phosphorous content (equivalent to Jones' Factors for Nitrogen) (ii) were obtained from (USDA, 2018) at food item level (Table A2.5).

Table A2.5. Conversion factors protein content/ phosphorous content (kgCRPR/kgP)

Food group	Food item	kgCRPR/kgP
Fish	Average fish and shellfish	87.733
Dairy	Milk	37.500
	Sheep and goat milk	35.465
	Butter	35.417
	Cheese	54.388
	Concentrated milk	31.265
	Cream	33.750
	Fresh milk products	36.440
	Raw milk	37.500(1)
	Skimmed milk powder	37.355
	Whey powder	10.811
	Casein	10.811(2)
	Whole milk powder only	33.918
Eggs	Eggs	63.434
Cereals	Soft wheat	26.592
	Durum wheat	26.929
	Rye	31.145
	Barley	47.273
	Oats	32.294
	Grain maize	44.857
	Rice	23.844
	Other cereals	37.143
	Rice milled	30.299
Fruits	Citrus	47.273
	Apples, pears and peaches	35.581
	Other fruits	47.308
	Table grapes	34.773
	Wine	3.500
Vegetables	Pulses	69.328
	Tomato	36.667
	Other vegetables	37.248
Potatoes	Potato	67.632
	Starch	20.000
Sugarbeet	Sugarbeet	30.000(3)
	Processed sugar	30.000
	Molasse	30.000(3)
Oilcrops	Table olives	280.00
	Olives for oil	280.00
	Sunflower	31.485
	Rape	31.498
	Soya	51.832
	Sunflower seed cake	31.485(4)
	Rapeseed cake	31.498(5)
	Soya cake	51.832(6)
	Olive oil	27.753

(1) Missing in USDA, used the value of cow milk
(2) Missing in USDA, used the value of whey
(3) Missing in USDA, used the value of processed sugar
(4) Missing in USDA, used the value of sunflower
(5) Missing in USDA, used the value of rape
(6) Missing in USDA, used the value of soya

Source: USDA (2018)

The P content was calculated in the same way as for N for the following food groups: dairy, eggs, fruits, vegetables, potatoes, sugarbeet and oilcrops. Instead, P content in fish was derived from FAO/INFOOD (FAO, 2016) and it was assumed no difference between the food product and the food waste (table 1) as well as among supply chain stages. Protein content of cereals for human consumption was available with specific values for whole and bran cereals. The former was used for the food product and the later for food waste.

A different approach was used for meat, where data and the derivation principle reflect an adaptation of the work by van Dijk et al. (2016), allowing to quantify the P content in meat products (generally muscles), by-products (at processing only meal fraction was considered, calculated as 0% of category 1, 100% of category 2 and for category 3 based on Ferronato et al. (2021) and waste (generally bones). P content in live animal (Kemme et al, 2004; Jongbloed, 2010; Kemme et al, 2005; Bikker et al, 2014) and P content in meat products (from Danish food content database) (Frida, 2019) were obtained from literature and food composition databases, as in the work of van Dijk et al. (2016). The procedure for deriving the P content in waste and by-products (P_{WBP}) at processing is described below and illustrated in Figure 3, using beef as example. In details, the steps followed were:

— <u>Step 1:</u> the amount of P (in mass) in live weight (P_{LW}) was derived multiplying literature nutrient content data by an initial mass of 1000 kg of animal. In the figure it corresponds to the amount 7.19 kg P.

— <u>Step 2</u>: the share of animal by-products (ABP) was calculated from the percentages of category 1, 2 and 3 (Ferronato et al., 2021) while the share of bone-free meat (BFM) as it's complementary. These values were used to derive the mass of ABP and BFM. In figure A2.3 the example shows 1000 kg of Life weigh originates 650 Kg of ABP and 350 kg of BFM.

— <u>Step 3</u>: the amount of P in BFM (P_{BFM}) was calculated multiplying literature nutrient content data by the mass of BFM. In the figure it corresponds to the amount 0.6 kg P

— <u>Step 4</u>: the Phosphorus mass in ABP (P_{ABP}) was calculated as difference between the P_{LW} and P_{BFM}. In the figure it corresponds to the amount 6.59 kg P.

— Last, <u>step 5</u>: P content in waste and by-products (P_{WBP}) was estimated as the ratio between the amount of Phosphorus in ABP and the ABP mass. In the figure it corresponds to the amount 0.01015 kg P.

Figure A2.3. Steps taken for the estimation of the amount of phosphorus in waste and by-products, exemplified for beef. Literature data sources: a: Kemme et al. (2004); b: Frida (2019); c: Ferronato et al. (2021).

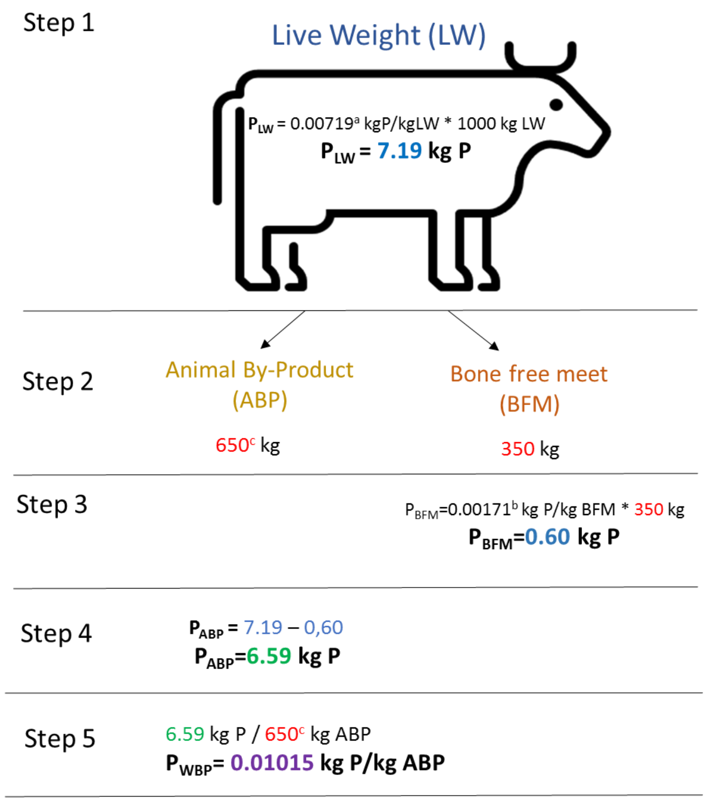


Table A2.6 presents the P content in food waste, by-products and food products calculated.

Table A2.6. P content in food products and food waste. PP: Primary Production; P&M: Processing and Manufacturing; R&D: Retail and Distribution; Cons: Consumption (household and food service).

P content in food waste and by-products (g P/kg waste or by-product)										
	Meat	**Fish**	**Dairy**	**Eggs**	**Cereal**	**Fruits**	**Vegetable**	**Potato**	**Sugar**	**Oilcrop**
PP	6.0	2.4	0.9	1.9	e	e	0.5	0.3	0.5	5.6
P&M	8.9	2.4	1.1	2.1	3.9	0.3	0.5	0.3	1.9	7.5
R&D	1.7	2.4	1.0	1.9	2.0	0.2	0.6	0.5	0.4	5.3
Cons	1.7	2.4	1.6	2.1	2.0	0.3	0.5	0.5	0.3	0.7
P content in food product (g P/kg product)										
PP	6.0	2.4	0.9	1.9	5.9	0.2	0.5	0.3	0.5	5.6
P&M	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7
R&D	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7
Cons	1.7	2.4	1.6	1.9	2.0	0.2	0.5	0.5	0.3	0.7

Similarly to what was done for N, an alternative source of information on P content from FAO INFOODS (FAO, 2016), was used to assess the values obtained. The P content in food products based in FAO INFOOD was calculated as:

$$P\ content\ FAO\ INFOOD[kg\ P/kg\ prod] = P\ [mg]/100000.$$

The 100000 factor is used for unit of measure conversion. Correspondent classes for sugarbeet and oilcrops were missing, their protein content was retrieved from (DietGrail, 2021b).

The values calculated from FAO INFOOD for P content in food products are presented in table A2.7. Differences across the supply chain stages and between products and waste cannot be captured using this source. We therefore use it to assess the values calculated for food products at consumption.

Table A2.7. P content in Food Product based on FAO INFOOD (g P/kg product).

	Meat	Fish	Dairy	Eggs	Cereal	Fruits	Vegetable	Potato	Sugar	Oilcrop
Cons	1.9	2.3	0.7	4.5	4.2	0.6	0.2	0.6	0.4	2.4

PART 2. Quantification of nutrient losses to the environment

As illustrated in figure A2.1, the nutrient losses to the environment were calculated from the nutrient losses occurring in the different destinations of the food waste (landfill, incineration, anaerobic digestion, composting and wastewater treatment), and human excrements (wastewater treatment, not treated and not collected). Nutrient content in sludge from wastewater treatment that are then directed to landfill and incineration, from where additional losses occur, are also considered.

The destinations for food waste and human excrements are those considered in Corrado et al. (2020). However, instead of EU shares for each destination, coefficients for share of destination for waste and sludge were updated using country-specific values for each MS and for each year. Coefficients were retrieved for waste at each stage of the supply chain, following the same calculation procedure from Corrado et al. (2020). Data for MSs and years were downloaded through an R API from Eurostat (env_wasgen, env_wastrt) (Eurostat, 2021a; Eurostat, 2021b) and from EEA (European Environmental Agency) (EEA, 2021). Missing data from Eurostat and EEA were replaced with the value from the last available year. This approach was preferred in order to valorise existing data, instead of using interpolations.

A2.2.1 Nitrogen losses to the environment

We quantified N emissions to the environment from the different waste treatments. Emissions of N can occur as non-reactive, i.e. molecular N (N2), not harmful for the environment, or reactive N (Nr), including nitrates, ammonia, and nitrogen oxides, responsible for various pollution phenomena. The remaining N, i.e. the share that is neither emitted as Nr nor as N2, remains in the food waste or in the output of the waste treatments. This flow can be valorised as input either to the food chain or to other productive chains. The quantification of N emissions was done according to Corrado et al. (2020) as follow:

— <u>N emissions from composting</u>: average reactive N factors from composting were assumed as 23.2% of initial N for ammonia and N2O (being the later a minor share of about 0.5-1%) and 0.3% for N leaching (median values reported by Körner, 2009). The same N emission factors were considered for the case in which anaerobic digestion is followed by composting. For home composting the shares assumed were 59.5% N_2 and 5.2 Nr (Andersen et al 2011). It was assumed that N captured in ventilation air purification systems is not recycled.

— <u>N emissions from incineration</u>: a share of 30% emitted as Nr and 70% emitted as N2 was assumed, calculated as weighted average of the emissions from the two DeNOx technologies (selective non-catalytic reduction and selective catalytic reduction) used to remove Nitrogen oxides (NOx) from the flue gas thanks. This information was taken from Corrado et al. (2020) based on information provided in the Doka Tool for calculation of emissions from incineration (Doka G., 2003). A share of 2 % of N was considered to be present in the residues (i.e. ash) from the incineration and assumed to be disposed in landfills.

— <u>N emissions from landfills:</u> N losses from landfills are highly variable and depend on different elements, such as the type of waste and the climatic conditions. Bacterial activity and related N emissions vary during the lifetime of the landfill. The first phase is characterised by aerobic conditions and N emissions assumed for compost were considered. The following phases, instead, are characterised by a lack of oxygen. Emissions of N in these stages were modelled as according information reported by Cossu and Raga (2005) and Brandstätter et al. (2015). Emissions from landfills are 47.4% of available N, thereof 39.2% as N2 and 8.2% as Nr, whereas the remaining 52.6% stays in the body of the landfill.

— N emissions from incinerated or landfilled sewage sludge and ash: the N emission factors abovementioned for incineration and landfill were used.

— N emissions from wastewater treatment: for primary and secondary treatment we assumed nitrogen removal efficiencies of 10% and 25%, respectively (Bonomo, 2008). For tertiary wastewater treatment technologies it was considered 30% of N emitted as Nr, and 51% emitted as N_2 (McCarty, 2018). For all the treatment, the share of N not emitted to the environment stays in the wastewater sludge.

A2.2.2 Phosphorous losses to the environment

Regarding P losses from waste treatments, contrary to N emissions, there is no distinction between reactive and non-reactive forms. Due to phosphorous characteristics, no significant losses to air take place. The emissions were assumed as follows:

— P emissions from composting: composting is subject to nutrient losses as not all is recirculated on agricultural land. Compost is used also in and in other non-agricultural applications, such as backfilling and mine sites rehabilitation and landfill (as cover material). On average, 40% of the phosphorous in compost is lost (Barth, 2006; EC, 2008; van Dijk et al., 2016). During the anaerobic digestion process and when handling its residues, almost no P losses occurs. Therefore, it was assumed a 1% P loss. It is mostly directly applied in agriculture, being a wet waste. When residues are destined to composting, it is assumed that the same share of P losses take place.

— P emissions from incineration: all P in waste incinerated and landfilled is assumed to be removed from the biogeochemical cycle. Ashes are mostly used as construction materials, because technologies that recover P from (sewage sludge) ashes (Tonini et al., 2019) are not yet being implemented at a relevant scale.

— P emissions from landfills: Current technologies do not enable the return of landfilled P into the biogeochemical cycle. Therefore, a 100% loss is assumed for both treatment destinations.

— P emissions from wastewater treatment: P losses from wastewater treatment were calculated as the complementary of the removal efficiency, as reported in Vigiak et al. (2020). Wastewater collected but not treated was assumed to be destined to septic tanks, which can remove 30% of P from the water. The same efficiency is reported for primary wastewater treatment plants. Secondary and tertiary treatments guarantee higher performances, by removing 60% of P. Removal efficiency increases up to 90% whether tertiary treatment plants are equipped with specific P-removal systems. The share of tertiary treatment plants with this technology per each MS was retrieved from elaboration of data available in UWWTD database (EEA, 2017).

Table A2.8 presents a summary of the coefficients for N and P losses at each destination used in the model.

Table A2.8. Summary of coefficients for nutrient losses to the environment in the different destinations.

	Share of nitrogen emissions (%)			Share of phosphorous emissions (%)	
Waste destination	**N reactive**	**N_2**	**Source**	**P**	**Source**
Composting + direct application	23.5	1.9	Korner (2009)	40	(Barth, 2006; EC, 2008; van Dijk et al., 2016).
Anaerobic digestion + composting	23.5	1.9	Assumed as composting + direct application	40	(Barth, 2006; EC, 2008; van Dijk et al., 2016).
Anaerobic digestion + direct application	0	0	Corrado et al. (2020)	1	Assumption
Other uses (e.g. pet	5.2	59.5	Adersen et al. (2011)	59.5	Assumed as N

feeding, home compost)					
Landfill	8.2	39.2	Corrado et al. (2020) modelled as: Cossu and Raga (2005) and Brandstätter et al. (2015).	100	assumption
Incineration	30.4	69.5	Corrado et al. (2020), assumption by the authors based on Doka G. (2003)	100	Tonini et al. (2019)
Wastewater, collected without treatment	100	0	Corrado et al. (2020), assumption by the authors	70	Vigiak et al. (2020)
Wastewater, primary treatment	90	0	Bonomo (2008)	70	Vigiak et al. (2020)
Wastewater, secondary treatment	75	0	Bonomo (2008)	40	Vigiak et al. (2020)
e	30	51.3	McCarty (2018)	40	Vigiak et al. (2020)
Wastewater, tertiary treatment with P removal	Not applicable			1	Vigiak et al. (2020)
Wastewater, non-collected	100	0	Corrado et al. (2020), assumption by the authors	100	Vigiak et al. (2020

References of Annex 2

Andersen,J.K., Boldrin,A., Christensen,T.H., Scheutz,C., 2011. Mass balances and life cycle inventory of home composting of organic waste. Waste Manag31 (9–10), 1934–1942.https://doi.org/10.1016/j.wasman.2011.05.004.

Barth J., ECN & INFORMA (2006) Marketing Strategies and Quality Requirements in Compost Applications. ISWA Beacon Conference, BTW, 2006, Perugia. Available at: https://www.iswa.org/uploads/tx_iswaknowledgebase/btw26.pdf

Bonomo, 2008. Trattamento Delle Acque Reflue. Mc-Graw Hill.

Brandstätter, C., Laner, D., Fellner, J., 2015. Carbon pools and flows during lab-scale degradation of old landfilled waste under different oxygen and water regimes. Waste Manag. 40, 100–111.

Bikker P., van Krimpen M.M., Jongbloed A.W., (2014). Actualisatie N- en P-excretieforfaits leghennen en eieren ; diercategorieën Ki2, Ki3, Ki4, Ki5

Britz, W., Witzke, H.P., 2014. CAPRI Model Documentation 2014. Institute for Food and Resource Economics, University of Bonn, Bonn Available online: http://www.caprimodel.org/docs/capri_documentation.pdf Accessed in May 2019

Caldeira, C., De Laurentiis, V., Ghose, A., Corrado, S., & Sala, S. (2021). Grown and thrown: Exploring approaches to estimate food waste in EU countries. Resources, Conservation and Recycling, 168(February). https://doi.org/10.1016/j.resconrec.2021.105426

Corrado, S., Caldeira, C., Carmona-Garcia, G., Körner, I., Leip, A., & Sala, S. (2020). Unveiling the potential for an efficient use of nitrogen along the food supply and consumption chain. Global Food Security, 25(February), 100368. https://doi.org/10.1016/j.gfs.2020.100368

Doka G. (2003) Life Cycle Inventories of Waste Treatment Services. ecoinvent report No. 13, Swiss Centre for Life Cycle Inventories, Dübendorf, December 2003.

Jones, D. B. (1941). Factors for converting percentages of nitrogen in foods andfeeds into percentages of protein. US Department of Agriculture-circ. 183.Washington, DC

Cossu, R., Raga, R., Vettorazzi, G., 2005. Carbon and nitrogen mass balance in some landfill models for sustainability assessment. In: "Proceedings Sardinia 2005".Tenth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, Italy, 3-7 October.

DietGrail, 2021a. http://dietgrail.com/protein/. Accessed April 2021.

DietGrail, 2021b. http://dietgrail.com/phosphorus/. Accessed April 2021.

European Environment Agency (EEA), 2019. Urban waste water treatment. https://www.eea.europa.eu/data-and-maps/indicators/urban-waste-watertreatment/urban-waste-water-treatment-assessment4. Accessed in June 2021.

European Environment Agency (EEA), 2017. Waterbase - UWWTD: Urban Waste Water Treatment Directive – reported data. v6, https://www.eea.europa.eu/data-and-maps/data/waterbase-uwwtd-urban-waste-water-treatment-directive-5, Permalink:8396aa5079544dab9d8b8966237a9f3b.

European Commission (2008) Green paper on the management of bio-waste in the European Union. SEC(2008) 2936

Eurostat, 2021a. Generation of waste by waste category, hazardousness and NACE Rev. 2 activity (env_wasgen). https://ec.europa.eu/urostat/data/database Accessed in June 2021.

Eurostat, 2021b. Treatment of waste by waste category, hazardousness and waste management operations (env_wastrt). https://ec.europa.eu/eurostat/data/database. Accessed in June 2021.

Feedipedia. Feedipedia: on-line encyclopedia of animal feeds: Institut National de la Recherche Agronomique (French National Institute for Agricultural Research, INRA), the Centre de Coopération Internationale en Recherche Agronomique pour le Développement (French Agricultural Research Center for International Development, CIRAD), Association Française de Zootechnie (French Association for Animal Production, AFZ) and the Food and Agriculture Organization of the United Nations (FAO). 2013. http://www.feedipedia.org

Ferronato, G., Corrado, S., De Laurentiis, V., & Sala, S. 2021. The Italian meat production and consumption system assessed combining material flow analysis and life cycle assessment. Journal of Cleaner Production, 321, 128705, Food and Agriculture Organization of the United Nations (2016). FAO/INFOODS Global Food Composition Database for Fish and Shellfish Version 1.0 – uFiSh1.0. Rome, Italy.

Food data (frida.fooddata.dk), version 4, 2019, National Food Institute, Technical University of Denmark

Jongbloed AW. Gehalte en uitscheiding van fosfor en stikstof door opfokzeugen van ongeveer 130 kg = Contents and excretion of phosphorus and nitrogen by gilts of approximately 130 kg. Lelystad: Wageningen UR Livestock Research, 2010.

Kemme PA, van Diepen JTM, van der Togt PL, Jongbloed AW. De gehalten aan stikstof, fosfor en kalium in blanke vleeskalveren. ID – Voeding, Lelystad, 2004.

Kemme P, Heeres-van der Tol J, Smolders G, Valk H, van der Klis JD. Schatting van de uitscheiding van stikstof en fosfor door diverse categorieën graasdieren. Wageningen UR, Animal Science Group, Wageningen, 2005.

Körner, I., Stegmann, R., Hrsg, 2009. Stickstoffhaushalt bei der Kompostierung: Bilanzen, Gehalte, Umsetzungs- und Austragsprozesse. Habilitationsschrift, Technische Universität Hamburg-Harburg. In: Hamburger Berichte 33. Verlag Abfall aktuell, Stuttgart, Deutschland ISBN: 978-3-9812867-0-0, 320 Seiten. https://d-nb.info/104845438X/34.

Mc Carty, P.L., 2018. What is the best biological process for nitrogen removal: when and why? Environ. Sci. Technol. 52 3855-3841.

Tonini D., Saveyn H. & Huygens D. (2019) Environmental and health co-benefits for advanced phosphorus recovery. Nature Sustainability 2: 1051-1061.

USDA, 2018. Food Composition Database, v.3.9.5.1_2018-09-23. Available online. https://ndb.nal.usda.gov/ndb/ Accessed May 2019.

Vigiak, O., Grizzetti, B., Zanni, M. et al. Domestic waste emissions to European waters in the 2010s. Sci Data 7, 33 (2020). https://doi.org/10.1038/s41597-020-0367-0

van Dijk, K. C., Lesschen, J. P., & Oenema, O. (2016). Phosphorus flows and balances of the European Union Member States. Science of the Total Environment, 542, 1078–1093. https://doi.org/10.1016/j.scitotenv.2015.08.048

Annex 3. National Nitrogen Budgets (CLRTAP)

Note of EPNB Co-Chairs Markus Geupel and Wilfried Winiwarter for the INMAP-Process

Background

Both the EU Biodiversity Strategy to 2030 and the Farm-to-Fork-Strategy, launched in 2020 by the European Commission, set ambitious targets regarding nitrogen emissions. The goal is to halve nitrogen losses to the environment, conceptually also covering losses induced by application of fertilizers, while fully maintaining soil fertility. This will result in the reduction of fertilizer use by at least 20%. To reach this goal, in 2022 the European Commission will develop with Member States an Integrated Nutrient Management Action Plan (INMAP), to better manage nitrogen throughout its life cycle. The JRC situated in Ispra collects methods and approaches to be included into the INMAP. In this context Bruna Grizzetti (JRC Ispra) reached out to the Co-Chairs of the Expert-Panel on Nitrogen Budgets of the Task Force on Reactive Nitrogen (operating under CLRTAP), Wilfried Winiwarter (IIASA) and Markus Geupel (UBA). The Expert Panel Co-Chairs agreed, to summarize the methodological approach of a “Guidance document on national nitrogen budgets”, an excellent tool to inventory all relevant nitrogen flows within different segments of a national economy and its environmental pools, also considering cross- border transport. Quantitative knowledge on flows help to identify sources and assess developments. The methodology also offers opportunities for benchmarking interim milestones towards the new goal of halving nitrogen losses.

National Nitrogen Budget (NNB)

Reactive Nitrogen is being emitted to the environment as different chemical compounds to various environmental compartments by a large variety of source sectors. Reactive nitrogen in the environment is highly mobile and easily available for chemical transformation, thus it can travel along the “nitrogen cascade” across environmental media. With a complex conversion/transport process, it gets challenging to trace the fate of any individual release. Mass balance considerations allow to postulate the conservation of the sum of all reactive nitrogen compounds, with specific consideration of sources and sinks. This can be implemented by collecting knowledge on all relevant reactive nitrogen emissions and flows across all sectors and environmental media, by creating a National Nitrogen Budget (NNB). Setting up an NNB using harmonized methodologies allows to assess standardized data and to compare data across borders.

The Expert Panel on Nitrogen Budgets, as part of the Task Force on Reactive Nitrogen under the Convention on Long-Range Transboundary Air Pollution developed such an integrated harmonized approach. It was designed to assist the calculation of nitrogen budgets, nitrogen use efficiency and nitrogen surpluses and their improvements within the geographical area of the CLRTAP. The technical guidance of the approach is published in a guidance document together with detailed annexes for each of the eight pools comprised in the national nitrogen budget. In 2012, the guidance document has been adopted by the Executive Body of the Convention to enable fulfillment of Article 7, paragraph 3 (d), of the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone, as amended in 2012[29].

Method

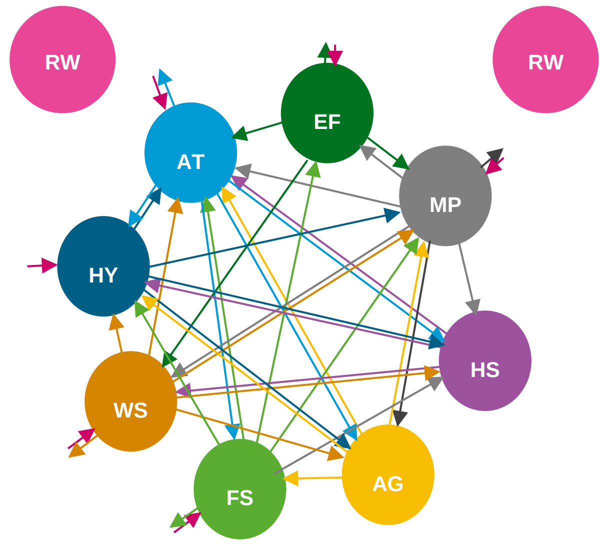
NNB uses a fully integrated approach, that means that it covers the whole biosphere and the whole economy and multiple, relevant sectors for nitrogen emissions or nitrogen handling. This is expressed in eight different pools, namely, a) - Energy and fuels, b) - Material and products in industry, c) - Humans and settlements, d) - Agriculture, e) - Forest and semi-natural vegetation including soils, f) - Waste management, g) - Atmosphere and h) - Hydrosphere. In the quantitative assessment, all different forms of reactive nitrogen, such as e.g. ammonia, nitrate, nitrogen oxide or nitrous oxide are normalized to their nitrogen amounts and all flows between compartments (pools and sub-pools) are provided as annual totals in units of mass N per year. The same is done for N contained in products, e.g. in agricultural commodities or chemicals. In its different forms, nitrogen is transported and exchanged

[29] ECE/EB.AIR/113/Add.1, decision 2012/10

between pools. The approach collects and displays country-internal flows but also allows to consider cross-border flows. The internal flows desccribe the exchange processes between the eight individual pools (Figure A3.1). Additionally, flows within each pool (between sub-pools) have been identified and can be quantified. Uncertainties should be handled according to the standards set by the Task Force on Emission Inventories and Projection. During 2022 a review of the methods and the development of a harmonized reporting template are planned.

Figure A3.1. Overview on Nitrogen Pools and Fluxes

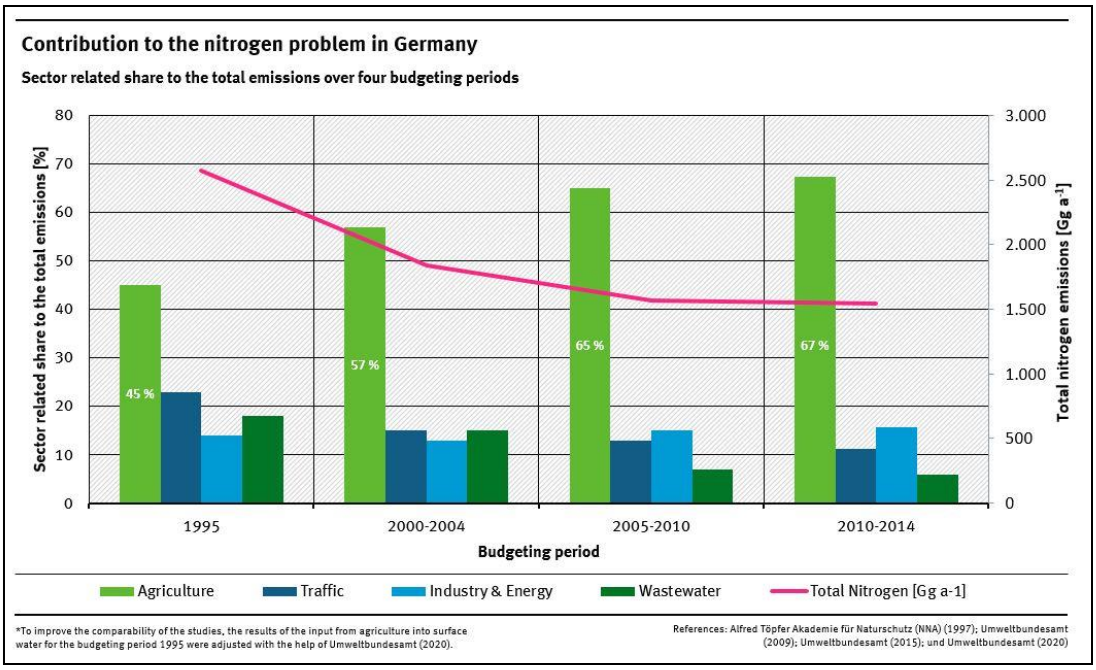
Pool-ID	Name of pool
AT	Atmosphere
EF	Energy and fuels
MP	Material and products in industry
HS	Humans and settlements
AG	Agriculture
FS	Forest and semi-natural vegetation
WS	Waste management
HY	Hydrosphere
RW	Rest of the world (cross-border)



Results and case-study Germany

While NNBs have been prepared for a number of countries, the majority of approaches were performed as a scientific or project-related task. No operative routine procedure of regularly repeated NNBs for an individual country has been established yet. The most comprehensive study with full application of the Guidance Document has been executed in Germany. A report has been published by the German Environment Agency, and a scientific publication is pending. The report shows the relevant sources, sinks and flows of the nitrogen cycle in Germany. A total of around 150 flows of ammonia, nitrogen oxides, nitrous oxide and nitrate between soil, plants, animals, food and feed, industrial products etc. have been quantified. The results are given as annual N flows for the years 2010-2014. Based on these quantities, strategies for reducing nitrogen emissions can be developed and the impact of measures can be assessed. Figure A3.2 shows comprehensive results of this report (right column group) in comparison to earlier, similar quantifications of the same entities, producing a trend over time.

Figure A3.2. Share of four aggregate sectors to the overall nitrogen emissions in Germany



Outlook

With a working example available, the network of NNB experts in place and the needs to observe N issues increasing, it is expected that other UNECE countries will start to apply NNBs in the near future. The approach targets as much as possible (in terms of methods, sectors/pools, and data use) other reporting requirements as those for greenhouse gases and air pollutants. Once established, the additional demand of NNBs to inventory agencies is considered manageable and certainly worthwhile if aiming to protect ecosystems as well as the human health from pollution hazards.

Annex 4. Example fiches developed for IMAP wiki website

Figure A4.1. Fiche 1: Example of a general fiche (3 pages): “Nitrogen inhibitors”.

GENERAL FICHE - NITRIFICATION INHIBITORS GENERAL

Data extracted in October 2020

Note to the reader: This *general fiche* summarises the environmental and climate impacts of the application of NITRIFICATION INHIBITORS (NI) in nitrogen fertilization practices (both mineral and organic) found in a systematic review of 16 synthesis research papers [1]. These papers were selected, according to the inclusion criteria reported in section 4.

As each synthesis research paper involves a number of individual papers ranging from 4 to 376, the assessment of impacts relies on a large number of results obtained mainly in field experiments (carried out in situations close to real farming environment), and sometimes in lab experiments or from model simulations.

1. DESCRIPTION OF THE FARMING PRACTICE

Description	Nitrification inhibitors are part of a broader category of fertilization techniques called enhanced-efficiency fertilizers. Nitrification inhibitors are substances that, coupled to fertilizers, delay the bacterial oxidation of NH_4^+ (ammonium) to NO_2^- (nitrite) for a certain period by suppressing the activity of *Nitrosomonas spp.* (nitritation, step one of nitrification), and therefore the formation of NO_3^- (nitrate). In this way, mineral nitrogen (N) is retained as ammonium, which is less prone to leaching than nitrate, and which cannot be lost to the atmosphere by denitrification. Therefore, nitrification inhibitors are combined with fertilizers in order to increase fertilizer use efficiency (Chaves et al., 2006) [2].
Key descriptors	• Nitrification inhibitors include different active substances. They can be coupled to the application of both organic and mineral nitrogen fertilizers (urea- and ammonium-based). They cannot be applied to nitrate-containing mineral fertilizers. • Nitrification inhibitors do not include other types of enhanced-efficiency fertilizers (EEFs), such as controlled-release fertilizers (urease inhibitors, polymer-coated mineral fertilizers, organic-mineral fertilizers, etc.). However, nitrification inhibitors can be used in combination with other EEFs. Such results are reported in a separate set of fiches.

2. DESCRIPTION OF THE IMPACTS OF THE FARMING PRACTICE ON ENVIRONMENT AND CLIMATE

We consider the impacts of nitrogen-fertilization coupled to nitrification inhibitors (NI), as compared to nitrogen-fertilization without NI.

The table below shows the number of synthesis papers reporting positive, negative or no effect, based on the statistical comparison of the intervention and the control. In addition, we include, if any, the number of systematic reviews reporting relevant results but without statistical test of the effects (uncertain). The numbers between parentheses indicate the number of synthesis papers with a quality score of at least 50%. Details on quality criteria can be found in the methodology section of this WIKI.

Out of 16 synthesis papers1, 6 included studies conducted in Europe and 14 have a quality score higher than 50%. Some synthesis papers reported more than one impact.

Impact		Positive	Negative	No effect	Uncertain
Decrease air pollutants emissions	NH_3	0	5 (5)	5 (5)	1 (0)
	NO	3 (3)	0	1 (1)	0
Decrease GHG emissions	N_2O	9 (8)	0	0	1 (0)
	CH_4	1 (1)	0	2 (2)	0
	CO_2	1 (1)	0	1 (1)	0
Decrease nutrient leaching and run-off		4 (4)	2 (2)	1 (1)	0
Increase plant nutrient uptake		5 (5)	0	4 (4)	0
Increase soil nutrients	NH_4^+	2 (2)	0	0	0
	NO_3^-	0	2 (2)	0	0
	All N-forms*	1 (1)	0	2 (2)	0
Increase crop yield		6 (6)	0	3 (3)	0

* All nitrogen (N) forms include dissolved inorganic nitrogen forms (NH_4^+, NO_3^-, NO_2^-) and organic N.

3. DESCRIPTION OF THE KEY FACTORS INFLUENCING THE SIZE OF THE EFFECT

Only the factors explicitly studied in the reviewed synthesis papers are reported below. Details regarding the factors can be found in the Summaries of the meta-analyses.

IMPACTS	FACTORS
Decrease air pollutants emissions	NI type (Ref 7, 9, 12); Type of fertilizer (Ref. 12); Total nitrogen content (Ref 7); Soil type (Ref 12); Soil pH (Ref 5, 7); N rate (Ref 7); Type of land use (Ref. 12).
Decrease GHG emissions	NI forms (Ref 7, 16); Crop type (Ref 5, 7); Fertiliser type (Ref 8); Fertilizer application timing (Ref 10); Type of land use (Ref 8, 16); Cereals type (Ref 10); Baseline N_2O emission (Ref 5, 16); Soil type (Ref 2, 8); Soil texture (Ref 5); Soil pH (Ref 5, 10); SOC (Ref 5).
Decrease nutrient leaching and run-off	SOC content (Ref 7); Fertilizer type (Ref 12); N forms (Ref 11); N application rate (Ref 7, 11); Soil texture (Ref 12); Type of land use (Ref 12).
Increase plant nutrient uptake	Crop type (Ref 6); NI type (Ref1, 5, 8, 15); Fertilizer type (Ref1); Fertilizer total N content (Ref 8); N application rate (Ref1); Soil texture (Ref1); SOM content (Ref1); Rainfall (Ref 5); Temperature (Ref 5); Soil pH (Ref 5); Water management (Ref 5); Type of land use (Ref 15).
Increase soil nutrients	NI type (Ref 1, 11, 12); Fertilizer type (Ref 12); N application rate (Ref 1, 11); N application timing (Ref 1); Soil texture (Ref 1, 12); Soil pH (Ref 1, 11); Type of land use (Ref 12).
Increase crop yield	Crop type (Ref 5, 8, 13, 14, 15); Rainfall (Ref 5); Fertilizer N-form (Ref 14); N application rate (Ref 8, 14); NI type (Ref 13, 15); Soil type/texture (Ref 15); Soil pH (Ref 14).

4. SYSTEMATIC REVIEW SEARCH STRATEGY

Keywords	TOPIC: ("nitr* inhibit*" OR "controlled-release fert*" OR "urease inhibit*" OR "enhanced-efficiency fert*") AND TOPIC: ("meta-analy*" OR "systematic* review*" OR "evidence map" OR "global synthesis" OR "evidence synthesis" OR "research synthesis")
Search dates	No time restrictions
Databases	Web of Science and Scopus, run on 6 October 2020
Selection criteria	Three main criteria led to the exclusion of a synthesis paper: (1) the paper does not deal with nitrification inhibitors; (2) the paper does not assess the impacts of fertilization using nitrification inhibitors in comparison to another fertilization technique; (3) the paper is neither a meta-analysis nor a systematic review. Synthesis papers that passed the relevance criteria were subject to critical appraisal carried out on paper-by-paper basis. From an initial number of 55 synthesis papers, we finally selected nine meta-analysis.

5. LIST OF SYNTHESIS PAPERS INCLUDED IN THE REVIEW

Ref. num.	Authors	Year	Article Title	Source Title	DOI
1	Sha, ZP; Ma, X; Wang, JX; Lv, TT; Li, QQ; Misselbrook, T; Liu, XJ	2020	Effect of N stabilizers on fertilizer-N fate in the soil-crop system: A meta-analysis	Agriculture, Ecosystems and Environment, 290, 106763	10.1016/j.agee.2019.106763
2	Mazzetto, AM; Styles, D; Gibbons, J; Arndt, C; Misselbrook, T; Chadwick, D	2020	Region-specific emission factors for Brazil increase the estimate of nitrous oxide emissions from nitrogen fertiliser application by 21%	Atmospheric Environment, 230, 117506	10.1016/j.atmosenv.2020.117506
3	Ti, CP; Xia, LL; Chang, SX; Yan, XY	2019	Potential for mitigating global agricultural ammonia emission: A meta-analysis	Environmental Pollution, 245, 141-148	10.1016/j.envpol.2018.10.124
4	Gao, WL; Man, XM	2017	Evaluation of the Agronomic Impacts on Yield-Scaled N_2O Emission from Wheat and Maize Fields in China	Sustainability, 9, 1201	10.3390/su9071201
5	Li, T; Zhang, W; Yin, J, Chadwick, D; Norse, D; Lu, Y; Liu, X; Chen, X; Zhang, F; Powlson, D; Dou, Z	2017	Enhanced-efficiency fertilizers are not a panacea for resolving the nitrogen problem	Glob Change Biol. 2018;24:e511–e521.	10.1111/gcb.13918
6	Liu, SW; Lin, F; Wu, S; Ji, C; Sun, Y; Jin, YG; Li, SQ; Li, ZF; Zou, JW	2017	A meta-analysis of fertilizer-induced soil NO and combined NO+N_2O emissions	Global Change Biology, 23, 2520–2532	10.1111/gcb.13485

5. LIST OF SYNTHESIS PAPERS INCLUDED IN THE REVIEW

7	Xia, LL; Lam, SK; Chen, DL; Wang, JY; Tang, Q; Yan, XY	2017	Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis	Global Change Biology, 23, 1917–1925	10.1111/gcb.13455
8	Gilsanz, C; Baez, D; Misselbrook, TH; Dhanoa, MS; Cardenas, LM	2016	Development of emission factors and efficiency of two nitrification inhibitors, DCD and DMPP	Agriculture, Ecosystem & Environment, 216, 1-8	10.1016/j.agee.2015.09.030
9	Pan, BB; Lam, SK; Mosier, A; Luo, YQ; Chen, DL	2016	Ammonia volatilization from synthetic fertilizers and its mitigation strategies: A global synthesis	Agriculture, Ecosystems and Environment, 232, 283-289	10.1016/j.agee.2016.08.019
10	Thapa, R; Chatterjee, A; Awale, R; McGranahan, DA; Daigh, A	2016	Effect of Enhanced Efficiency Fertilizers on Nitrous Oxide Emissions and Crop Yields: A Meta-analysis	Soil Sci. Soc. Am. J. 80: 1121-1134	10.2136/sssaj2016.06.0179
11	Yang, M; Fang, YT; Sun, D; Shi, YL	2016	Efficiency of two nitrification inhibitors (dicyandiamide and 3, 4-dimethypyrazole phosphate) on soil nitrogen transformations and plant productivity: a meta-analysis	Scientific Reports, 6	10.1038/srep22075
12	Qiao, C.; Liu, L.; Hu, S.; Compton, JA.; Greaver, TL.; Li, Q.	2015	How inhibiting nitrification affects nitrogen cycle and reduces environmental impacts of anthropogenic nitrogen input	Global Change Biology, 21, 1249–1257	10.1111/gcb.12802
13	Hu, Y; Schraml, M; von Tucher, S; Li, F; Schmidhalter, U	2013	Influence of nitrification inhibitors on yields of arable crops: A meta-analysis of recent studies in Germany	International Journal of Plant Production 8 (1), 33-50	10.22069/IJPP.2014.1371
14	Kim, DG; Saggar, S; Roudier, P	2012	The effect of nitrification inhibitors on soil ammonia emissions in nitrogen managed soils: a meta-analysis	Nutr Cycl Agroecosyst 93:51-64	10.1007/s10705-012-9498-9
15	Linquist, BA; Adviento-Borbe, MA; Pittelkow, CM; van Kessel, C; van Groenigen, KJ	2012	Fertilizer management practices and greenhouse gas emissions from rice systems: A quantitative review and analysis	Field Crops Research, 135, 10–21	10.1016/j.fcr.2012.06.007
16	Akiyama, H; Yan, XY; Yagi, K	2010	Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N_2O and NO emissions from agricultural soils: meta-analysis	Global Change Biology, 16, 1837–1846	10.1111/j.1365-2486.2009.02031.x

[1] Research synthesis papers include a formal meta-analysis (MA) or systematic reviews (SR) with some quantitative results. 2 Chaves, B., Opoku, A., De Neve, S., Boeckx, P., Van Cleemput, O., & Hofman, G. (2006). Influence of DCD and DMPP on soil N dynamics after incorporation of vegetable crop residues. Biology and Fertility of Soils, 43(1), 62-68.

[2] Chaves, B., Opoku, A., De Neve, S., Boeckx, P., Van Cleemput, O., & Hofman, G. (2006). Influence of DCD and DMPP on soil N dynamics after incorporation of vegetable crop residues. Biology and Fertility of Soils, 43(1), 62-68.

Figure A4.2. Fiche 2: Example of a Single impact fiche (2 pages) “The impact of Nitrogen inhibitors on Nitrogen leaching and run-off”.

SINGLE-IMPACT FICHE
NITRIFICATION INHIBITORS

IMPACT: NUTRIENT LEACHING AND RUN-OFF

Data extracted in October 2020

Note to the reader: This fiche summarises the impact of coupling nitrification inhibitors (NI) to nitrogen fertilization practices (both mineral and organic) on NUTRIENT LEACHING AND RUN-OFF. It is based on four peer-reviewed synthesis research papers[1], each of them including from 16 to 376 individual studies.

1. WEIGHT OF THE EVIDENCE

- CONSISTENCY OF THE IMPACT:
  All four synthesis papers report a positive effect - a decrease of nitrogen leaching/run-off of nitrate (NO_3^-) and dissolved inorganic nitrogen forms (DIN)*] - by the application of different types of NI coupled to nitrogen-fertilizers (either mineral or organic), *when NI are assessed all together* (**Table 1**). However, meta-analyses (MAs) report contrasting effects on NH_4^+ leaching/runoff, *when different types of NI are assessed separately* (**Table 2**). In particular, two MAs reported a negative effect for the widely used NI 3,4-dimethylpyrazole phosphate (DMPP) while the same MAs reported a positive effect (i.e., a decrease of NH_4^+ leaching) for the NI dicyandiamide (DCD). Another MA reported no effect of NI on NH_4^+ leaching.

  Out of the four synthesis papers, three reported studies conducted at global scale (including EU) and one in China.

Table 1. Summary of effects. The numbers between parentheses indicate the number of synthesis papers with a quality score of at least 50%. Details on quality criteria can be found in the next section.

Impact	Positive	Negative	No effect	Uncertain
Decrease nitrogen leaching and run-off (NO_3^-, NH_4^+, DIN*)	4 (4)	2 (2)	1 (1)	0

* DIN: Dissolved inorganic nitrogen forms (the sum of NH_4^+, NO_3^-, NO_2^-)

- QUALITY OF THE SYNTHESIS PAPERS: *The quality score summarises 16 criteria assessing the quality of three main aspects of the synthesis papers: 1) the literature search strategy and studies selection; 2) the statistical analysis; 3) the potential bias. Details on quality criteria can be found in the methodology section of this WIKI.*

2. IMPACTS

The main characteristics and results of the synthesis papers are summarized in **Table 2**. Summaries of the meta-analyses provide fuller information about the results reported in each synthesis paper, in particular about the modulation of effects by factors related to soil, climate and management practices.

Table 2. Main characteristics of the synthesis papers reporting impacts on nitrogen leaching and run-off. The references are ordered chronologically with the most recent publication date first.

Reference	Population	Geographical scale	No. individual studies	Intervention	Comparator	Conclusion	Quality score
Li, T; Zhang, W; Yin, J; Chadwick, D; Norse, D; Lu, Y; Liu, X; Chen, X; Zhang, F; Powlson, D; Dou, Z 2017	Grassland, dryland (wheat, maize, vegetables), paddy.	Global (including EU)	16	Fertilization (mineral or organic) with NI (DCD, DMPP, nitrapyrin, neem oil, piadin, chlorinated pyridine, calcium carbide), using the same N rate and N source as the comparator.	Fertilization without NI	Overall, NI amendment reduced NO_3- leaching. More specifically, DCD and DMPP application significantly reduced NO_3- leaching.	62%
Xia, LL; Lam, SK; Chen, DL; Wang, JY; Tang, Q; Yan, XY 2017	Rice, wheat, and corn cropping systems	China	376	Fertilization with NI (DCD, DMPP, and nitrapyrin), using the same N rate and N source than comparator.	Fertilization without NI	Overall, NI application decreased the N leaching and runoff (N-forms not specified). Among different types of NI, DCD and DMPP application decreased N leaching (results for N run-off not reported).	56%
Yang, M; Fang, YT; Sun, D; Shi, YL 2016	Cereals, forage, vegetables-industrial crops	Global (including EU)	81	Fertilization with NI (DCD and DMPP)	Fertilization without NI	DCD and DMPP application decreased NO_3- leaching, DIN leaching. NH_4+ leaching was increased by DMPP application, while it was decreased by DCD application.	62%
Qiao, C.; Liu, L.; Hu, S.; Compton, JA.; Greaver, TL.; Li, Q. 2015	Cropping systems and pastures	Global (including EU)	99	Fertilization (mineral or organic or mixture) with NI (DCD, DMPP, nitrapyrin, calcium carbide, and organic NI)	Fertilization without NI	Overall, NO_3- leaching and DIN leaching were reduced by NI use. NI application had no significant impact on NH_4+ leaching. Among NI types, DMPP and DCD application both decreased NO_3- leaching and DIN leaching. NH_4+ leaching was increased by DMPP application, while it was decreased by DCD	69%

Figure A4.3. Fiche 3: Example of a Summary fiche "The impact of Nitrogen inhibitors on Nitrogen leaching and run-off extracted from the meta study Li et al. 2017" (1 page).

Nitrification inhibitors and N leaching

Reference 4

Li, T; Zhang, W; Yin, J, Chadwick, D; Norse, D; Lu, Y; Liu, X; Chen, X; Zhang, F; Powlson, D; Dou, Z 2017 Enhanced-efficiency fertilizers are not a panacea for resolving the nitrogen problem Glob Change Biol. 2018;24:e511–e521. doi: 10.1111/gcb.13918

Background and objective

Enhanced-efficiency fertilizers (EEFs) (polymer-coated fertilizers, nitrification inhibitors, i.e. NI, urease inhibitors and double inhibitors, i.e. urease and nitrification inhibitors combined) have been developed to better synchronize fertilizer nitrogen (N) release with crop uptake, offering the potential for enhanced N use efficiency (NUE) and reduced losses. Can EEFs play a significant role in helping address the N management challenge? The present work aims at obtaining a global view on the productivity and N-loss reduction performance of EEFs through a holistic evaluation of all four EEF types in different cropping systems under wide ranging conditions. Here the results regarding the effect of NI on nitrate (NO3-) leaching are reported.

Search strategy and selection criteria

We searched for peer-reviewed publications between 1980 and 2016 on efficacy of EEFs via the Web of Science and Google Scholar using search terms of enhanced-efficiency fertilizer, polymer coated fertilizer, urease inhibitor, nitrification inhibitor, yield, NUE, nitrous oxide, ammonia, and nitrate leaching. 1)One or more of the efficacy indicators had to be reported; 2) data must have originated from field experiments (i.e. laboratory-based studies were excluded); 3) the experiments must have been replicated; 4) information needed to be provided regarding cropping systems, inhibitor type or the coating material of controlled release fertilizer; 5) management practices such as N fertilizer type, application rate, method, and placement, crop residue management (retention or removal), tillage, irrigation, and use of other agricultural chemicals had to be the same for the EEF treatment and control.

Data and analysis

The efficacy of a given EEF type was assessed by meta-analysis procedure. More than half of the datasets did not report standard error/deviation. To overcome this hurdle while maintaining a robust meta-analysis, we used the bootstrap resampling procedure (5000 iterations) to obtain the mean RR with a bias-corrected 95% confidence interval (CI) (Adams, Gurevitch, & Rosenberg, 1997). In addition, we calculated the heterogeneity in lnRR between all studies (QT), within-group (QW), and between-group (QB).

Number of papers	Population	Intervention	Comparator	Outcome	Quality score
16	Grassland, dryland (wheat, maize, vegetables), paddy.	Fertilization (mineral or organic) with NI (dicyandiamide, i.e. DCD, 3,4-dimethylpyrazole phosphate, i.e. DMPP, nitrapyrin, neem oil, piadin, chlorinated pyridine, calcium carbide), using the same N rate and N source than comparator.	Fertilization without NI	Considered metric: NO3-leaching; Effect size: logarithm of ratio of the considered metrics in fertilization with NI to the considered metrics in fertilization without NI.	62%

Results

- NO3- leaching was reduced by 45% (CI: 38 to 50%) by NI amendment. DCD and DMPP application significantly decreased NO3- leaching by 43.8 (37.0 to 50.2%) and 54.2% (48.6 to 59.1%), respectively.
- NI affected N losses in a similar pattern across all cropping systems, reducing NO3- leaching loss by 35% (CI: 13 to 52%) in dryland to 52% (CI: 45 to 53%) in paddy.
- NA
- NA
- NA

Factors influencing effect sizes

NA

Conclusion

Overall, NI amendment reduced NO3- leaching. More specifically, DCD and DMPP application significantly reduced NO3- leaching.

GETTING IN TOUCH WITH THE EU

In person

All over the European Union there are hundreds of Europe Direct centres. You can find the address of the centre nearest you online (european-union.europa.eu/contact-eu/meet-us_en).

On the phone or in writing

Europe Direct is a service that answers your questions about the European Union. You can contact this service:

— by freephone: 00 800 6 7 8 9 10 11 (certain operators may charge for these calls),

— at the following standard number: +32 22999696,

— via the following form: european-union.europa.eu/contact-eu/write-us_en.

FINDING INFORMATION ABOUT THE EU

Online

Information about the European Union in all the official languages of the EU is available on the Europa website (european-union.europa.eu).

EU publications

You can view or order EU publications at op.europa.eu/en/publications. Multiple copies of free publications can be obtained by contacting Europe Direct or your local documentation centre (european-union.europa.eu/contact-eu/meet-us_en).

EU law and related documents

For access to legal information from the EU, including all EU law since 1951 in all the official language versions, go to EUR-Lex (eur-lex.europa.eu).

Open data from the EU

The portal data.europa.eu provides access to open datasets from the EU institutions, bodies and agencies. These can be downloaded and reused for free, for both commercial and non-commercial purposes. The portal also provides access to a wealth of datasets from European countries.

The European Commission's science and knowledge service

Joint Research Centre

JRC Mission

As the science and knowledge service of the European Commission, the Joint Research Centre's mission is to support EU policies with independent evidence throughout the whole policy cycle.

EU Science Hub
joint-research-centre.ec.europa.eu

@EU_ScienceHub

EU Science Hub - Joint Research Centre

EU Science, Research and Innovation

EU Science Hub

EU Science

Publications Office
of the European Union