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Assessing the effectiveness of the Nitrates Directive: implications for agriculture and for water, soil and air quality

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Contents

Abstract

The Nitrates Directive has reduced nitrogen input to waters and promoted better management practices, however it takes time to see its effects because of the lag time of nitrate pollution in groundwater. The 50 mg/L nitrate threshold seems insufficient to protect the ecological quality of water systems. In some European regions, high livestock densities and manure production exceed safe limits, driving nitrate pollution. The 170 kg/ha manure limit has contributed to capping livestock numbers and improve air quality, but a more integrated approach to nutrient management is needed to balance agricultural production with environmental protection. An integrated approach to crop nutrient management and reduced nitrogen input can minimize environmental impacts without significant yield losses. Adopting sustainable farming practices and manure recycling can reduce nitrogen losses and support EU green transition and climate neutrality goals, ensuring water resilience and sustainable food production.

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1 Introduction

The Nitrates Directive (91/676/EEC) is an ambitious legislation of the European Union adopted in 1991 aiming at protecting waters against pollution caused by nitrates from agricultural sources. However, after more than three decades since its implementation diffuse pollution from agriculture is still a major pressure on water ecosystems in many European regions, preventing the achievement of water quality objectives. In 2021, only 37% of European surface water bodies were classified in good or excellent ecological status, with the most significant pressure affecting both surface and groundwater being agricultural activity, associated with both water abstractions and nutrients and pesticides applications (EEA, 2024). According to the implementation report of the Nitrates Directive (reporting period 2016-2019) 14% of groundwater stations exceeded the annual average 50 mg nitrates/L, and 81% of marine waters and about one third of rivers, lakes, transitional and coastal waters were reported as eutrophic. In spite of some progress, the level of implementation of the Directive is still insufficient to reach the objectives (COM(2021) 1000 final).

The purpose of this scientific study was to shed light on the effectiveness of the Nitrates Directive in achieving its goal and reflect critically on the implication/efficacy of the limit of manure application (170 kgN/ha), set by the directive in Nitrates Vulnerable Zones, considering environmental, health and agronomic implications. Our overall goal was to offer scientific evidence for informing the policy discussion whether the Nitrates Directive is still fit for purpose, in view of the current EU objectives of green transition and climate neutrality. This is timely and relevant, considering that the Directive is at the crossroads between the EU Common Agricultural Policy, water policy, societal changes and future visions for sustainable food production and water resilience.

Specifically, in this study, we first analyse whether the Directive was effective in achieving its goal, providing an overall analysis of water quality (nitrates) trends for Europe, and two case studies, one for France and another for the Danube river basin (Section 2). Then we explore if the limit of manure application appears appropriate to protect the environment and human health, considering water, soil and air quality and their sectoral policies (Section 3), and we delve into the implication of the limit for crop production and livestock density in Europe (Section 4). Finally, we examine the manure limit in view of future opportunities, i.e. farming practices, recycling, and challenges related to climate, agriculture and water management (Section 5). In the Conclusions we provide an overview of the key messages emerging from the analysis and offer some points of reflection (Section 6).

2 Was the Nitrates Directive effective in achieving its goal?

Assessing the effectiveness of the implementation of the Nitrates Directive is a complex task, as it requires disentangling the relationships between agricultural pressures including fertilization and farming practices, local environmental factor such soil type and climate, and larger scale environmental pressures such as climate variability and climate change. In addition, the inertia of environmental systems, or lagged response to measure implementation to reduce nutrient losses requires careful consideration of temporal and spatial variability.

In this chapter, after defining the objectives of the Nitrates Directive (§ 2.1), we examine data on the evolution of nutrient and water quality in Europe since the 1990s (§ 2.2), when the Nitrates Directive entered into force. Then we analyse the presence of detectable trends in nitrate concentration of surface water and groundwater considering the monitoring data reported by the Member States under the Nitrates Directive since the 7th Reporting Period (§ 2.3.1). In the last part, we delve deeper into the analysis of the effectiveness of the Nitrates Directive, providing two case studies, one for France, based on the work of Bouraoui et al. (2026) (§ 2.3.2), and the other for the Danube region, drawing on the results of Udias et al. (under review) (§ 2.3.3)

2.1 The Nitrates Directive: objectives and reporting

The objective of the EU Nitrates Directive (91/676/EEC) is to reduce water pollution caused by nitrates from agricultural sources and prevent further pollution. The overall aim of the Directive is to protect groundwater and surface water quality by reducing the levels of nitrates and to prevent eutrophication (i.e. the excessive growth of algae and aquatic plants caused by an overabundance of nutrients, including nitrates).

To achieve these objectives, the Nitrates Directive requires EU Member States to:

1. Identify **Nitrates Vulnerable Zones** (NVZ), that are areas of land in their territories which drain into the waters where the nitrate concentration in the groundwater or surface water exceeds or could exceed 50 mg/l (or 11.3 mg/l of nitrate-nitrogen), if action is not taken (Article 3, Annex I), and water bodies affected by eutrophication or that could be so if no measures are taken.

2. Establish a code or **codes of good agricultural practice**, to be implemented by farmers on a voluntary basis, regarding inter alia the application of fertilisers, the storage capacity for livestock manure, crop rotation and fertilizer plans, taking account of conditions in the different regions (Article 4).

3. Within Nitrates Vulnerable Zones, establish **Action Programs** to reduce nitrate pollution (Article 5). The measures in the Action Programs “will ensure that, for each farm or livestock unit, the amount of livestock manure applied to the land each year, including by the animals themselves, shall not exceed a specified amount per hectare. The specified amount per hectare be the amount of manure containing 170 kg N” (Annex III).

4. Implement suitable **monitoring programmes** to assess the effectiveness of Action Programs (Article 6) and report on the effectiveness of these measures every 4 years (Article 10).

Based on the reports from the Member States, the Commission periodically publishes a report on the status of implementation of the Directive (Article 11). The implementation report of the Nitrates Directive, covering the period 2016-2019, reveals that nitrate pollution remains a persistent environmental issue. Specifically, 14% of groundwater monitoring sites exceeded the acceptable annual average limit of 50 mg of nitrates per litre. Furthermore, a staggering 81% of marine waters and approximately one-third of rivers, lakes, and coastal waters were found to be affected by eutrophication. Despite some notable improvements, the overall implementation of the Directive falls short of achieving its intended goals, highlighting the need for continued efforts to address the ongoing environmental concerns (COM(2021) 1000 final).

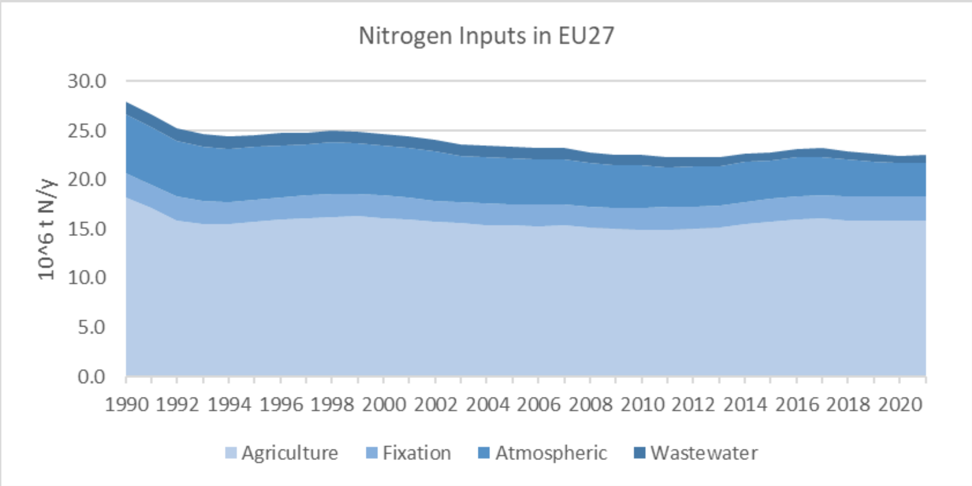
2.2 Evolution of nutrients input and water quality in EU27

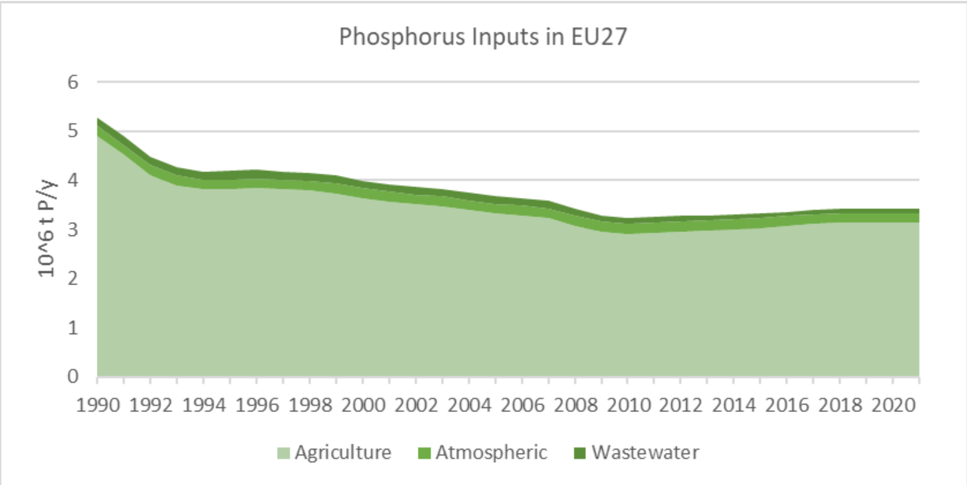
In EU27 the total nitrogen input to soils and surface waters was estimated to 22 10^6 tonnes of nitrogen per year in 2020, of which around 71% is represented by agricultural inputs, 15% by atmospheric deposition, 11% by soil and crop fixation, and the rest by urban and industrial wastewaters discharging directly into surface waters (Figure 2.1, Grizzetti et al. 2025). While nitrogen input from both atmospheric deposition and wastewaters has almost been halved between 1990 to 2020, the input from agriculture has slightly decreased (agricultural data from JRC CAPRI baseline, 2023), with the share of nitrogen from manure increasing from 31% to 36% (data of 2018). The average nitrogen manure application per hectare per NUTS2 regions have increased in 17 EU countries out of 27 (Figure 2.2).

Nutrients from mineral fertilisers and manure are in large part used by crops to grow and removed from soils, the rest is in part denitrified back into the atmosphere (as inert gas) and in part lost to the environment, polluting air and water, or accumulating in soils and aquifers. The gross nitrogen balance informs on the potential amount of nitrogen lost to the environment, as it represents the difference between the nitrogen inputs to the soil and the outputs (plant uptake). Its evolution since the '90s shows a general improvement of EU agricultural practices, with potentially less nitrogen losses to the environment in more recent years, although patters differ per each EU country (Figure 2.3).

Data on water quality reported by the European Environment Agency (EEA) provide some positive signals for nutrient pollution of European waters (EEA, 2024). Nitrate and phosphate concentrations in rivers and total phosphorus in lakes have declined since the '90ies. Surface water concentrations have stabilized, except for river phosphate, which has increased since 2013. Groundwater nitrate concentrations have remained relatively constant over time (Figure 2.4). The analysis is based on data reported by Member States to EEA covering all Europe, not only agricultural areas, but the spatial and temporal coverage of the monitoring network is not homogeneous.

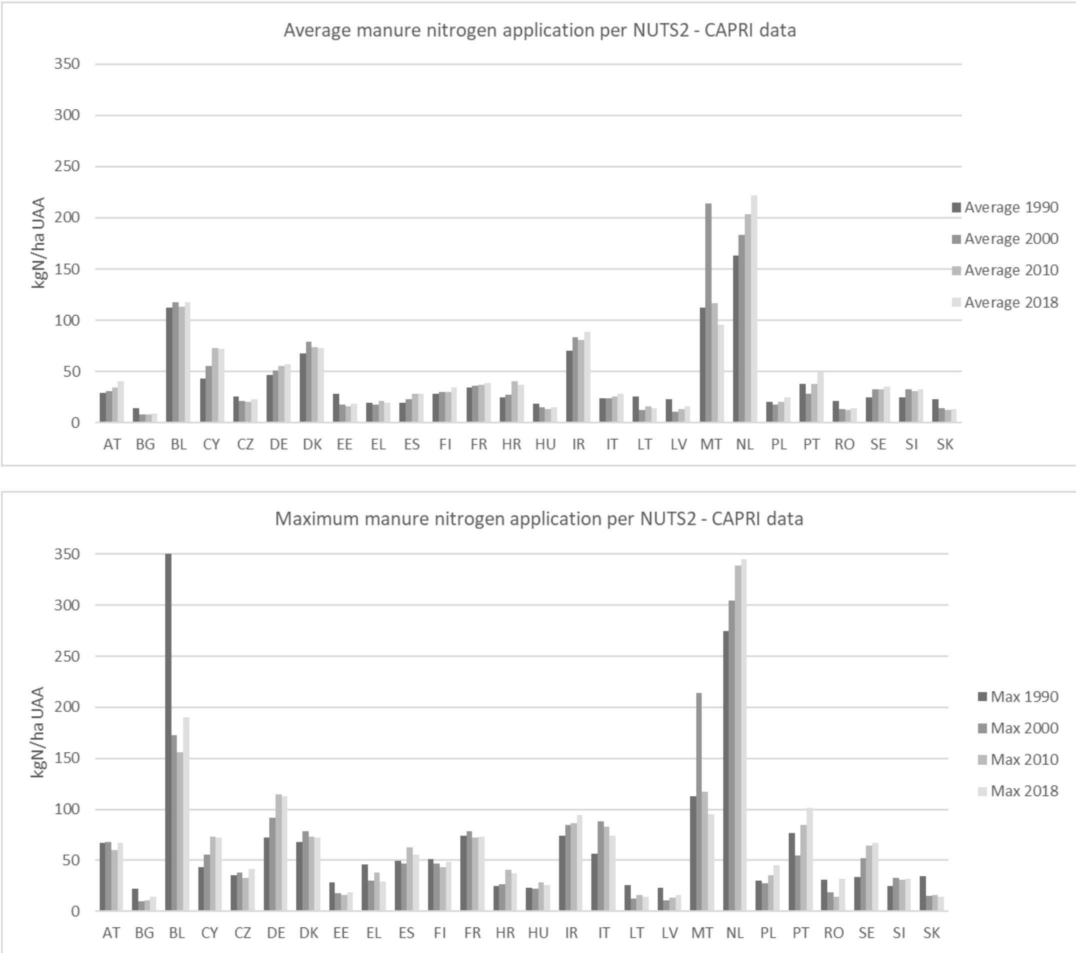
Figure 2.1. Total inputs of nitrogen (top) and phosphorus (bottom) to the river basins in EU27 countries from different sources: agriculture (including mineral and manure fertilisers), fixation (only for nitrogen, including crop and soil nitrogen fixation), atmospheric deposition and wastewater discharges from domestic and industrial sources.





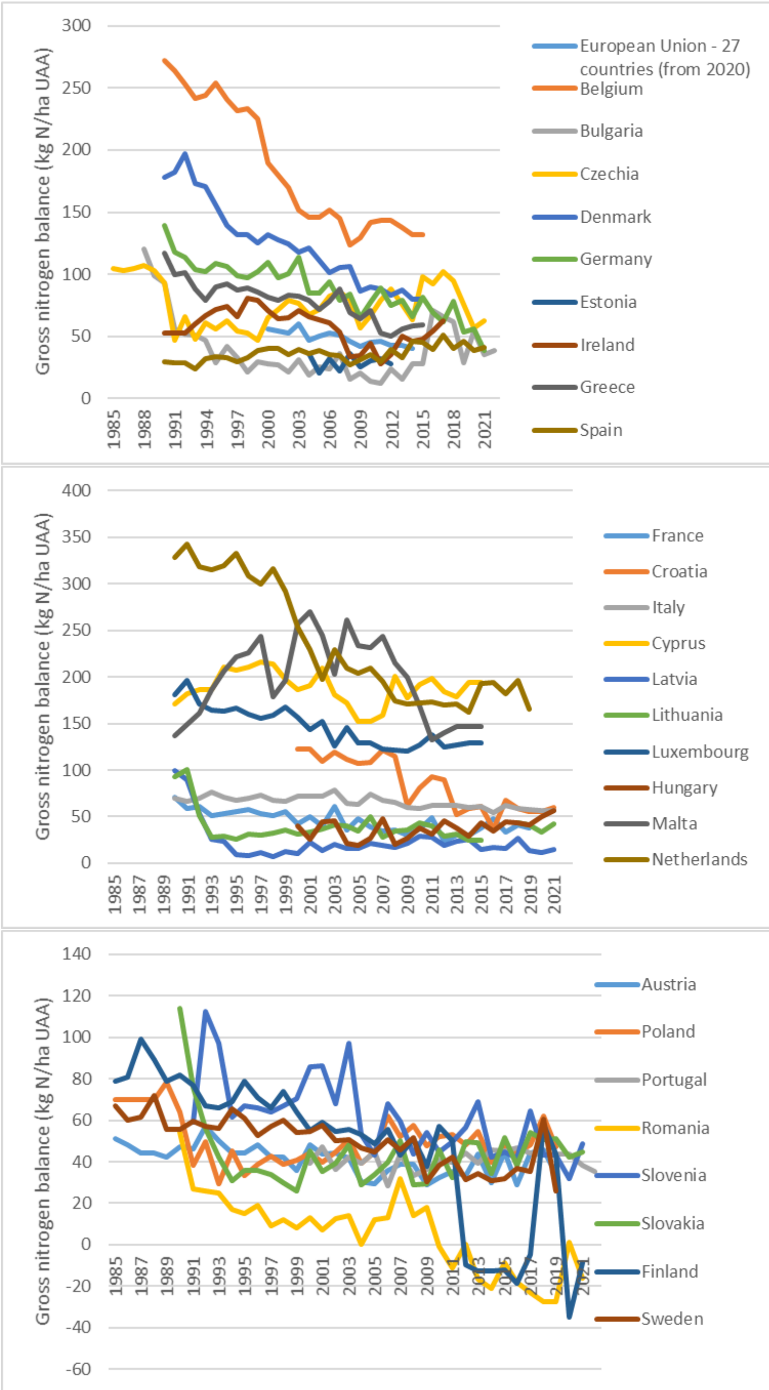
Source: JRC analysis, data from Grizzetti et al. (2025)

Figure 2.2. Manure nitrogen application rates per hectare of utilised agricultural area in EU27 countries, according to CAPRI model data: average rates (top) and maximum rates (bottom) by NUTS2 region for the years 1990, 2000, 2010, and 2018. *BL indicates Belgium and Luxembourg.



Source: JRC analysis, data from JRC CAPRI baseline (2023)

Figure 2.3. Evolution of the gross nitrogen balance per hectare of utilised agricultural area (UAA) between 1985 and 2021 in EU27 countries.



Source: EUROSTAT data, indicator [aei_pr_pnb] accessed on 14/11/2024.
https://ec.europa.eu/eurostat/databrowser/product/page/aei_pr_gnb__custom_14024042

Figure 2.4. Evolution of nutrient concentrations in European water bodies: nitrate in groundwater (top left) and rivers (top right), and phosphate in rivers (bottom left) and lakes (bottom right).

Source: EEA, 2025, 'Nutrients in freshwater in Europe', European Environment Agency. (https://www.eea.europa.eu/en/analysis/indicators/nutrients-in-freshwater-in-europe) accessed 13 May 2025

2.3 Effectiveness of the Nitrates Directive

2.3.1 Trend analysis based on Nitrates Directive reported data

We analysed the evolution of nitrates concentrations in surface and groundwater in Europe using the most recent data reported by Member States under the Nitrates Directive for the Reporting Period 2016-2019 (RP7) and 2020-2023 (RP8) for a total of 8 years. We computed the Regional Mann-Kendall trend test to identify statistically significant upward or downward trends in nitrate concentrations in surface and groundwater between 2016 and 2023 (Box 2.1).

Box 2.1. Mann-Kendall trend tests.

The **Mann-Kendall test** is a non-parametric method used to identify whether there is a trend (upward or downward) in a time series over time. It is particularly useful in environmental studies where data may not meet assumptions of normality.

To calculate the trend:

1. For each pair of observations (x_i, x_j) where j > i, compute the difference (x_j - x_i).

2. Assign a sign to each difference:

   +1 if x_j > x_i

   0 if x_j = x_i

   -1 if x_j < x_i

3. Sum all the signs. This total is the S-statistic.

Interpretation:

   S > 0: upward trend (potential)

   S < 0: downward trend (potential)

   S ≈ 0: no trend (potential)

The **Regional Mann-Kendall test** extends the single-site test to multiple sites in a region. It is used to assess whether a consistent trend (in nitrate concentrations) exists across all sites.

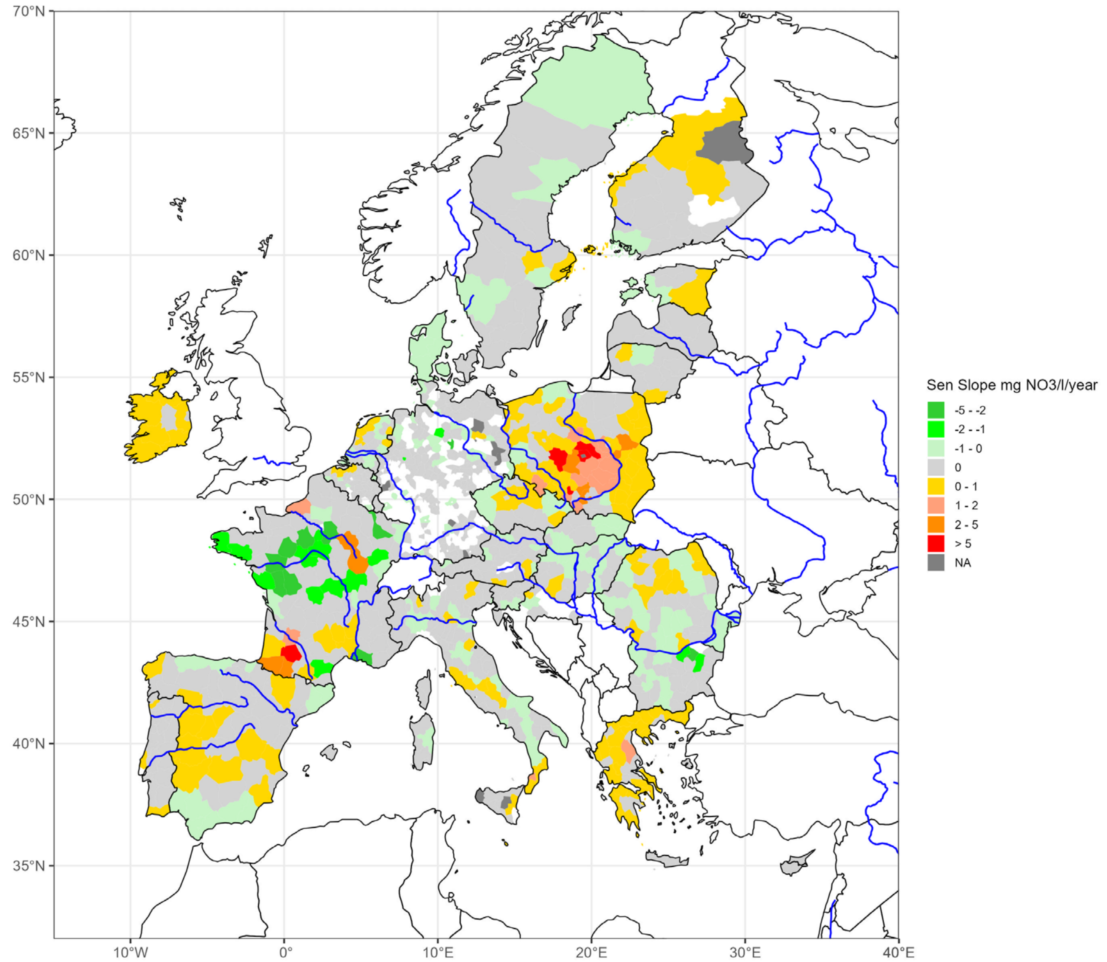
To calculate the trend:

1. Apply the Mann-Kendall test to each site's time series (of nitrate concentrations)
2. Convert each site's result into a Z-score (a standardized statistic)
3. Combine the Z-scores across sites using a method such as the average or sum adjusted for the number of sites
4. Evaluate the combined Z-score against a standard normal distribution to determine regional significance

This method helps identify broad spatial trends in environmental pollution and is ideal for water quality assessments like nitrate levels to detect widespread environmental changes. This method is particularly useful in environmental monitoring to identify consistent regional changes in pollutant levels such as nitrate concentration.

The nitrate concentration data was retrieved on May 25 2025 and included reported concentration for all Member States. Annual nitrate concentration in surface water was reported for 34,430 stations in the RP7 and for 30,828 stations in the RP8. Matching between the RP7 and RP8 monitoring networks resulted in only 21,918 common stations for surface waters. Each station was assigned to its corresponding NUTS3 region and the regional Mann-Kendall test was used for calculating the trend in each NUTS3 region (Figure 2.5). The analysis generally shows a mixed picture with both increasing and decreasing trends of nitrate concentration in several EU countries. Most prominent increasing trends are apparent in some regions of Poland, France and Greece, while marked improvements are evident in some part of France and Bulgaria. Many EU regions shows no significant trends.

Figure 2.5. Regional Mann-Kendall trend analysis of nitrate concentration in EU NUT3 regions based on data reported under the Nitrates Directive for surface water for the period 2016-2023.



Source: JRC analysis (Nitrates Directive data access 25th May 2025)

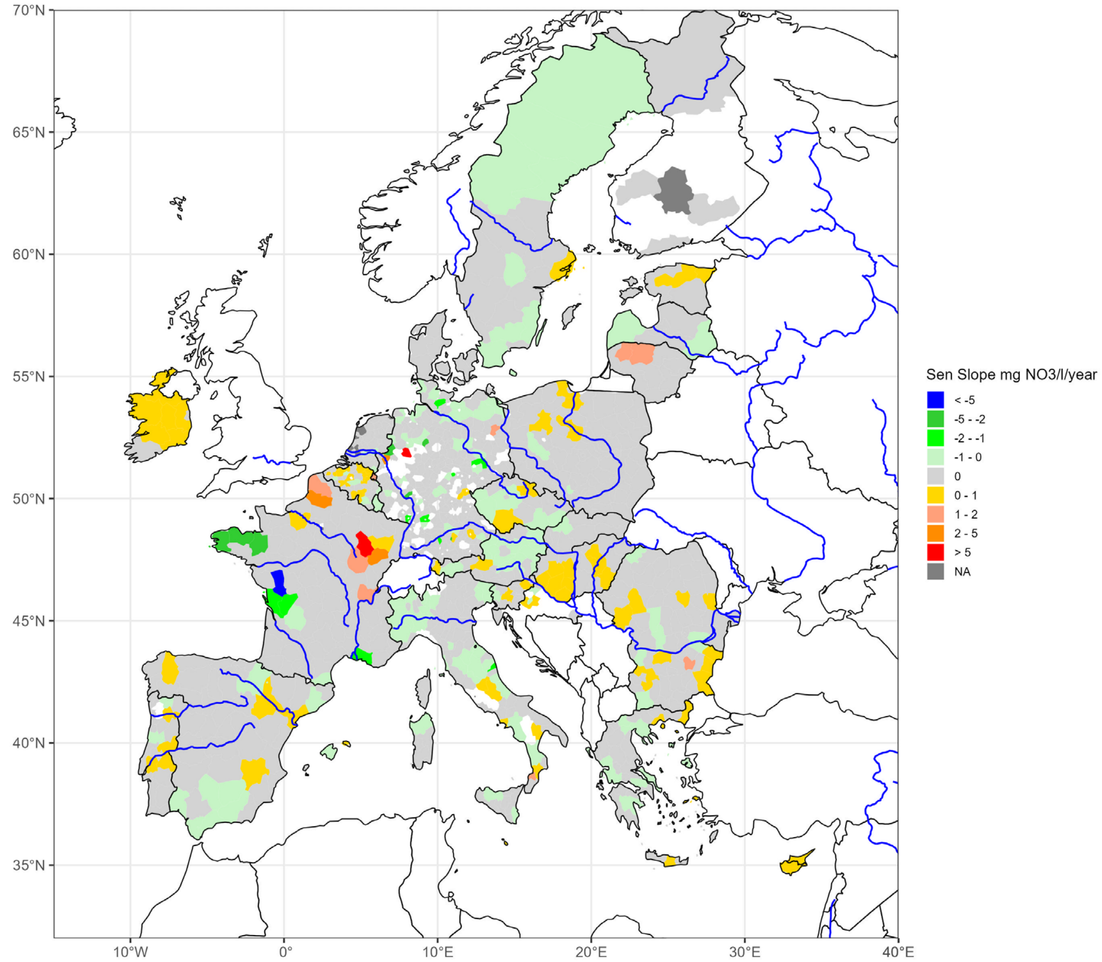
Annual nitrate concentration in groundwater was reported for 34,430 stations in the RP7 and for 30,090 stations in the RP8, with 27,402 matching stations between the RP7 and RP8 monitoring networks. The trend analysis of groundwater nitrate concentration, by the regional Mann-Kendall test for NUTS3 regions, shows no significant trends for large part of the EU (Figure 2.6). Increasing trends are detected in Ireland, Estonia, Latvia and Hungary, decreasing trends are present in

Sweden and Greece. Overall, there is a mix of increasing and decreasing trends in all countries, with prominent hotspots in France.

In some countries trends of nitrate concentration in surface and groundwater points in the same direction, such as in Ireland, in other countries, for example in Greece and Bulgaria, trends are opposite, showing higher disconnection between surface and groundwater nitrate pollution or different lag time.

Finally, it is important to notice that trends offer insight into changes over the 2016-2023 study period. Nevertheless, these trends should be considered alongside the absolute concentrations, taking into account whether they surpass or remain below the legally established threshold of 50 mg NO3/l.

Figure 2.6. Regional Mann-Kendall trend analysis of nitrate concentration in EU NUT3 regions based on data reported under the Nitrates Directive for groundwater for the period 2016-2023.



Source: JRC analysis (Nitrates Directive data access 25th May 2025)

2.3.2 France case study

This case study is based on Bouraoui et al. (2026).

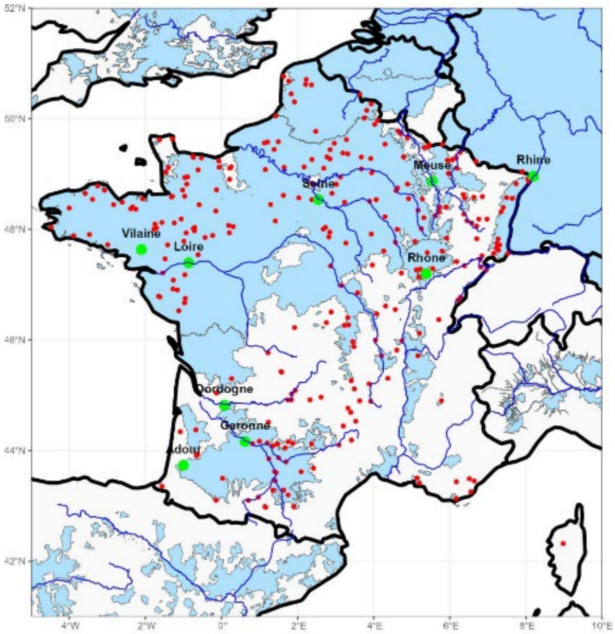
To accurately identify trends in water-quality time series, it is essential to remove the influence of interannual streamflow variability. In addition, accounting for systematic long-term changes in streamflow can help distinguish hydrological effects from changes associated with watershed management and anthropogenic pressures. We assessed long-term nitrate trends at water-quality monitoring stations in France after removing interannual streamflow variability and apportioning the effects of management and streamflow change using the WRTDS framework and generalized flow normalization. We further examined the evolution of management-related nitrate changes in relation to agricultural nitrogen inputs, including mineral and manure application, biological nitrogen fixation, and atmospheric deposition, using pressure data from Vigiak et al. (2023) and Grizzetti et al. (2025) for the period 1990–2021.

Box 2.2. Approach for assessing long-term trends of nitrate concentration in surface water

> We used the Weighted Regressions on Time, Discharge, and Season (WRTDS) method, implemented in the R package EGRET (Exploration and Graphics for RivEr Trends), to analyze long-term changes in water quality. WRTDS is a statistical framework for assessing water-quality trends by modeling concentration as a function of time, discharge, and season. Its flow-normalization procedure reduces the influence of interannual streamflow variability on estimated trends. Because classical flow normalization assumes a stationary discharge regime, we also applied generalized flow normalization to account for nonstationary streamflow conditions and long-term changes in flow patterns. This approach enables a more robust evaluation of the respective influences of watershed management, land-use change, and hydrological variability on water-quality trends.

WRTDS requires discrete water quality samples, including concentration values for the parameter of interest and a complete record of daily mean streamflow or discharge for the duration of the water quality record. The initial dataset for France was retrieved from EAUFRANCE (2024). Matching the positions of the water quality stations and applying the data requirements narrowed down the network to 268 stations (Figure 2.7), of which 187 fall inside Nitrate Vulnerable Zone (NVZ) areas. The concentration data were processed to determine, for each station, the overall trend in nitrate concentration from 1990 to 2022, i.e. the trend in concentration without year-to-year flow variability, and then the trend due to management, where year-to-year variability and long-term flow changes were removed. The difference between the overall trend and the trend due to management indicates the trend due to long-term flow changes, for example due to climate or changes in water availability.

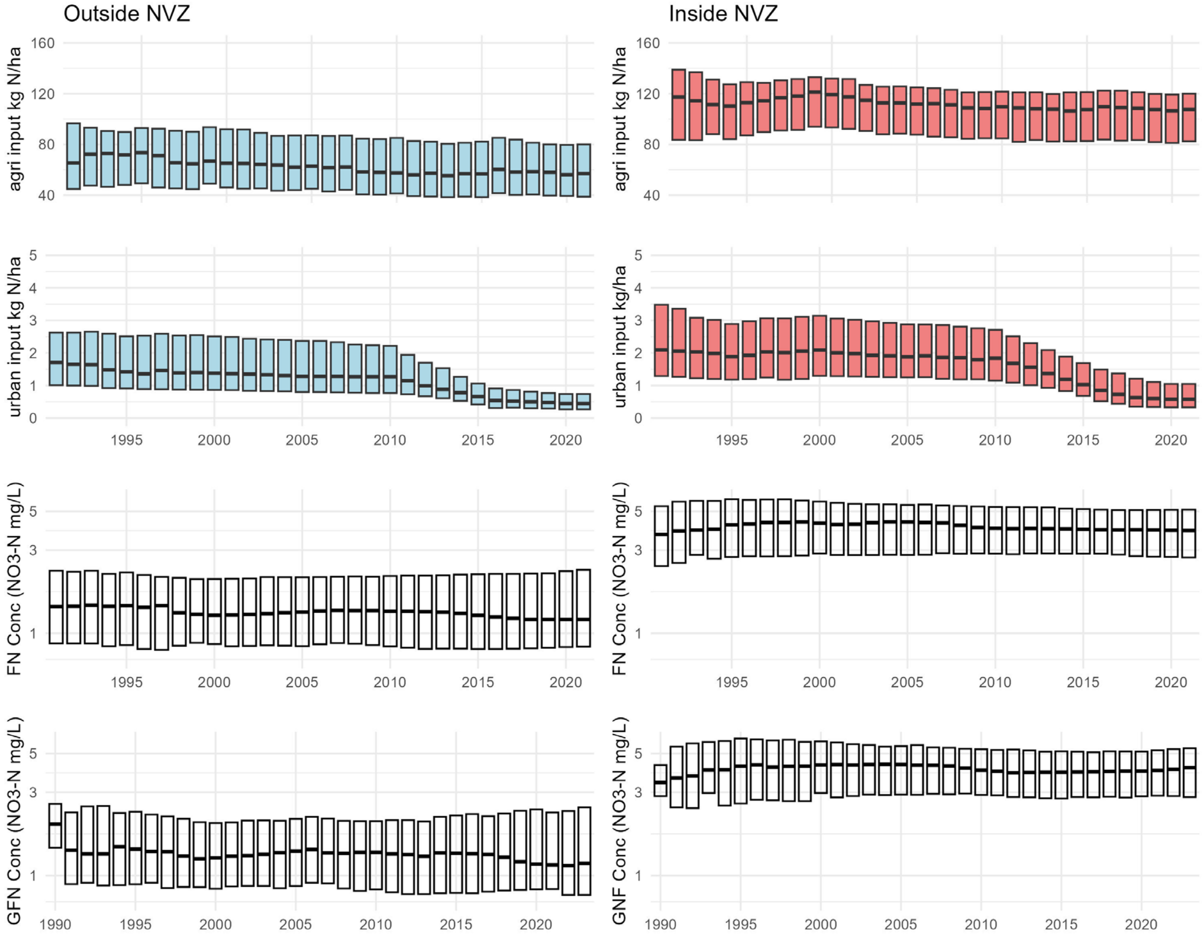
Figure 2.7. Monitoring stations of nitrate concentration in surface water included in the study. Legend: red dots indicate the monitoring network and green dots highlight the most downstream stations for individual basins. Light blue areas represents the designated Nitrate Vulnerable Zones for the period 2016–2019 (JRC, 2025a)



Source: JRC analysis, Bouraoui et al. (2026)

The analysis shows that agricultural inputs, calculated as the sum of manure and mineral fertilizer application, nitrogen fixation, and nitrogen deposition on agricultural areas, are much higher in NVZ areas than outside them (Figure 2.8). Flow-normalized nitrate concentrations, representing the management-related component of the trend, are much lower outside NVZ areas than inside, indicating the significant impact of intensive agriculture on water quality. The median nitrate concentration remained stable outside NVZ areas, while inside NVZ areas the management-related nitrate concentration increased from 1995 to around 2005 and then decreased significantly after 2005, with current concentrations close to those measured in 1995 (Figure 2.8).

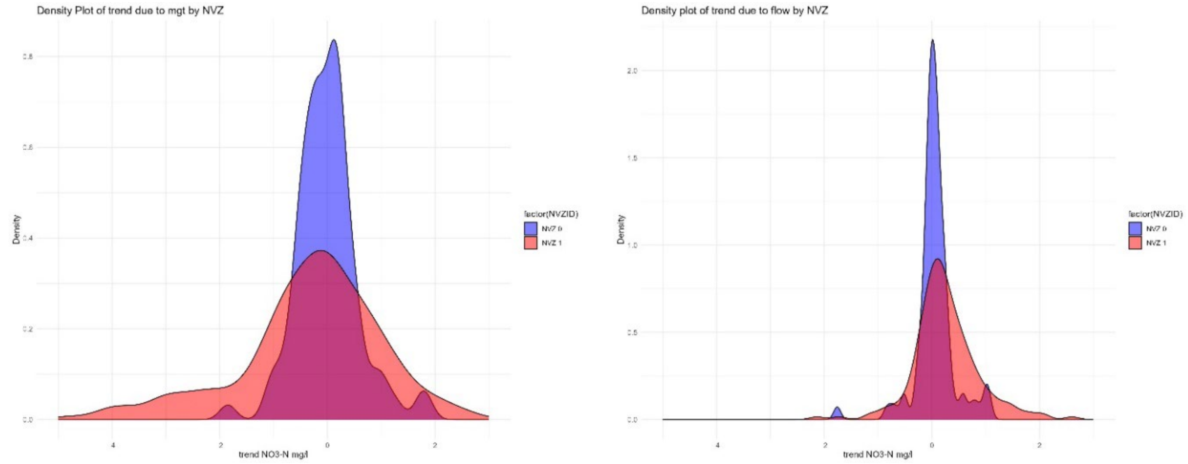
Figure 2.8. Temporal evolution of agricultural nitrogen inputs, calculated as the sum of manure and mineral fertilizer application, nitrogen fixation, and atmospheric deposition on agricultural areas (first row), and nitrogen discharges from wastewater treatment plants (second row), in the drainage areas of the monitoring stations. The third row shows flow-normalized nitrate concentrations, representing the management-related component of the trend, while the fourth row shows generalized flow-normalized nitrate concentrations, representing the overall trend after removing interannual flow variability. Results are shown outside (left column) and inside (right column) Nitrate Vulnerable Zones (NVZs) from 1990 to 2022 for the 268 French monitoring stations included in the analysis. The black line in each boxplot indicates the median value.



Source: JRC analysis, Bouraoui et al. (2026)

We examined the differences between nitrate concentration trends inside and outside NVZ areas (Figure 2.9). The trends outside NVZ areas are centered around zero, with a relatively narrow range of values, indicating the presence of both positive and negative trends. Inside NVZ areas, the distribution of the management-related trend is skewed to the left, showing a larger number of stations with decreasing trends than with increasing trends. The magnitude of the trend is also more pronounced inside than outside NVZ areas. On the other hand, long-term flow changes tend to counteract this management effect inside NVZ areas, with a large majority of stations exhibiting an increasing trend due to flow changes (Figure 2.9). Overall, the combination of flow and management trends still results in a significantly larger number of stations with decreasing nitrate concentrations inside NVZ areas.

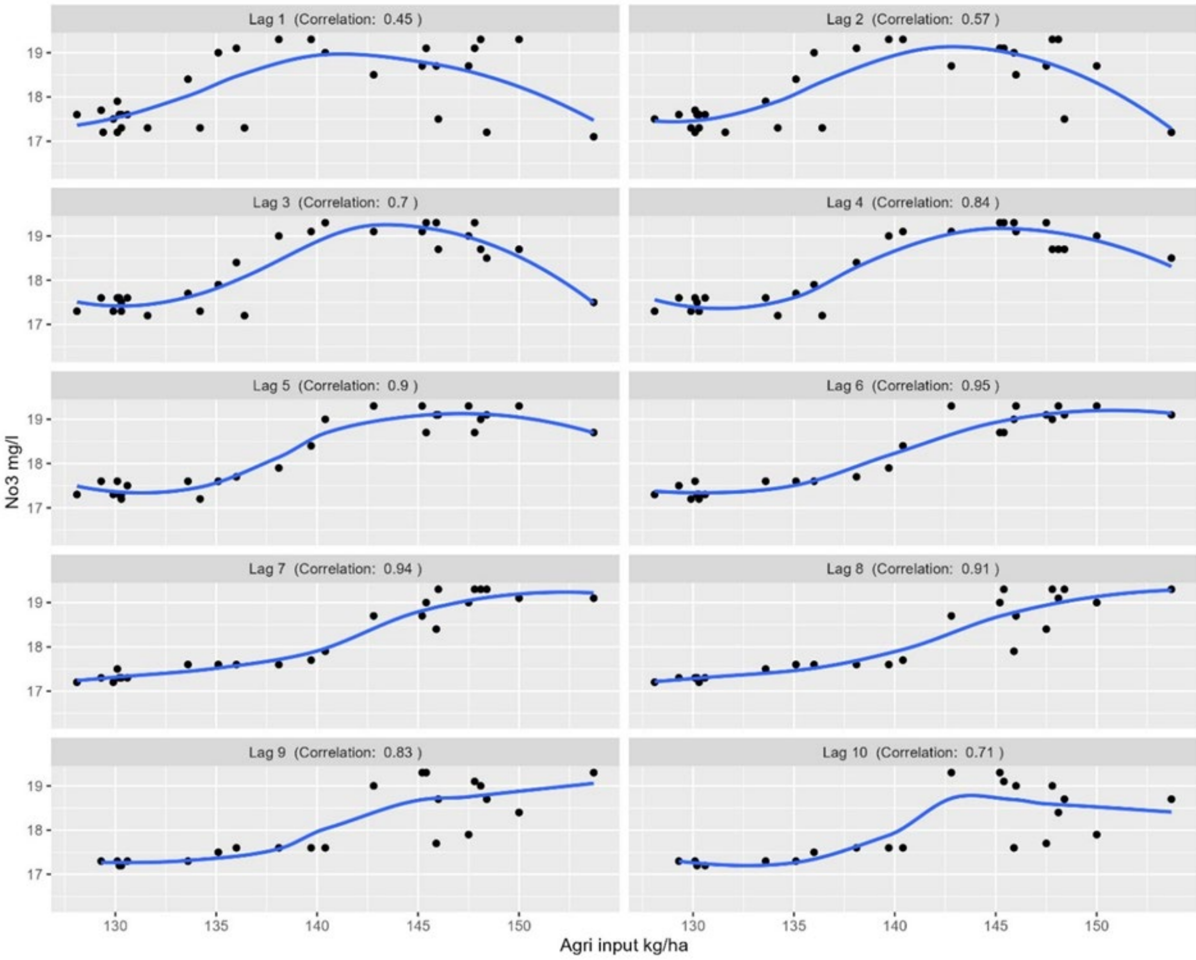
Figure 2.9. Distribution of trends of nitrate concentration inside (red) and outside (blue) NVZ areas due to management (left) and due to long term flow changes (right).



Source: JRC analysis

A lag analysis was performed to understand the time shift between changes in agricultural pressure and nitrate concentrations in surface water (Figure 2.10). The results showed a lag time of about 6 years, corresponding to the strongest correlation between agricultural nitrogen input and flow-normalized nitrate concentration, although with large variability. Lag times ranged between 0 and 10 years, values that are coherent with those reported in the literature (Figure 2.10).

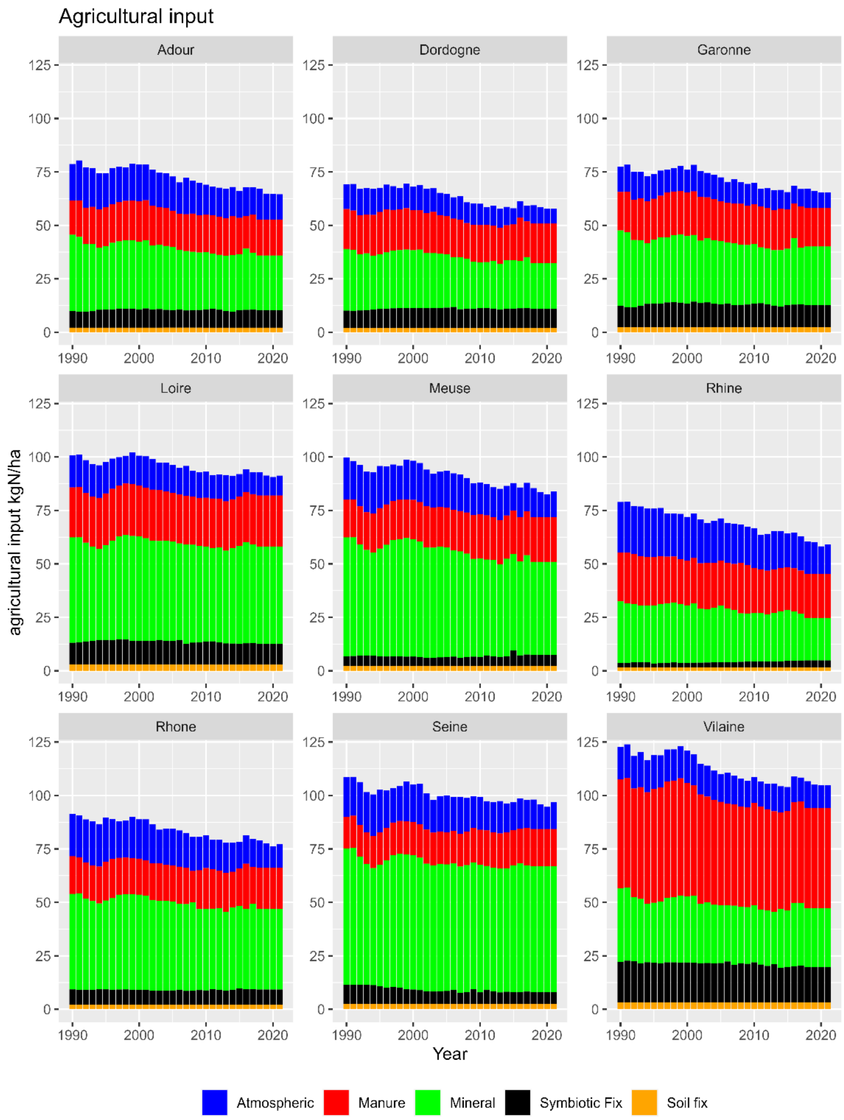
Figure 2.10. Lag time analysis: correlation between flow normalised nitrate concentration (without interannual flow variability) and agricultural input per different lag times (from 1 to 10 years).



Source: JRC analysis

The trends due to management were analysed at the most downstream monitoring stations for the nine major river basins with available monitoring data. Agricultural inputs are displayed in Figure 2.11. All catchments showed a significant decrease in agricultural inputs, particularly in mineral N fertilizer use and atmospheric N deposition. The use of N from manure application has remained stable since 1990. Symbiotic fixation, mostly in temporary and permanent grasslands, remains high in the Vilaine river basin.

Figure 2.11. Agricultural input (kgN/ha) for the large French river basins (drainage areas at to the most downstream monitoring stations available in the analysis).

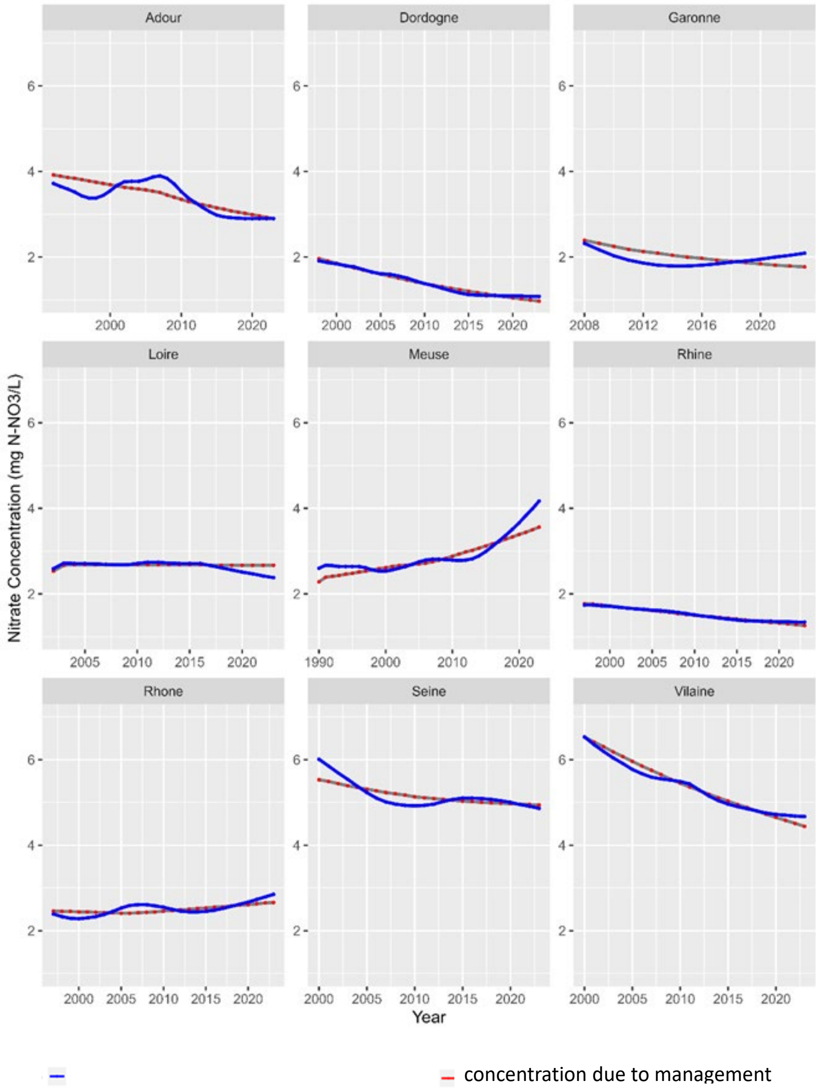


Source: JRC analysis

The overall trend and the trend due to management were calculated for the nine basins (Figure 2.12). The difference between the two trends indicates the effect of long-term flow changes. Most basins exhibited a decreasing nitrate concentration trend, while only the Meuse and the Rhone showed a continuous increasing trend. For the Rhone and the Meuse, both the overall concentration and the management-related concentration increased despite significant reductions in agricultural and urban inputs. This probably indicates that legacy effects and lag times are influencing the basin response. For the Loire, the overall nitrate concentration showed a slight increase, while the management-related concentration showed a slight decrease, suggesting that long-term flow changes partly offset the management-related improvement. An antagonistic effect can also be observed for the Garonne, where management resulted in a decrease in nitrate concentration, while flow changes had an opposing effect, leading to an increase in concentration after 2012. The most significant decrease occurred in the Vilaine river basin, located in Brittany, which was the basin with the highest initial nitrate concentration.

The changes attributed to land use and management practices in the analysis can be related to the Nitrates Directive, although other policies or changes may also have contributed.

Figure 2.12. Evolution of annual nitrate concentration due to management and without flow interannual variability (overall trend, flow normalised concentration) at the most downstream monitoring stations of the large French river basins.



Source: JRC analysis, Bouraoui et al. (2026)

Conclusions from Bouraoui et al. (2026):

Effectiveness of the EU Nitrates Directive

- Implementation of the Nitrates Directive since the 1990s in designated Nitrate Vulnerable Zones (NVZs) has contributed to measurable improvements in water quality.
- However, spatial variability in implementation, environmental conditions, and catchment response times limits its overall effectiveness across regions.
- Phosphorus reductions, largely linked to improvements in urban wastewater treatment, have been more pronounced than nitrate reductions, resulting in an increased nitrogen-to-phosphorus (N:P) ratio.

Recommendations for improved nitrate management

- Enhanced implementation of agricultural Best Management Practices (BMPs), such as precision agriculture, optimized fertilizer use, improved manure management, and ecological restoration measures, including buffer strips and floodplain reconnection.
- Strengthened policy implementation, enforcement, and stakeholder engagement to ensure widespread adoption and long-term commitment.
- Climate change impacts should be explicitly considered, since climate-driven changes in rainfall, temperature, runoff, and hydrological regimes can undermine the effectiveness of nitrate reduction measures.
- Changes in rainfall and streamflow patterns may partly offset gains achieved through BMPs.

Strategic reporting and monitoring

Member States should assess the cumulative impacts of both current and previous Action Programmes, because nitrate responses are delayed by legacy effects and catchment lag times.

Regional differences and targeted strategies

Targeted regional efforts have been effective, for example in the Vilaine basin in Brittany, where nitrate reductions were particularly marked.

Regions such as the Rhône and Meuse basins require more sustained and adaptive strategies, continuous monitoring, and community engagement.

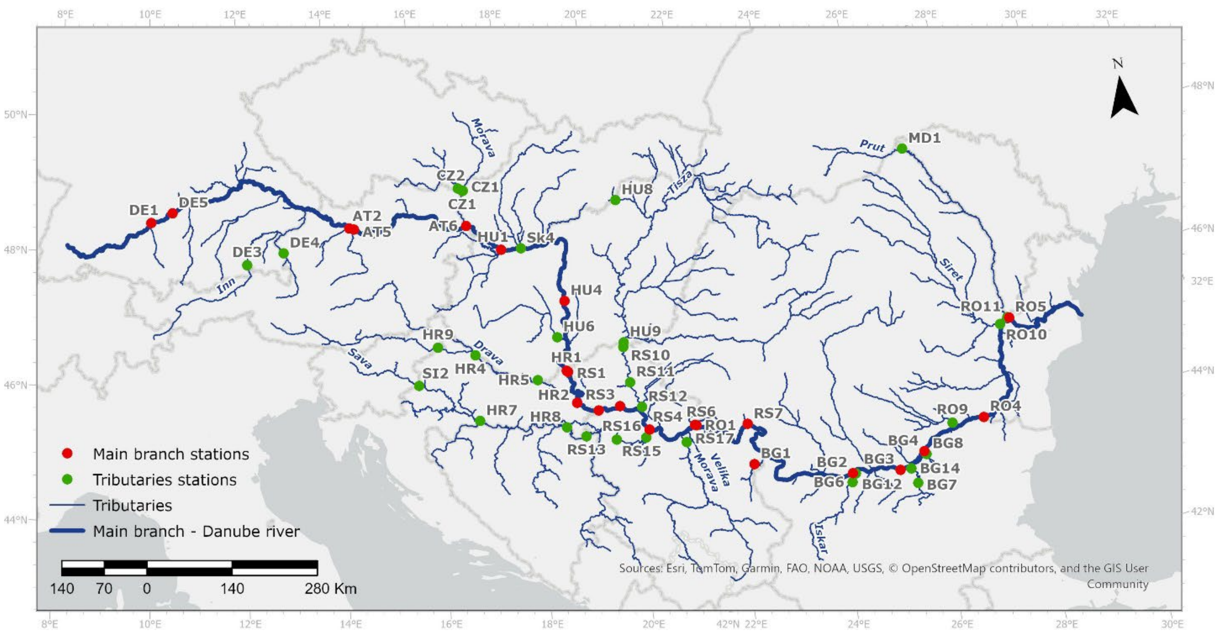
Integrated nutrient management

An integrated approach is needed to manage multiple nutrients simultaneously, accounting for nitrogen-phosphorus interactions in order to maintain ecosystem health and reduce eutrophication risks.

2.3.3 Danube case study

A comprehensive assessment of nutrient dynamics in the Danube River over a 25-year period (1996-2021) has been conducted by Udias et al. (under review) using an extensive dataset from numerous monitoring stations (34 stations along the Danube's main channel, and 28 stations on key tributaries) (Figure 2.13). This analysis focuses on the main nutrient species: Nitrate (NO3-N), Nitrite (NO2-N), Ammonium (NH4-N), Total Nitrogen (TN), Total Phosphorus (TP), and Orthophosphate-Phosphorus (PO4-P).

Figure 2.13. Water quality monitoring stations included in the study.



Source: JRC analysis, Udias et al. (under review)

The Weighted Regressions on Time, Discharge, and Season (WRTDS) methodology (see Box 2.2) was employed to estimate flow-normalized (FN) nutrient concentrations and fluxes. This approach is crucial for discerning long-term trends attributable to management interventions or environmental changes, effectively separating them from the confounding effects of hydrological variability, including seasonal patterns and long-term discharge trends (see also 2.3.2). In short, WRTDS removes the 'noise' of weather variability (dry vs. wet years) to reveal whether water quality policies are actually working.

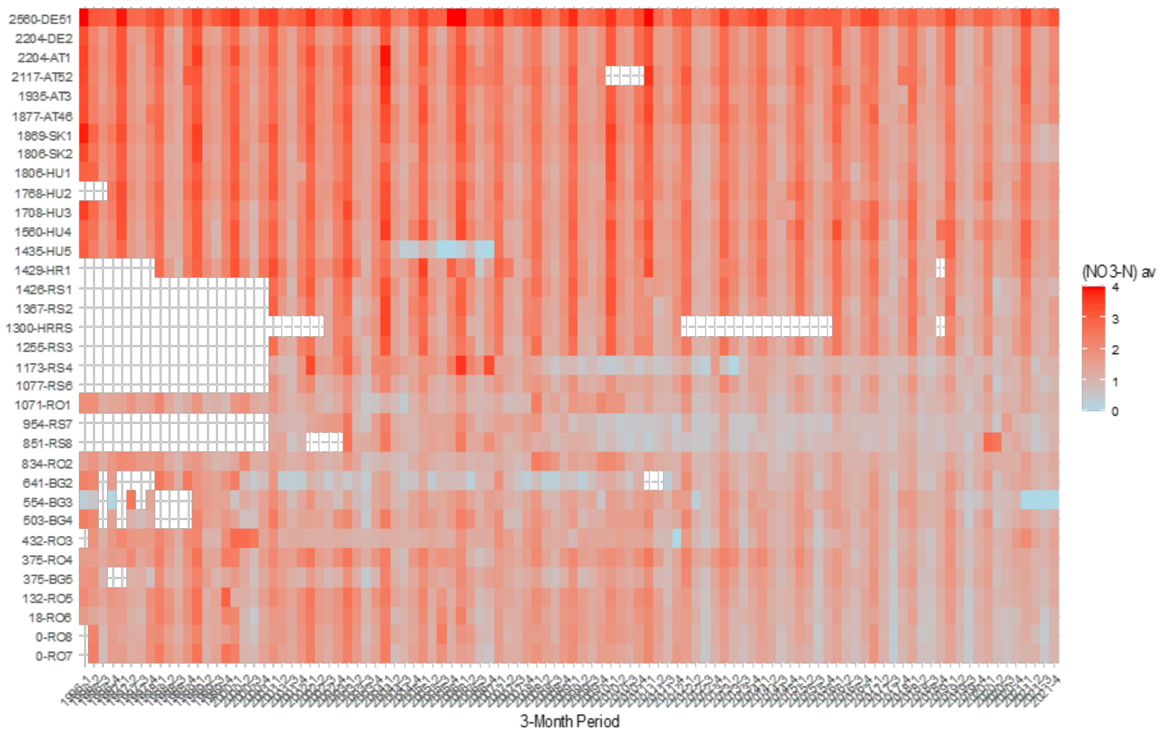
Contextual Streamflow Trends

Before detailing nutrient trends, it is important to note that the study period revealed complex streamflow dynamics. A general, though spatially variable, decrease in streamflow was observed along the Danube's main channel, particularly significant in the upper reaches (approximate annual slope of -0.5%). This downward trend generally moderates downstream, with slight increases even observed in the lower Danube for higher flows. A similar analysis of major tributaries indicated varied responses, with a decrease in flow observed in the Sava River (slope of approximately -0.4%), contrasting with slight increases of around 0.2% in the Tisza River and very slight increases in the Drava River. Beyond these long-term discharge trends, it is crucial to acknowledge the very pronounced seasonal variations in flow across the basin, where, for instance, daily maximum flows can be more than ten times greater than daily minimum flows, and typical monthly maximum flows are approximately 70% higher than monthly minimum flows. These significant inter-annual and intra-annual (seasonal) hydrological fluctuations underscore the absolute necessity of flow normalization techniques, like WRTDS, for an accurate assessment and interpretation of water quality trends.

Nitrate (NO3-N)

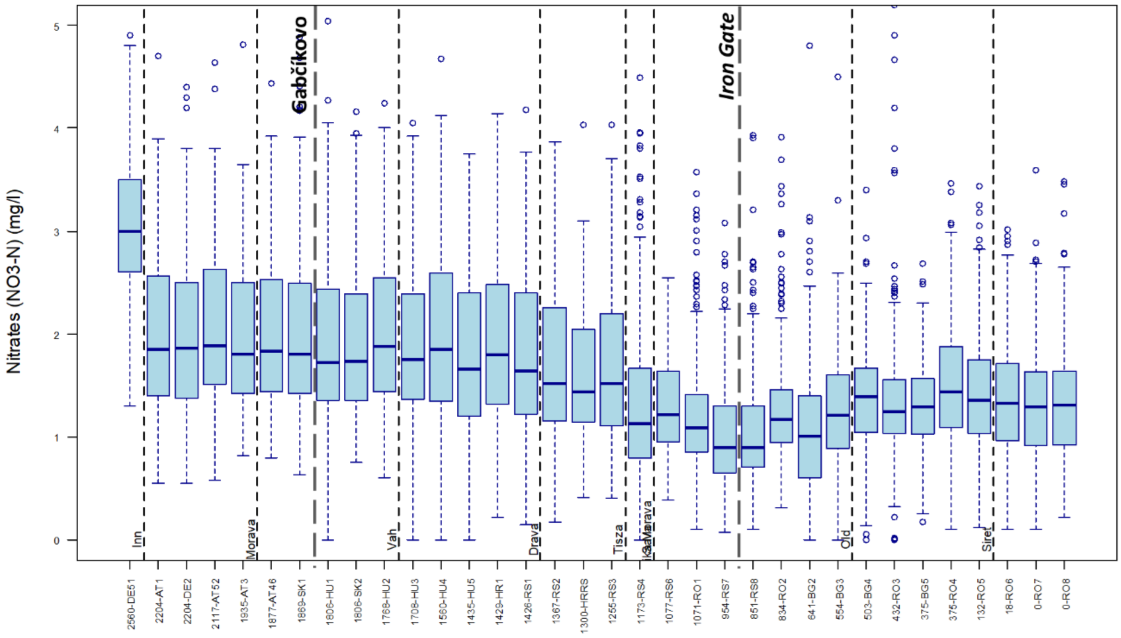
Initial graphical observations of raw nitrate concentration samples from the Danube's main channel, as depicted in the heatmap (Figure 2.14) and boxplots (Figure 2.15), reveal distinct spatial and temporal patterns. Figure 2.14, showing 3-month average concentrations from 1996 to 2021 across 34 monitoring stations, illustrates a general decline in concentrations over the 25-year study period. Spatially, as detailed in Figure 2.15 which illustrates concentration distributions relative to river kilometer and major confluences/reservoirs (Gabčíkovo and Iron Gate), a significant decrease in median nitrate concentrations is evident as one moves downstream from the source (from approximately 3 mg/L to 0.9 mg/L), particularly up to the Iron Gate reservoir (at river km 940). Beyond this point, a slight increase in concentration is often noted.

Figure 2.14. Heatmap showing the 3 month average concentration of NO3-N from 1996 to 2021 in 34 the monitoring stations along the Danube after data preprocess. Each station is identified by a number indicating its kilometer point along the river, followed by an identifier corresponding to the country in which it is located.



Source: JRC analysis, Udias et al. (under review)

Figure 2.15. Boxplot illustrating the concentration distributions of NO3-N for the monitoring stations along the Danube River. Each station is identified by a number indicating its kilometer point along the river, followed by an identifier corresponding to the country in which it is located. Also context regarding its relative position to the confluences of major tributaries and the two primary reservoirs: Gabčíkovo and Iron Gate.



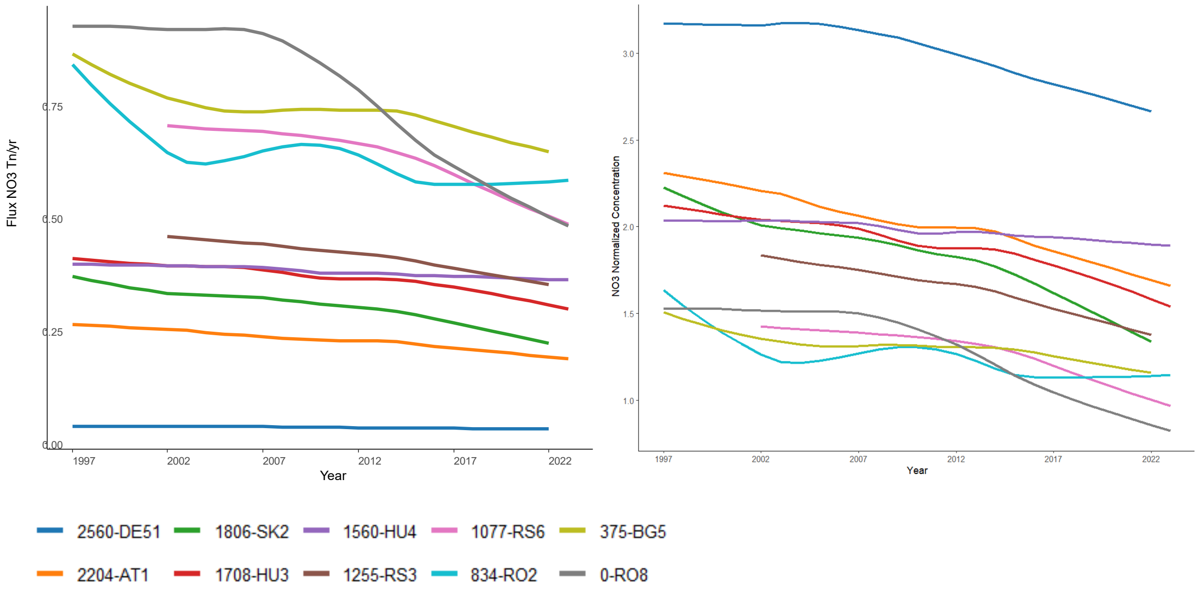
Source: JRC analysis, Udias et al. (under review)

However, deriving clear insights into long-term nutrient variations solely from raw concentration data is challenging due to the inherent variability in nutrient measurements and the significant fluctuations in Danube River flow rates (as discussed previously, including daily, seasonal, and long-term discharge trends. The WRTDS method effectively addresses these confounding factors by normalizing for discharge, allowing for a more precise analysis of underlying changes in nutrient concentrations and loads (Hirsch et al., 2010).

The WRTDS derived flow-normalized (FN) trends for NO3-N along the main channel, illustrated in Figure 2.16 (showing temporal evolution of FN fluxes on the left and FN concentrations on the right for selected representative stations), demonstrate a clear trend of reduced concentrations and fluxes across the entire Danube basin over the 1996-2021 period. FN flux reductions ranged from approximately 13% in the upper reaches to over 20% at the river's mouth. In absolute terms, this corresponds to a decrease in the nitrate flux entering the Black Sea, from around 820 tonnes per year in 1996 to approximately 650 tonnes per year in 2021.

A similar analytical approach was applied to data from 28 monitoring stations in the Danube's major tributaries. Nitrate concentration levels among these tributaries exhibit initial variability; for instance, recorded values were approximately 3.9 mg/L in the Morava River in 1996, compared to around 1.3 mg/L in the Drava, Tisza, and Sava Rivers for the same year. Despite this, a discernible decline in NO3 concentrations is evident across all tributaries, apparent in both raw data and flow-normalized estimates. Specifically, WRTDS-derived NO3 concentration reductions between 1996 and 2021 are estimated at approximately 25% in the Drava and Morava Rivers, and up to 30% in the Tisza River, all of which show statistically significant trends. In contrast, the seven monitoring stations along the Sava River indicate a reduction in nitrate concentrations of approximately 15%, albeit with lower statistical significance.

Figure 2.16. Temporal evolution of normalized NO3-N fluxes (left) and concentrations (right) for selected representative stations along the main channel of the Danube River from 1996 to 2021.



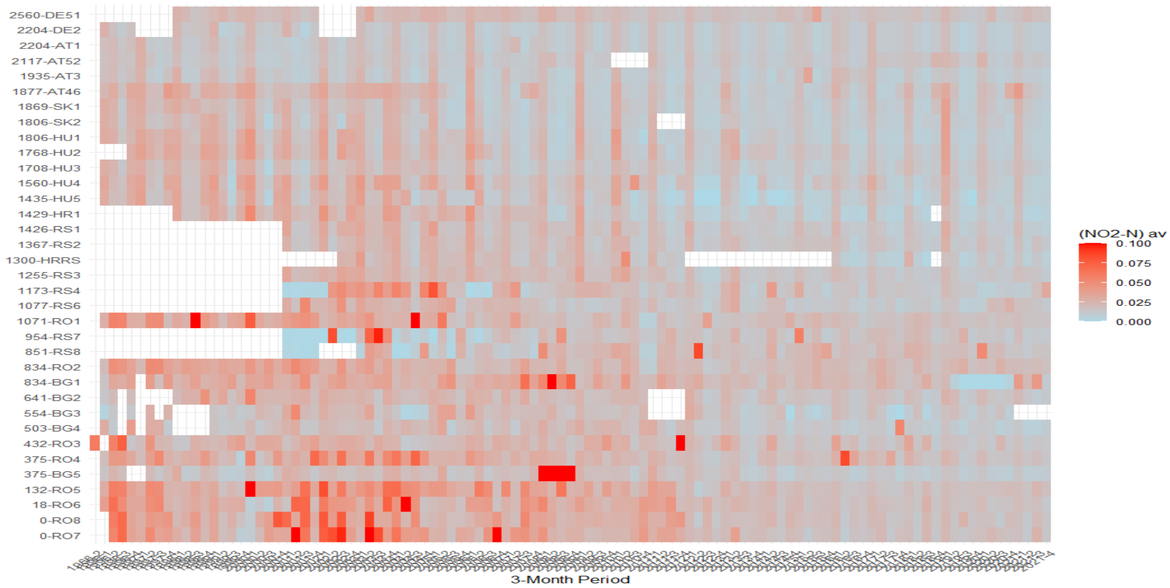
Source: JRC analysis, Udias et al. (under review)

Nitrites (NO2-N) and Total Nitrogen (TN)

Nitrite (NO2-N) concentrations are generally very low, often below 0.08 mg/L, with improvements over the study period, and an increase along the main river course (see heatmap Figure 2.17). In the tributaries, levels are also quite low, with values somewhat lower than those of the main channel in the Drava river and somewhat higher in the Morava river.

Though the dataset is less complete for Total Nitrogen (TN) due to data availability at stations in both the main Danube channel and its tributaries, TN data exhibit trends similar to those of NO3-N.

Figure 2.17. Heatmap showing the 3 month average concentration of NO2-N from 1996 to 2021 in 35 monitoring stations along the Danube after data preprocessing. Each station is identified by a number indicating its kilometer point along the river, followed by an identifier corresponding to the country in which it is located.

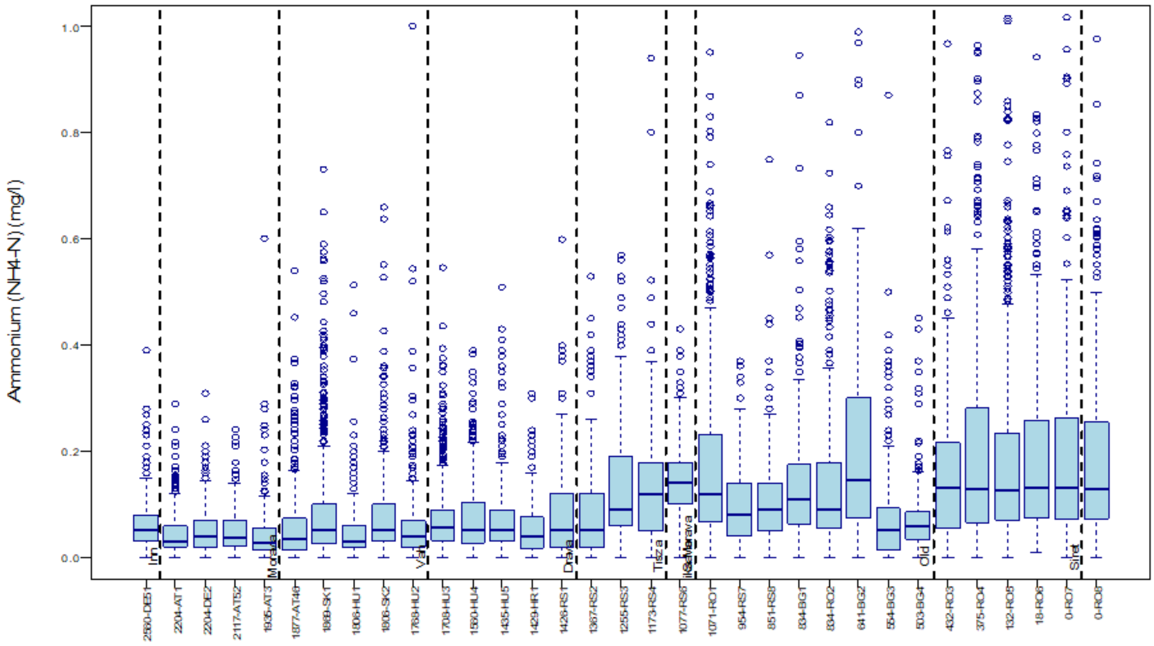


Source: JRC analysis, Udias et al. (under review)

Ammonium (NH4-N)

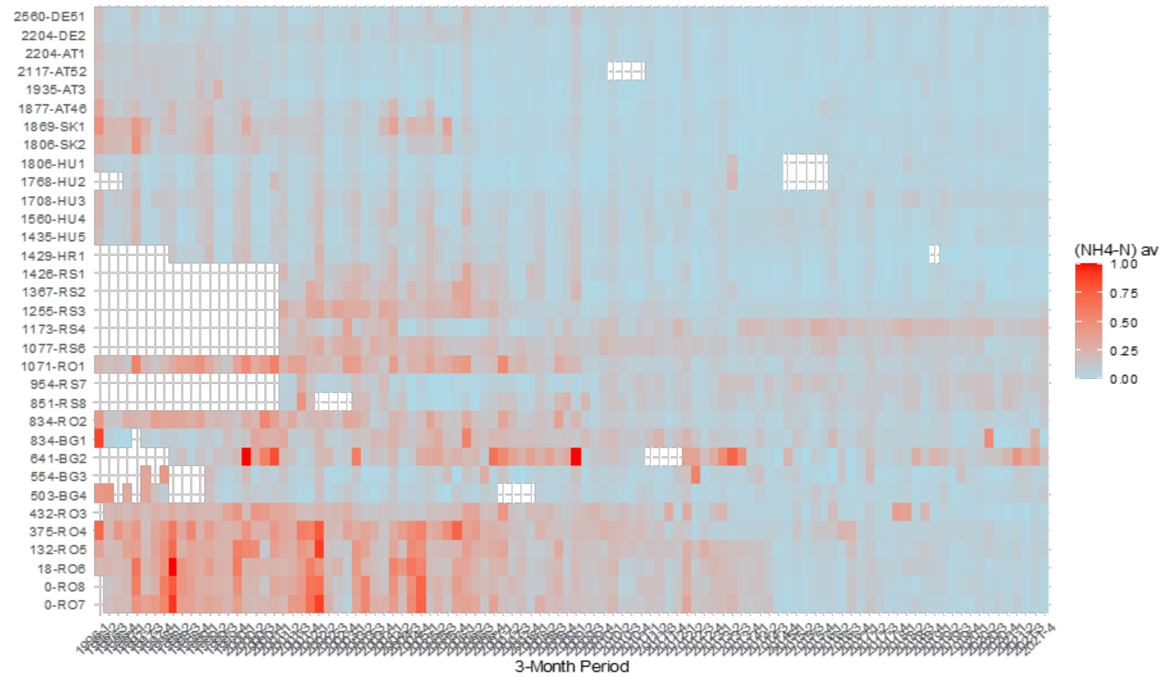
Regarding ammonium (NH4-N) concentrations, both the boxplot (Figure 2.18) and the heatmap (Figure 2.19) indicate a notable increase along the river course. This increase is particularly pronounced at the confluences of major tributaries, such as the Tisza, Sava, and Drava, areas that coincide with high population density. Additionally, the heatmap (Figure 2.19) also shows a general improvement in NH4-N concentrations over the 25-year study period.

Figure 2.18. Boxplot illustrating the concentration distributions of NH4-N for the monitoring stations along the main course of the Danube River after data preprocessing. Each station is identified by a number indicating its kilometer point along the river, followed by an identifier corresponding to the country in which it is located. Also context regarding its relative position to the confluences of major tributaries and the two primary reservoirs: Gabčíkovo and Iron Gate.



Source: JRC analysis, Udias et al. (under review)

Figure 2.19. Heatmap showing the 3 month average concentration of NH4-N from 1996 to 2021 in 35 monitoring stations along the Danube after data preprocess. Each station is identified by a number indicating its kilometer point along the river, followed by an identifier corresponding to the country in which it is located.

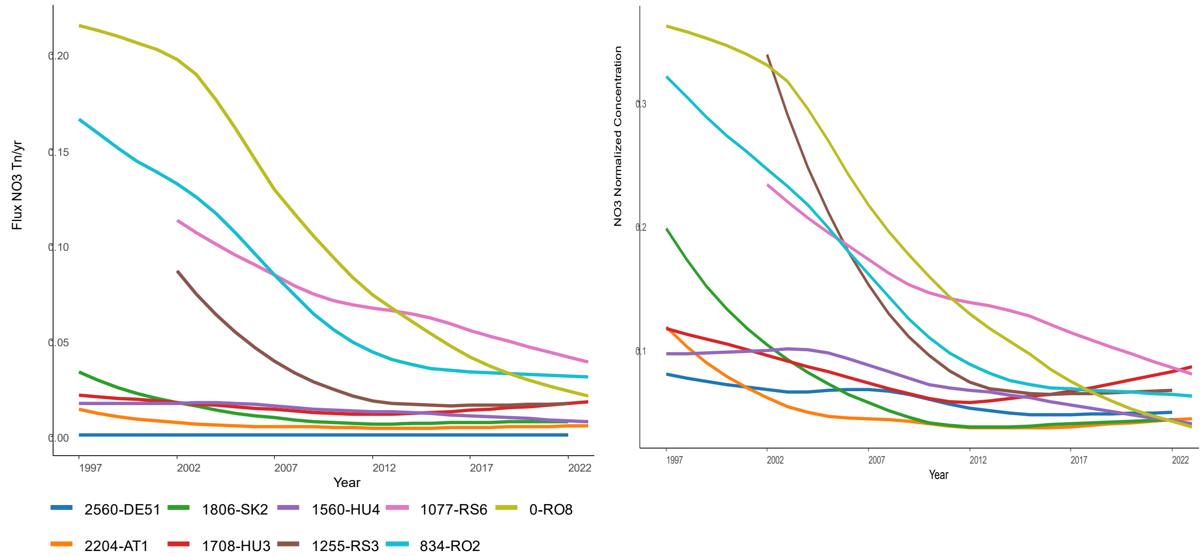


Source: JRC analysis, Udias et al. (under review)

We analyzed temporal trends in NH4-N, considering flow-normalized data, in order to mitigate the influence of seasonality in the analysis (Figure 2.20). The results show significant reductions in ammonium fluxes and concentrations throughout the entire Danube in the 1996 to 2021 period, with these decreases being more intense before year 2013 (probably linked to the widespread implementation of the EU Urban Waste Water Treatment Directive (UWWTD). These improvements are particularly relevant in the final region of the Danube, which initially exhibited the highest levels; in this section, the normalized mean NH4-N concentration decreased from approximately 0.32 mg/L in 1996 to about 0.07 mg/L in 2021 (Figure 2.20, right panel). Analyzing the ammonium flux reaching the Black Sea, the quantity of ammonium discharged by the Danube decreased from approximately 180 t/year in 1996 to 40 t/year in 2021 (Figure 2.20, left panel).

Considering flow-normalized data in the major tributaries, the trends of decreasing NH4-N concentration are similar, with a greater intensity of reduction also observed before year 2012. The Sava River shows concentration changes very similar to those of the Danube, decreasing from 0.3 mg/L in 1996 to approximately 0.04 mg/L in 2021. The Drava, Tisza, and Inn rivers started from lower initial concentrations (approximately 0.1 mg/L in 1996, reaching about 0.06 mg/L in 2021). Conversely, the Morava and Prut rivers showed initial concentrations that were approximately double (approximately 0.6 mg/L in 1996, decreasing to about 0.08 mg/L in 2021). A particular case is the Velika Morava River, where an increase in NH4-N concentration is observed, rising from approximately 0.09 mg/L in 1996 to about 0.2 mg/L in 2021.

Figure 2.20. Temporal evolution of normalized NH4-N fluxes (left) and concentrations (right) for various stations along the main channel of the Danube River from 1996 to 2021.

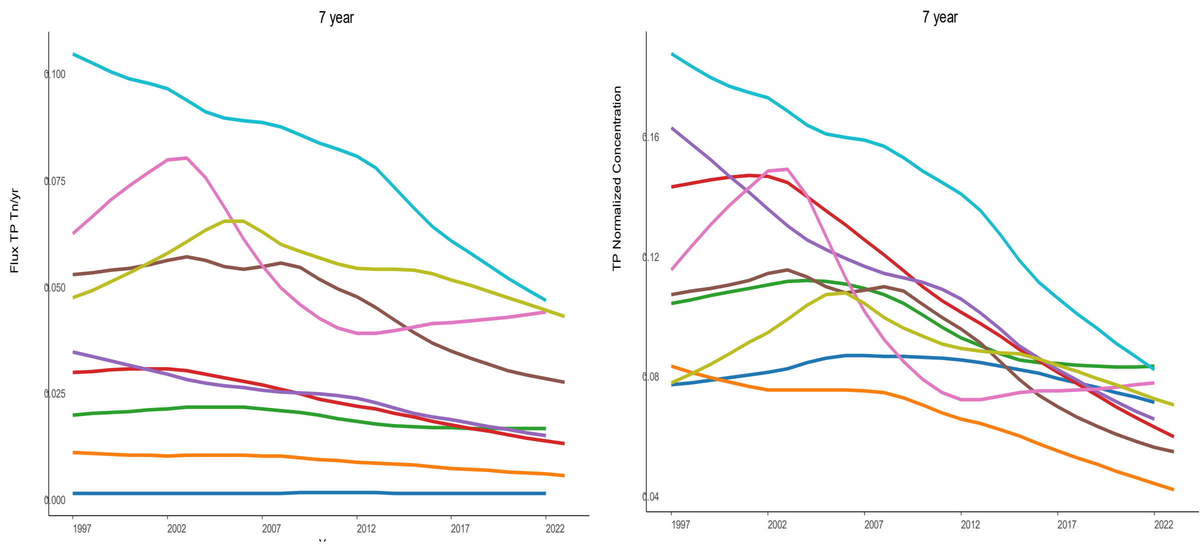


Source: JRC analysis, Udias et al. (under review)

Total Phosphorus (TP) and Orthophosphate (PO4-P)

Significant improvements were also recorded for phosphorus. Normalized Total Phosphorus (TP) loads and concentrations decreased by approximately 40% at many stations between 1996 and 2021. This reflects improvements in wastewater treatment (P removal) and likely the impact of reductions or bans on phosphates in detergents. FN TP concentrations in the upper Danube were around 0.08 mg/L in 1996, increasing in the central region before decreasing again towards the mouth (Figure 2.21). The estimated TP flux to the Black Sea was approximately 50 t/year in 2021. Orthophosphate (PO4-P) trends were similar to those for TP, with generally lower concentrations.

Figure 2.21. Temporal evolution of normalized TP flows (left) and concentrations (right) for various stations along the main channel of the Danube River from 1997 to 2021.

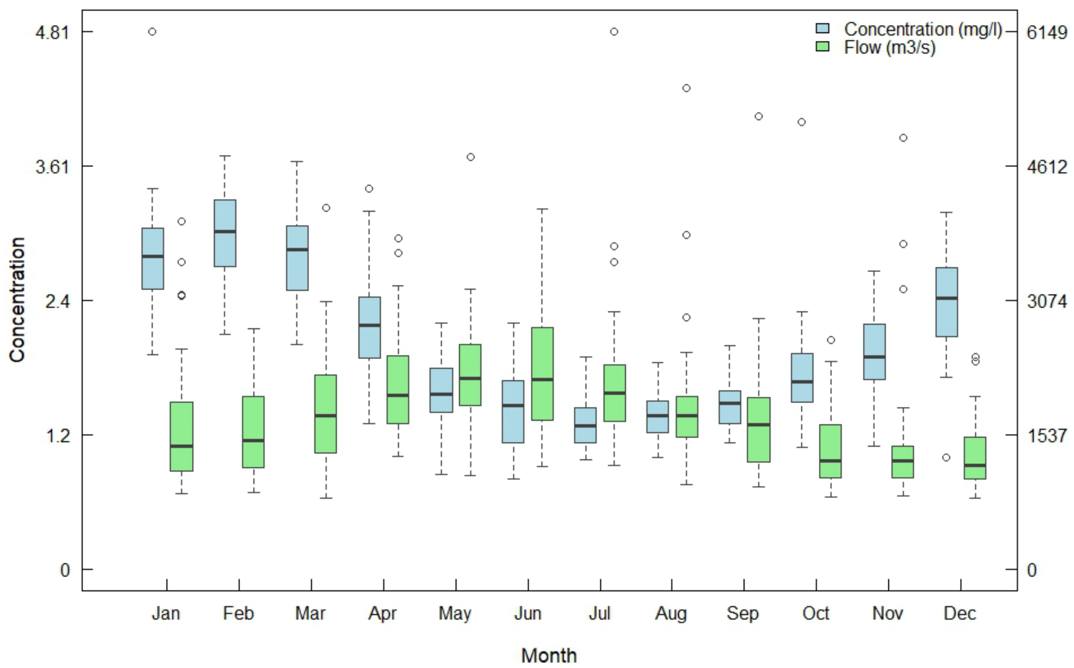


Source: JRC analysis, Udias et al. (under review)

Seasonal Variations in Nutrient Dynamics

All analyzed nutrients exhibited significant seasonal variation, with peak concentrations often two to three times higher than the lowest annual concentrations (exemplified for nitrates in Figure 2.22). Normalized concentration time series consistently showed the highest nutrient levels during winter (January-February) and the lowest during summer (July-August). This pattern is strongly influenced by, and generally out of phase with, streamflow seasonality, as river discharge typically peaks between May and August and is lowest between December and February. This inverse relationship suggests a dilution effect during high-flow periods and nutrient concentration during low-flow periods

Figure 2.22. Boxplot showing the distribution of monthly average nitrate concentrations (blue, left scale) and river discharge (green, right scale) throughout the year at the Hainburg station from 1996 to 2021



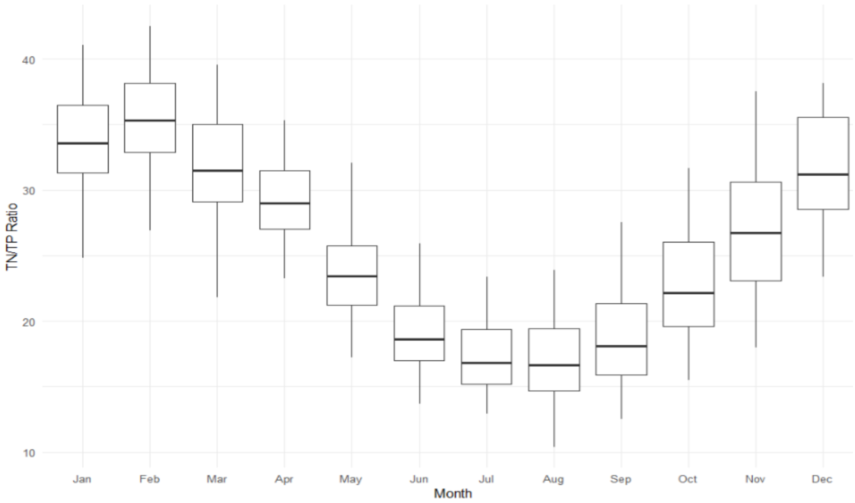
Source: JRC analysis, Udias et al. (under review)

While the opposing seasonality of flow (high in summer) and concentration (low in summer) might suggest relatively uniform nutrient fluxes throughout the year, this is counteracted by temporal lags between changes in flow and the corresponding concentration response. A delay of approximately one month is observed between high flow onset and the subsequent decrease in concentration, while a longer delay of two to three months occurs between low flow onset and the subsequent increase in concentration. Consequently, the maximum nutrient fluxes are generally observed in March, lagging approximately one month behind the peak concentrations. Minimum nutrient fluxes typically occur in October, lagging two to three months behind the minimum concentrations.

Although substantial, the seasonal variation in fluxes (maximum annual fluxes are typically 50% to 70% higher than minimum fluxes, depending on the nutrient) is less pronounced than the two- to three-fold variation observed in concentrations.

The seasonal dynamics also significantly influence nutrient stoichiometry, particularly the TN/TP ratio. While the ratio calculated from the maximum observed concentrations of TN and TP might exceed 45, the typical seasonal fluctuation of the ratio itself ranges considerably throughout the year, commonly oscillating between minimum values of 10-15 and maximum values of 35-40 (Figure 2.23).

Figure 2.23. Boxplot of normalized monthly TN/TP concentration ratio at station 1869-SK1 for the period 1996-2021.



Source: JRC analysis, Udias et al. (under review)

This pronounced seasonal oscillation in the TN/TP ratio primarily stems from a phase difference in the seasonal cycles of TN and TP. TP concentrations typically start decreasing about a month earlier than TN in mid-winter (mid-January vs. mid-February), causing the TN/TP ratio to peak between late February and mid-April. Conversely, TN concentrations begin their increase in early August, while the TP increase timing is more variable (shifting from early June to late April over the study period). This interplay results in the TN/TP ratio rising from its minimum around July or August.

Discussion

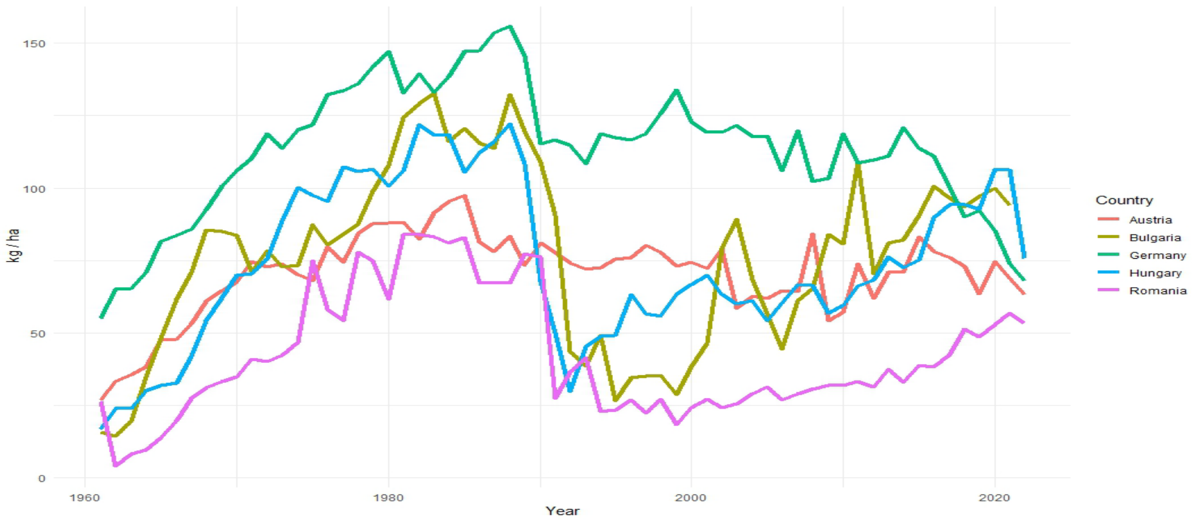
While the comprehensive flow-normalization provided by WRTDS offers a distinct lens to evaluate nutrient dynamics, several other investigations employing different analytical techniques corroborate the general direction of our observed reductions. For instance, Frîncu & Iulian (2020), using Spearman rank correlation on annual means for the Lower Danube (1996-2017), similarly identified decreasing trends for both NO3-N and NH4-N at most Danube stations. Ţuchiu (2019) also reported a temporal decrease in NH4-N concentrations across all studied stations in the Romanian Lower Danube (2006-2015) and a decreasing NO3-N trend at Reni, aligning with our WRTDS-derived reductions for these parameters. Even studies like Salvai et al. (2022), which predominantly found no significant trend for NO3-N in the Serbian Middle Danube (2004-2018) based on Mann-Kendall tests likely on raw concentrations, did confirm significant downward NO3-N trends at several key locations, suggesting localized improvements captured even without full flow normalization. Conversely, observations of 'rather stable' NO3-N concentrations from reviews like Mănoiu & Crăciun (2021), potentially based on TNMN yearbook data using raw averages, or the spatial increase in mean annual NO3-N noted by Țuchiu (2019) downstream in the Lower Danube, may reflect the confounding influence of hydrological variability when flow-normalization techniques like WRTDS are not applied.

As seen in Figure 2.24, Germany exhibits the highest nitrogen inputs among the countries studied, which aligns with the elevated nitrate concentrations observed in the monitoring stations located within Germany. Notably, station 2560-DE51, situated in Germany, shows the highest nitrate concentrations (Figure 2.16, right panel). This correlation underscores the influence of national nitrogen input levels on local water quality, highlighting the direct impact that high agricultural inputs can have on riverine nitrate levels.

However, Figure 2.16 indicates only modest declines in nitrate concentrations, approximately 15%, in these areas. This can be explained by the concept of "catchment legacies and time lags" (Van Meter & Basu, 2017; Liu et al., 2024). Nitrogen applied to agricultural land, especially during periods of high surplus before the 1990s, does not immediately impact water bodies. Instead, a significant portion is stored in soil organic matter, the unsaturated zone, and groundwater systems, forming a "legacy" pool (Van Meter et al., 2016; Wang et al., 2025).

The release of this legacy nitrogen into rivers is a slow process, characterized by considerable time lags that can span years to decades (Meals et al., 2010; Vero et al., 2018). Consequently, the decreasing nitrate trends observed in the Danube from 1996 to 2021 likely reflect, in part, the delayed impact of the substantial reductions in nitrogen inputs that occurred in the early 1990s (Figure 2.24). The river system is still responding to these past changes in nitrogen loading, with legacy nitrogen gradually being flushed out of the catchment.

Figure 2.24. Evolution of nitrogen application in 5 main Danube countries extracted from FAO data.



Source: JRC analysis, FAOSTAT data accessed on 15 May 2025

Concurrently, policies such as the EU Nitrates Directive, aimed at improving nitrogen use efficiency and reducing N surpluses from agriculture, have been in effect throughout much of our study period. While fertilizer application rates might have stabilized, these directives encourage better management practices (e.g., optimized fertilization timing and rates, cover crops) that can reduce the fraction of applied nitrogen lost to the environment (Grizzetti et al., 2025; Bouraoui & Grizzetti, 2014). These ongoing improvements in agricultural management, even with stable N inputs, would contribute to a gradual reduction in the net N surplus entering the soil and groundwater systems, further supporting the observed downward trend in riverine nitrate concentrations.

Conclusions

The long-term assessment of nutrient fluxes in the Danube Basin yields the following politically relevant takeaways for future water management:

- Flow normalization validates policy impact: Advanced methodologies successfully quantified actual nutrient trends by filtering out the "noise" of hydrological variability and declining streamflows, providing a reliable measure of water quality policy effectiveness.
- Point-source controls are a major environmental success: Interventions targeting point-source emissions, driven largely by the UWWTD, have been highly effective, reducing phosphorus (TP) and ammonium (NH4) loads by more than 50% across extensive river sections.
- Reduced pressures on the Black Sea: European policies have successfully alleviated eutrophication pressures on the receiving marine environment. Over the last 25 years, total nutrient loads delivered to the Black Sea dropped by 38% for Total Nitrogen and 51% for Total Phosphorus.
- Agricultural legacies delay nitrate recovery: The much slower decline in nitrate levels highlights the inertia of diffuse pollution. Achieving further reductions requires sustained, targeted landscape-level interventions and continuous implementation of the EU Nitrates Directive to gradually deplete historical nitrogen pools stored in soils and groundwater.
- Uneven nutrient abatement poses new ecological challenges: The rapid success in reducing phosphorus compared to the slower, legacy-hindered decline of nitrogen has amplified seasonal imbalances in the natural N:P ratio (stoichiometry), creating complex new challenges for managing algal blooms during warmer months.

3 Is the limit of manure application (170 kg N/ha) appropriate in view of water, soil and air quality objectives?

The Nitrates Directive (ND, Council Directive 91/676/EEC) mandates the designation of Nitrate Vulnerable Zones (NVZs) in catchment areas prone to eutrophication. Within these zones, the Directive enforces the adoption of best environmental practices and sets a maximum limit of 170 kg nitrogen per hectare per year from livestock manure. Considering current environmental and climate challenges and objectives of the EU, is this nitrogen limit still sufficient or appropriate to protect water, soil and air quality? Our analysis delves into whether this threshold remains effective in mitigating agricultural pollution and aligns with contemporary environmental policy objectives for water, soil and air quality.

3.1 Water

The Water Framework Directive (WFD, 2000/60/EC) establishes a framework for protecting and enhancing aquatic ecosystems, as well as prompting the sustainable management of water resources, and sets the environmental objective of achieving good status for all water bodies within the EU, including rivers, lakes, groundwater, transitional and coastal waters. Nutrient conditions contribute to achieving Good Ecological Status (GES) (WFD, Annex V). For surface water, the appropriate nutrients such as nitrogen and/or phosphorus and their thresholds deemed to support GES are established at the national level by each Member State[1]. Boundaries of Biological Quality Elements (BQEs) (not nutrient boundaries) are compared and harmonised through an intercalibration process (Birk et al., 2013; Poikane et al., 2014b; Kelly et al., 2014), but there are no European wide nutrient targets per water body types. In groundwater, nitrates concentration should not exceed 50 mg/L to be in good chemical status (Groundwater Directive 2006/118/EC, Annex 1).

Nikolaidis et al. (2022) derived European-wide phosphorus and nitrogen targets for rivers and lakes from Member States national targets and compared Total Nitrogen (TN) observed values (from EEA Waterbase) with the respective Europe wide targets. For rivers, it was found that 41% of the monitoring sites achieve GES while 59% were less than good (Figure 3.1). Continued efforts are therefore needed in order to meet environmental objectives. In addition, comparing national nutrient thresholds Nikolaidis et al. (2025) found considerable variation among countries ranging from less than 1 mg/L to over 5 mg/L for river nitrate-N (NO_3-N) for example (Figure 3.2). In contrast, the variation between the WFD and the ND in nitrate standards was typically less than among countries. While concentrations of N and P, both dissolved and total fractions, are the most directly relevant for the ecological health of systems, how these relate to loads from the surrounding catchment depends on precipitation, other hydrogeomorphological factors and management (see section §2.3).

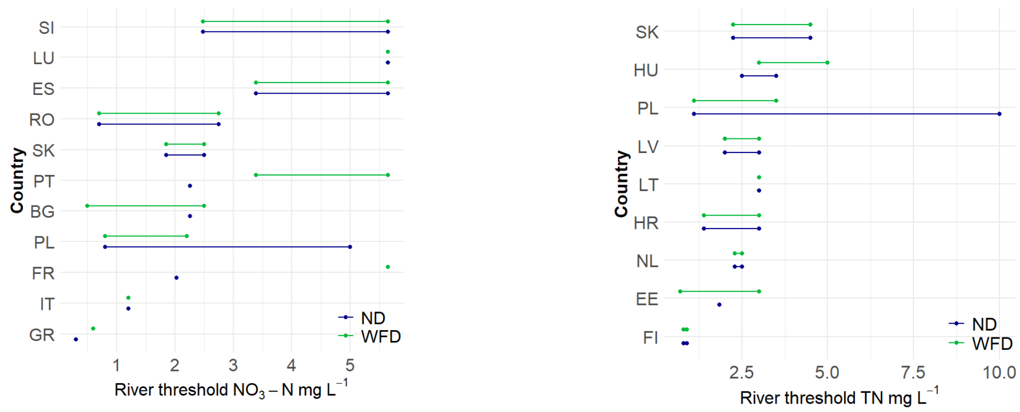
[1] The WFD does not specify which nutrients should be used, as a result, many countries use only P, not N, for inland waters and only N for coastal waters.

Figure 3.1. Gap analysis for total nitrogen (mg/L) for river types in Europe from Nikolaidis et al. (2022). River types RT1-RT20 are described in Nikolaidis et al. (2022).

TN Gap Analysis - Average Statistics per Body Type								
	Total #	Average TN	Target	Average Gap	# Good Status	# Bad Status	% Good Status	% Bad Status
RT1	683	2.872	2.1	0.772	283	400	41.4	58.6
RT2	1034	2.922	2.5	0.422	572	462	55.3	44.7
RT3	417	3.94	2.1	1.84	154	263	36.9	63.1
RT4	1854	3.847	2.5	1.347	705	1149	38.0	62.0
RT5	512	3.825	2.5	1.325	229	283	44.7	55.3
RT6	182	1.021	2.8	-1.779	169	13	92.9	7.1
RT7	75	2.984	3.5	-0.516	53	22	70.7	29.3
RT8	599	2.036	2.4	-0.364	431	168	72.0	28.0
RT9	171	2.11	2.1	0.01	110	61	64.3	35.7
RT10	504	2.435	1.6	0.835	218	286	43.3	56.7
RT11	209	2.559	2.2	0.359	138	71	66.0	34.0
RT12	17	0.79	0.8	-0.01	15	2	88.2	11.8
RT13	1	0.658	1.8	-1.142	1	0	100.0	0.0
RT14	285	1.481	1.3	0.181	191	94	67.0	33.0
RT15	169	1.736	0.9	0.836	44	125	26.0	74.0
RT16	231	1.176	1	0.176	131	100	56.7	43.3
RT17	611	3.876	0.8	3.076	66	545	10.8	89.2
RT18	816	3.259	0.5	2.759	73	743	8.9	91.1
RT19	379	3.277	0.7	2.577	70	309	18.5	81.5
RT20	104	4.807	1.1	3.707	9	95	8.7	91.3

Source: Nikolaidis et al. (2022)

Figure 3.2. River NO_3-N and TN thresholds for the WFD and ND where reported by countries (Nikolaidis et al., 2025).

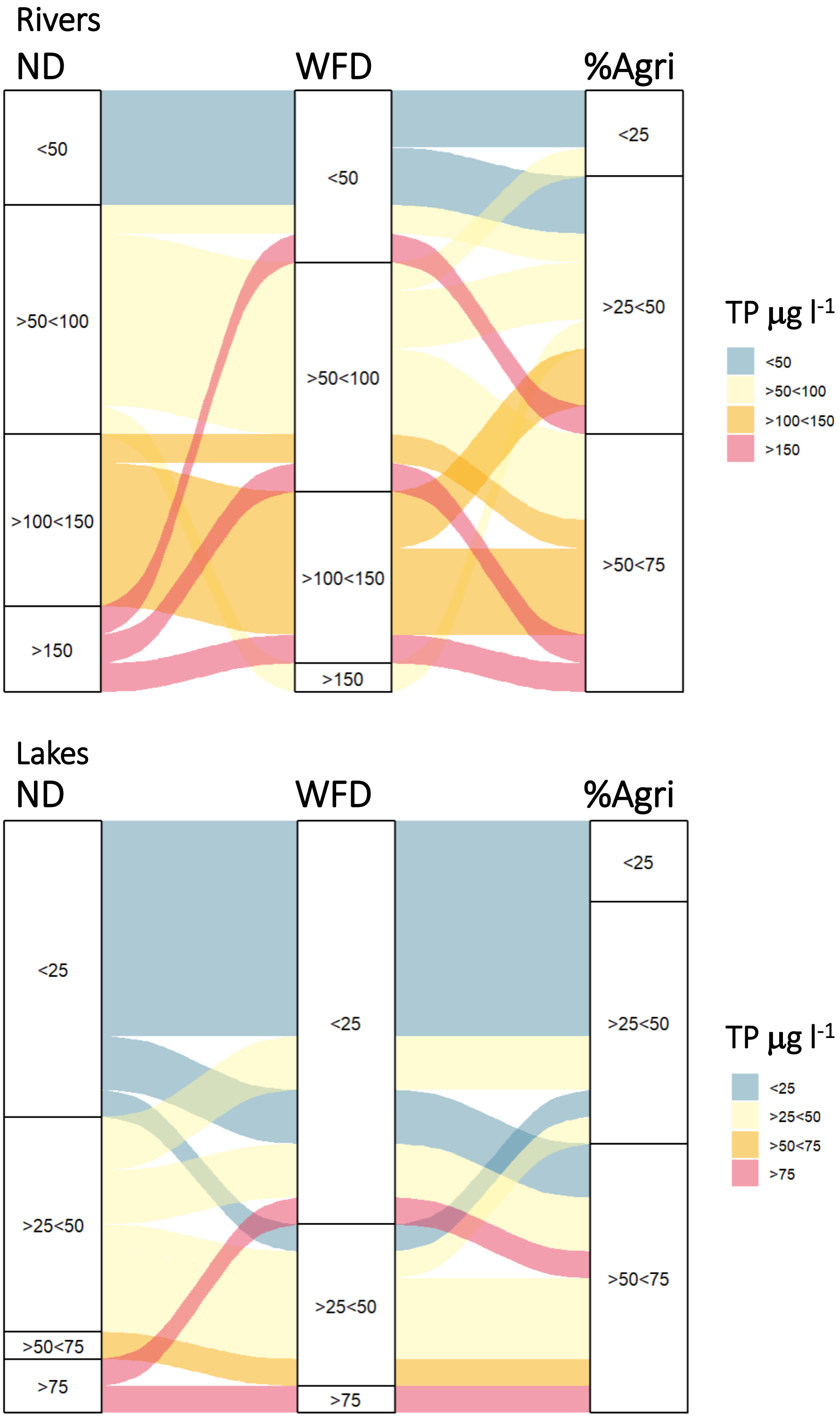


Source: Nikolaidis et al. (2025)

The report by Nikolaidis et al. (2024) compared the contrasting approaches of the Nitrates Directive and the WFD to assess eutrophication status, comparing for the first time the different physico-chemical and biological criteria. It was found that inconsistent nutrient assessment criteria and thresholds were used between directives to assess eutrophication, often with nitrogen thresholds being set to higher drinking water standards (50 mg nitrate/l) in the Nitrates Directive rather than lower and more ecologically relevant criteria, necessary for achieving GES for the WFD. Biological assessment also varied considerably with some member states using only physico-chemical criteria and / or chlorophyll-a values. Major inconsistencies exist among countries in the thresholds for nutrients for both directives with higher values used by countries with higher percentages of agriculture. This can be visualised using a Sankey diagram where the direction and width of flow represents connections between groups revealing clear relationships between the standards set for both directives and the percentage of agriculture nationally (Figure 3.3). Standards therefore seem to relate more to prevailing national conditions than target based environmental objectives. There is a need to review and place thresholds appropriately to support good ecological status.

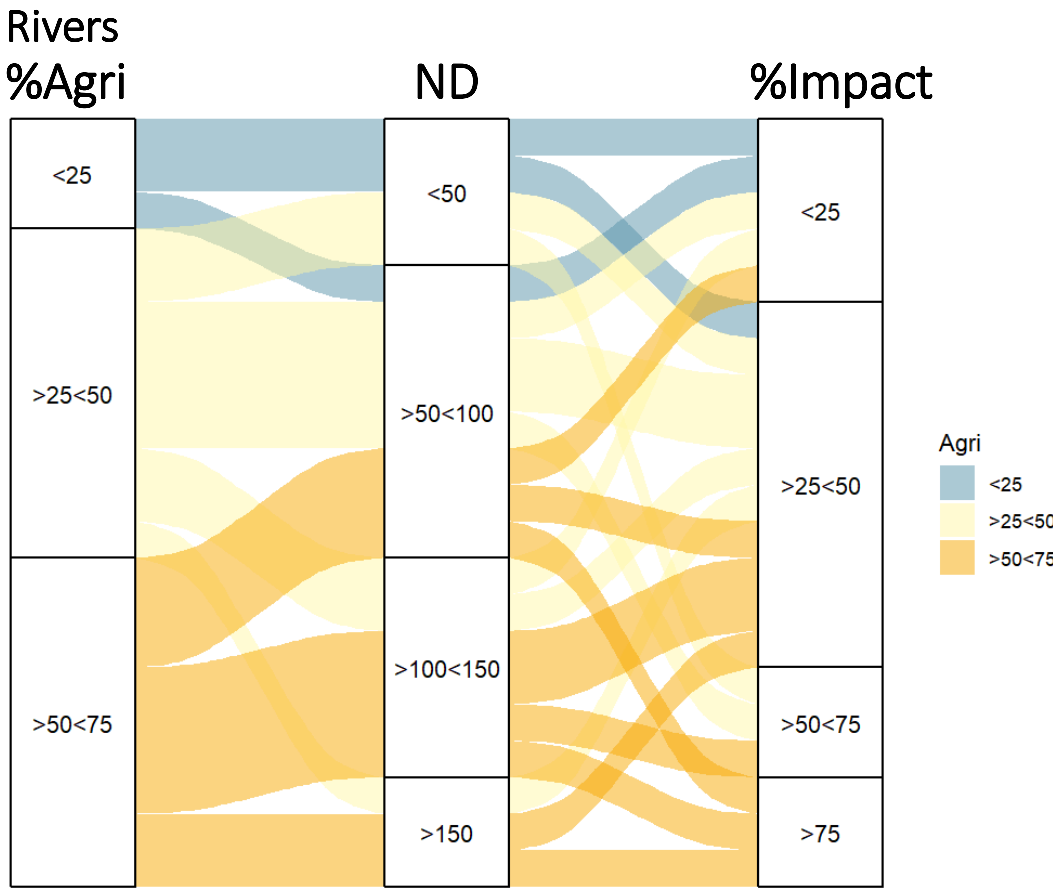
One way of further examining the implementation is to look at the overall pattern across Member States of percentage agriculture, the nutrient thresholds they apply (most commonly for total phosphorus for the WFD) and whether an impact is subsequently reported under the WFD. The latter can be taken from the 2nd RBMP reporting available through the EEA (https://www.eea.europa.eu/data-and-maps/dashboards/wise-wfd). If nutrient thresholds are set too high then there may be evidence of an under reporting of impact. In Figure 3.4 we can see that there is a broad pattern for rivers for countries with higher percentage of agriculture to have higher TP thresholds and also higher percentage of rivers impacted. However, looking at Figure 3.4, there is also evidence, many upward moving lines, where high nutrient (TP) thresholds are set in countries having over 50% agriculture coverage coinciding with a lower reported impact between 25 and 50%. While other parameters and the biological quality element response also determine classification of impact, the ability of Member States to set their own values for nutrient thresholds with limited oversight is likely to be a weakness in determining and achieving environmental objectives. Having and reviewing EU upper limits on nutrient use as part of current and future revisions of the Nitrates Directive is therefore a necessity in policy toolkits.

Figure 3.3. Sankey diagram showing lower limits of Total Phosphorus (µg l^{-1}) for national standards for the nitrates directive (ND) and the Water Framework Directive (WFD) and the percent of Agriculture nationally. Note higher (double) categorical values for rivers (upper graph) than for lakes (lower graph). In Sankey diagrams the direction and width of flow represents connections between groups.



Source: Nikolaidis et al. (2025)

Figure 3.4. Sankey diagram showing percentage of agricultural land, lower limits of Total Phosphorus (µg l^{-1}) for national standards for the nitrates directive (ND) and the percentage significant impact reported for rivers in the 2nd RBMP for nutrient pollution. In Sankey diagrams the direction and width of flow represents connections between groups.



Source: Nikolaidis et al. (2025)

3.2 Soil

Nutrients in European soils

The recent report on the State of Soils in Europe (Arias-Navarro et al., 2024) describes the state of soil conditions in the EU, including the excess and deficiencies of nutrients such as nitrogen and phosphorus.

Soil nitrogen (N) content ranges mostly between 1 and 2 g kg^{-1} in EU topsoils (Ballabio et al., 2019). Due to the high mobility of nitrate, N losses from the soil are highly correlated with the N surpluses (inputs minus crop uptake). There is a large variation in N surplus across the EU+UK from nearly 0 up to more than 150 kg N ha^{-1} yr^{-1} (de Vries et al., 2021). High N surplus is mostly located in areas with high N inputs, especially in intensive livestock areas, except for some regions with low (Poland) or high (Massif central in France, Ireland and the UK) N use efficiency (de Vries et al., 2021).

Between 1930 and 1990, N surplus increased by a factor of 2–3 (Batool et al., 2022), reaching its highest value around 1990 because of a peak in N inputs, but it declined in subsequent years. After

1990, total N inputs on croplands have been relatively stable with a slight increase from 138 to 145 kg N ha-1 yr-1 until 2021 (Einarsson et al., 2021).

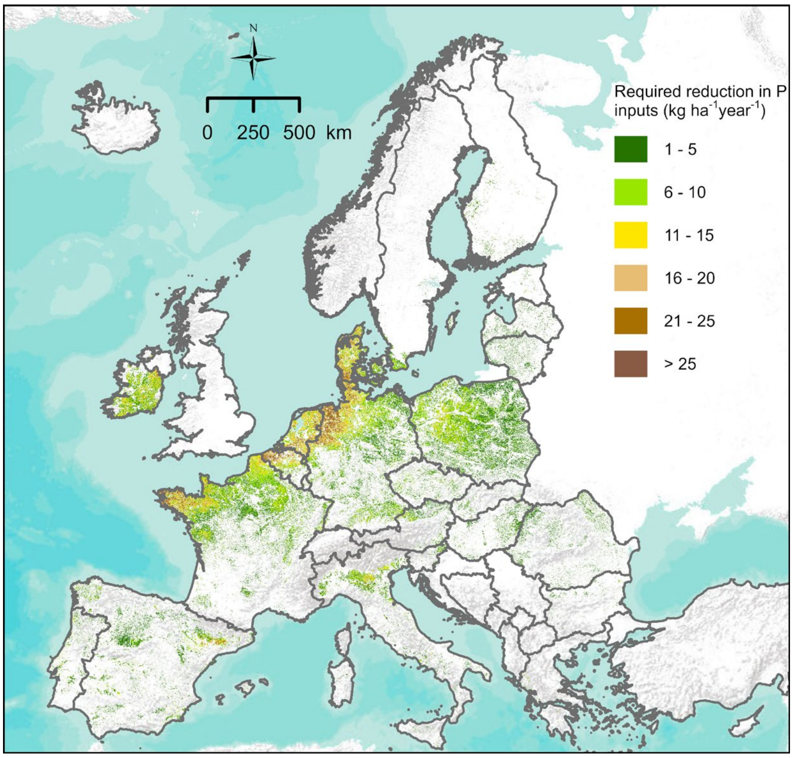
Available phosphorus (P) concentrations in topsoils vary considerably across the EU+UK, with an average of about 33 mg P kg^{-1} in arable land (based on P-Olsen) (Panagos et al., 2022), but higher levels occur in Northern France, Belgium, the Netherlands, Northern Germany, Denmark, UK, Ireland, North Italy and Poland (Ballabio et al., 2019). Despite these high soil P levels, balance calculations have shown an average surplus of P application in the EU+ UK between 0.11 and 0.80 kg P ha^{-1} yr^{-1} (Muntwyler et al., 2024; Panagos et al., 2022) or higher (De Vries et al., 2014; Einarsson et al., 2020). However, there is considerable variation among countries and extensive areas in the EU+UK are currently displaying surpluses of more than 10 kg P ha^{-1} yr^{-1}, despite the generally high soil P concentrations in these regions.

Many EU countries have experienced much more imbalanced P management in the past decades, as P inputs have peaked above 30 kg ha-1 around 1980 (Sattari et al., 2012), while P inputs are nowadays on average 16 kg ha-1 in the EU+UK (Panagos et al., 2022). Due to the lower mobility and high retention of P in soils, the positive budgets of the past have resulted in a build-up of soil P pools representing a large soil P legacy (Sattari et al., 2012). When taking 15, 30 or 50 mg kg-1 P-Olsen as threshold values for excess (Jordan-Meille et al., 2012; Steinfurth et al., 2022), about 93, 60 or 10 % of agricultural soils in the EU+UK can be defined as P-rich soils with possible adverse impacts on water quality, respectively.

Manure limits and phosphorus management

In European regions with excessive P concentration in soil, P inputs should be lower than P removal by crops, in order to decrease the soil P concentration over time (i.e. so-called mining of soil P). This is also relevant in the light of the EU Directive 2025/2360 on soil monitoring and resilience (Soil Monitoring Law) for which Member States will have to define a maximum value above which soils will be considered unhealthy due to excessive P concentrations. We calculated that in regions with high soil P concentrations (which were defined as being above 40 mg kg-1 P based on Olsen extraction), P inputs need to be reduced (Figure 3.5) by 384 k tons per year, which accounts for 21 % of total P inputs in the EU27 (Van Eynde et al., 2025).

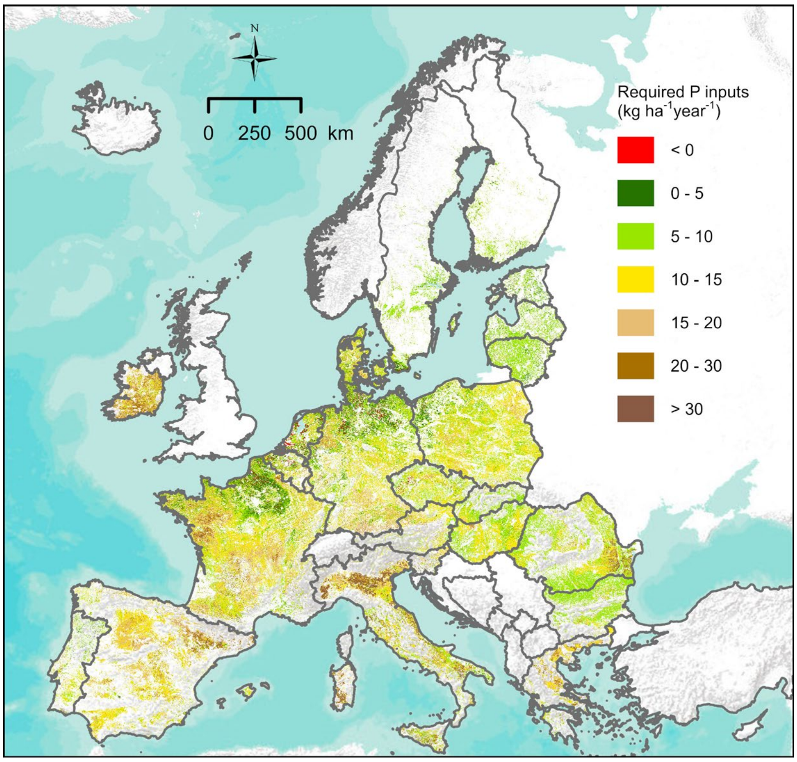
Figure 3.5. The reduction in P inputs in areas with currently high soil P concentrations, in order to reach optimal P concentrations (20 – 40 mg kg^{-1} based on Olsen) by 2050.



Source: Van Eynde et al., (2025)

Currently, there are no EU policies that limit directly the application of P to EU agricultural soils. The Nitrates Directive indirectly limits P application by manure, due to the limit of N application from manure, which is 170 kg ha^{-1} $year^{-1}$. However, using estimates of N-P ratios in different manure types, this limit on N application by the Nitrates directive is estimated to indirectly limit P application up to 27 - 70 kg ha-1 year-1 depending on the type of manure (Vlaamse Landmaatschappij, 2024). These application rates are higher than the required P inputs we calculated to reach optimal P levels by 2050 (Figure 3.6), especially in livestock dense regions such as Brittany (France), Flanders (Belgium), the Netherlands, Catalonia (Spain), Emilia Romagna (Italy), Denmark and Ireland.

Figure 3.6. The required P inputs to reach optimal soil P concentrations by 2050.



Source: Van Eynde et al., (2025)

Some of these regions do have explicit P application limits. For example, Denmark has defined specific P application limits per crop and depending on soil P status, mostly ranging between 15 and 30 kg ha^{-1} $year^{-1}$ (Danish Ministry of Food Agriculture and Fisherie, 2024). Flanders, the Netherlands and Ireland have specified limits also dependent on soil P concentrations and land use (Irish department of Agriculture Food and the Marine, 2024; Rijksdienst voor Ondernemend Nederland, 2019; Vlaamse Landmaatschappij, 2024). For soils with high P concentrations, Ireland has set the limit to zero P application per year, with exceptions for some crops such as potatoes. In Flanders and the Netherlands, maximal allowed application rates for soils with high P concentrations are between 20-30 kg ha^{-1} $year^{-1}$. Sweden applies the rule that not more than 22 kg of P ha^{-1} can be applied through manure (Jordbruksverket, 2024). When maximal allowed application rates exist in national legislations, they are generally higher than required P inputs to reach optimal soil P concentrations by 2050 according to our calculations (Van Eynde et al., 2025). Our calculations showed that for most agricultural land in these regions, between 0 and 20 kg ha-1 year-1 is required. Depending on the type of manure, this P application would mean an N application of maximum 50 - 125 kg ha-1 year-1. This N application contains the EU average maximal allowed N input calculated by De Vries et al. (2021). Our results show that limit values on manure application based on P requirements using the framework applied in this study, would result in lower N application limits than what is currently allowed in the Nitrates Directive.

3.3 Air

Ammonia (NH_3) is a key precursor to the formation of fine particulate matter, known as PM2.5—particles with a diameter of 2.5 micrometers or smaller—which significantly degrades air quality.

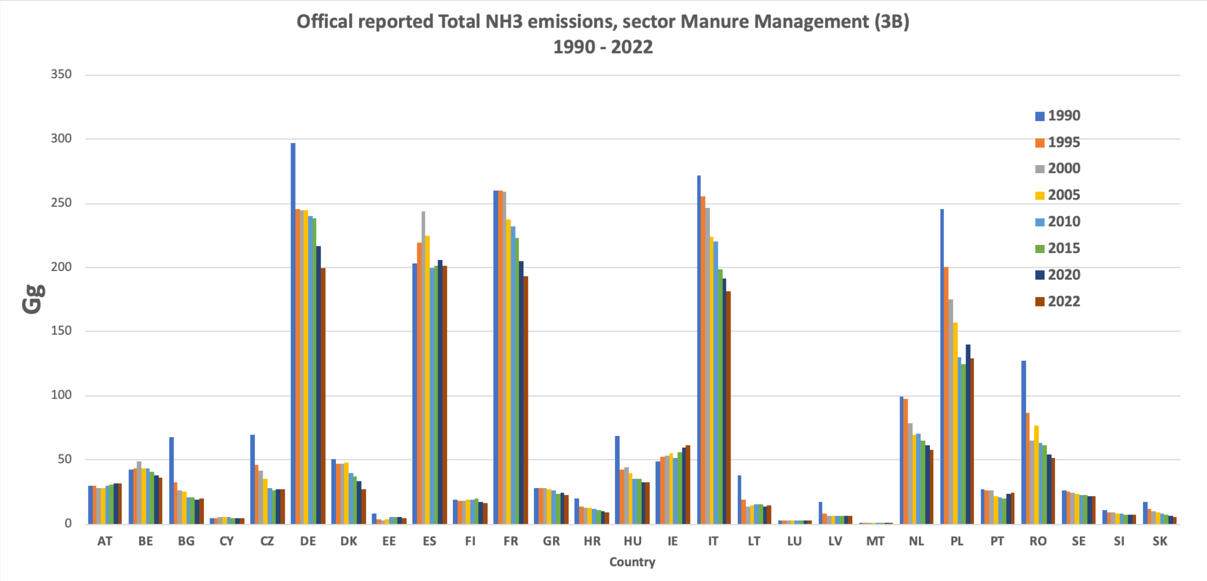
Exposure to PM2.5 is linked to several adverse health effects, ranging from respiratory and cardiovascular diseases, to an increased risk of cancer and premature death (European Environment Agency and Joint Research Centre 2025, Brauer et al. 2021). Ammonia (NH_3) emissions are released in into the atmosphere from various sources, including agricultural activities, such as fertilizer application and livestock manure management, industrial processes and vehicle emissions and plays a leading role in the formation of PM2.5. Pozzer et al., (2017) showed that a 50% reduction in agricultural emissions (NH_3 main contributor with around 80-90% of total agricultural emissions) could have a large impact on air pollution mortality, reducing it by around 8% worldwide, affecting ~250.000 people, and in Europe reducing air pollution mortality by ~19% (i.e. affecting 52.000 people).

In EU27, agriculture is by far the major sector responsible for ammonia emissions, 93% of the total in 2022, with 70% due to livestock and 30% to crops production[2]. While the Nitrates Directive is not targeting air pollution, indirectly the limit of 170 kg nitrogen per hectare per year from livestock manure has had an impact in limiting ammonia emissions from agricultural activities.

Data officially reported by the Parties to the Air Convention (CLRTAP) to the co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe (EMEP) show that there has been a decline of ammonia emissions by the Manure Management sector (source sector 3B) in all EU27 countries between 1990 and 2022, except for Austria, Estonia and Ireland. The highest emissions are found for Germany, France, Italy and Poland (Figure 3.7). However, despite the decline since the 90s, current ammonia emissions are still above the policy objectives and national reduction targets set in EU27 countries by the National Emission reduction Commitments Directive (NECD, 2016/2284/EU) for the periods 2020 – 2029 as well as more ambitious ones for 2030 and beyond (Figure 3.8). Changes in the limit of manure application in the Nitrates Directive could indirectly affect the achievement of ammonia emissions reduction targets for the future.

[2] Ammonia (NH_3) emissions per sectors according to the air pollutant emission inventory reported to EEA by EU Member States under the National Emission reduction Commitments Directive. Source: EEA https://www.eea.europa.eu/en/topics/in-depth/air-pollution/national-air-pollutant-emissions-data-viewer-2005-2022 (access 9 December 2024)

Figure 3.7. Officially reported NH3 emissions (Gg N, 1000 t N) by Manure Management by 27-EU countries between 1990 – 2022.



Source: JRC analysis, data from EMEP [*https://www.ceip.at/webdab-emission-database/reported-emissiondata*](https://www.ceip.at/webdab-emission-database/reported-emissiondata) *(access October 2024)*

Figure 3.8. Percentage emission reductions compared with 2019 levels required by EU Member States to meet their emission reduction commitments for 2020-2029 and 2030 onwards, under the National Emission reduction Commitments Directive.

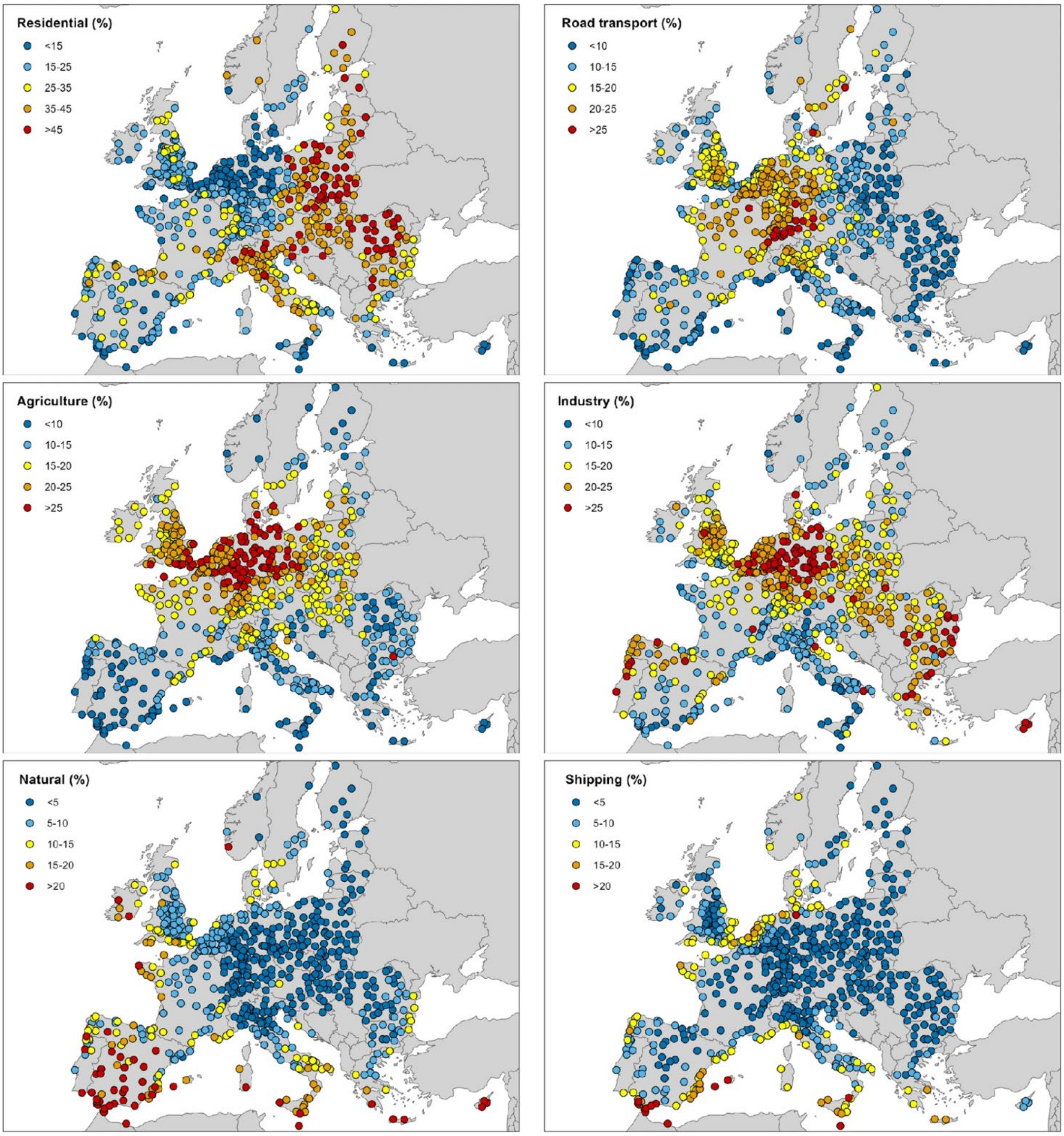
	2030				
Country Name	NH3	NMVOC	NOx	PM2.5	SO2
EU-27	🟠	🟠	🔴	🟠	🟠
Austria	🟠	✓	🔴	🟠	✓
Belgium	✓	✓	🟠	✓	✓
Bulgaria	✓	🟠	🟡	🔴	✓
Croatia	🟡	🟠	🟠	🔴	✓
Cyprus	✓	🟡	🟠	🔴	🟤
Czechia	🟡	🔴	🔴	🟤	🟠
Denmark	🟡	✓	🟠	🟠	✓
Estonia	🟡	✓	✓	✓	✓
Finland	🟡	🟡	🟡	✓	✓
France	🟡	✓	🔴	🟠	✓
Germany	🟠	✓	🔴	🟠	🟠
Greece	✓	🟡	🟠	🟡	🟠
Hungary	🔴	🔴	🔴	🟤	🔴
Ireland	🟡	🟠	🔴	🟡	✓
Italy	🟡	🟠	🟠	🟠	✓
Latvia	🟠	🟠	🟡	🟠	✓
Lithuania	🟡	🔴	🔴	✓	🟡
Luxembourg	🟠	🟠	🔴	✓	✓
Malta	✓	🟠	🟤	🟡	✓
Netherlands	🟡	✓	🟠	✓	✓
Poland	🟠	🟡	🟠	🔴	🟠
Portugal	🟡	🟠	🟠	🔴	🟡
Romania	🟡	🟠	🔴	🟤	🟠
Slovakia	🟠	✓	🟡	✓	🟡
Slovenia	🟡	🟠	🔴	🔴	🟠
Spain	🟠	🟠	🟠	🔴	🟡
Sweden	🟠	✓	🔴	✓	✓

Current emission levels below the emission reduction commitment	✓
Emission reduction needed by less than 10 % from current levels	🟡
Emission reduction needed by 10 % to 30 % from current levels	🟠
Emission reduction needed by 30 % to 50 % from current levels	🔴
Emission reduction needed by more than 50 % from current levels	🟤

Source: EEA https://tableau-public.discomap.eea.europa.eu/views/NECD_Briefing_2021_Table2/Reductioneffortto2020and2030 (access 9 December 2024)

The recent PM2.5 Atlas for Europe, as published in Zauli et al., 2024, shows the air pollution due to PM2.5 and sectoral contribution (on total PM2.5 yearly average concentrations) for major urban areas in EU27, based on 2019 emissions. (Figure 3.9). The residential sector is the largest contributor, accounting for, on average, 27% of local PM2.5 and is the highest source sector in 56% of the cities. The average contribution from agriculture is 17%. The largest contribution from this sector is observed in Germany, where in many cities the agriculture contribution is above 25%. High agriculture contributions are also found in other central and eastern European cities as well as in the United Kingdom, Denmark, Belgium, Netherland and Luxembourg. It is notable that, although emissions from agricultural sources are mainly concentrated in rural areas, secondary inorganic pollution associated with medium- and long-range transport means that this contribution is also very significant in urban areas (Zauli et al., 2024).

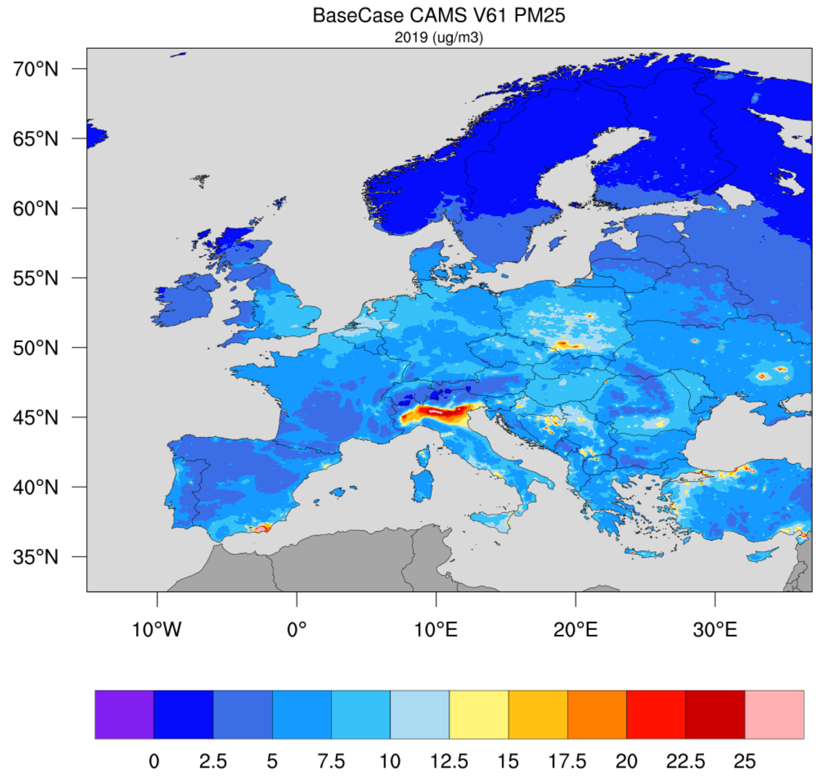
Figure 3.9. Contribution of the individual emissions source sectors (i.e. Residential, Road Transport, Agriculture, Industry, Natural and Shipping) to local PM2.5 concentrations. Each dot represents one urban area.



Source: Zauli et al. (2024)

Overall in EU27, elevated PM2.5 concentrations are found over the Po Valley (Italy), Poland, Romania and the Balkan region, mainly caused by the emissions from residential heating. While in Germany, Benelux, and parts of France, the agricultural sector has an important contribution to the yearly PM2.5 concentrations as mentioned before. The higher concentrations in the southern part of Spain are caused by natural dust (Figure 3.10). This is shown in Figure 3.10 where the yearly average PM2.5 concentrations for 2019 are shown, using the CAMS V61 emission inventory. Also elevated PM2.5 concentrations are found in the Po Valley (Italy) and in Poland. More detailed analysis on PM concentrations in Europe can be found in De Meij et al. (2024).

Figure 3.10. Yearly mean PM2.5 concentrations (µg/m3) for 2019, using CAMS_REG_V61 emissions.



Source: JRC analysis

4 Implications of the limit of manure application (170 kgN/ha) for crop production and livestock density

The Nitrates Directive's limit of 170 kg of nitrogen per hectare per year from livestock manure imposes constraints on manure application in Nitrate Vulnerable Zones (NVZs). This section examines the implications of this limit on crop production and livestock density, utilizing spatial livestock data from the 2020 Farm Structure Survey and nitrogen budget estimates derived from the disaggregated CAPRI model data (2000-2018). Furthermore, we assess the potential economic, environmental, and climate consequences of reducing livestock units (LSU) per hectare of utilized agricultural area (UAA) and decreasing mineral fertilizer application, using scenario-based modelling to explore these effects.

4.1 Livestock density and nitrogen surplus

4.1.1 Spatial livestock data from 2020 Farm Structure Survey

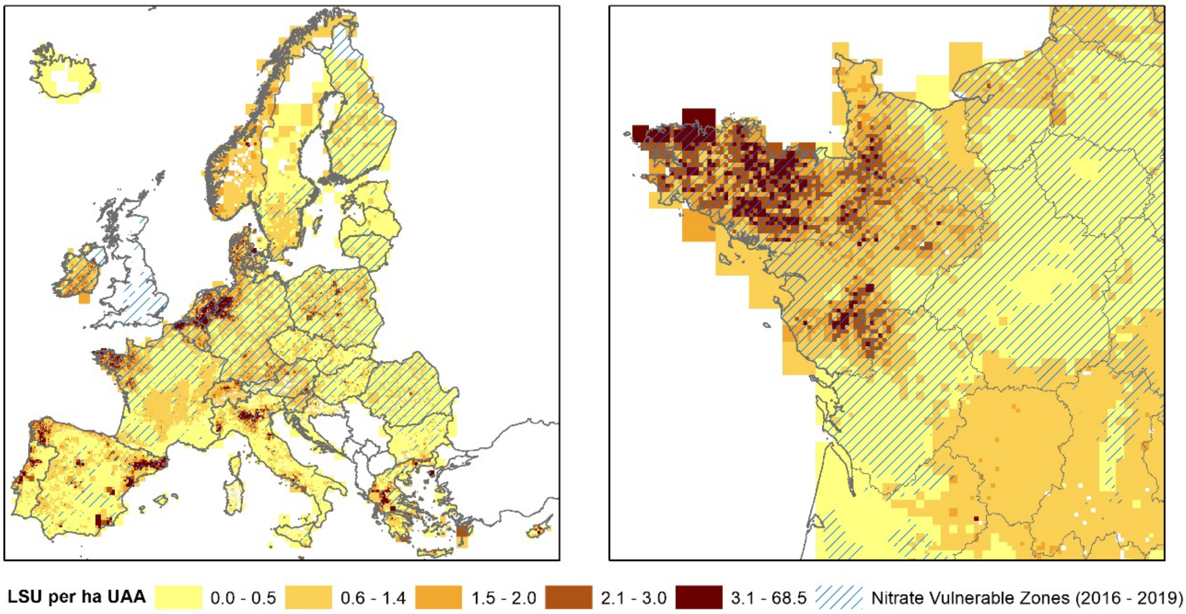
Data on the sales of mineral fertilizers are collected and accessible through Eurostat, while data on nitrogen input from livestock manure are generally inferred from livestock numbers and excretion coefficients, which introduces a higher level of uncertainty. Typically, these data are available at administrative units, making the extraction of spatial information challenging. Nonetheless, spatial information is essential for evaluating environmental impacts.

Here we gather recently developed layers (see Skøien et al., 2025 and Lampach et al., 2025) that spatially grid variables from the 2020 agricultural census. The last agricultural census took place in 2020, collecting more than 300 variables on agriculture and farm structure from 9.03 million farmers in the EU (and EFTA countries Iceland, Switzerland, and Norway). The data was recently published using a multigrid resulting from a method implementing confidentiality rules for statistical disclosure of farm holding data (see: Geospatial data from agricultural census - Eurostat).

The first focus is a set of Context Indicators of the Common Agricultural Policy (CAP), including data on livestock density and numbers (heads). This provides an unprecedented view on livestock in EU agriculture and allows to investigate current livestock numbers in relation to e.g. Nitrate Vulnerable Zones (see Figure 4.1).

This information is relevant in the context of limiting the excess manure from intensive and concentrated livestock production, and the indirect control on livestock density.

Figure 4.2. Livestock density (Livestock units / ha agricultural area) based on gridded Farm Structure Survey data for the year 2020 and Nitrate Vulnerable Zones.



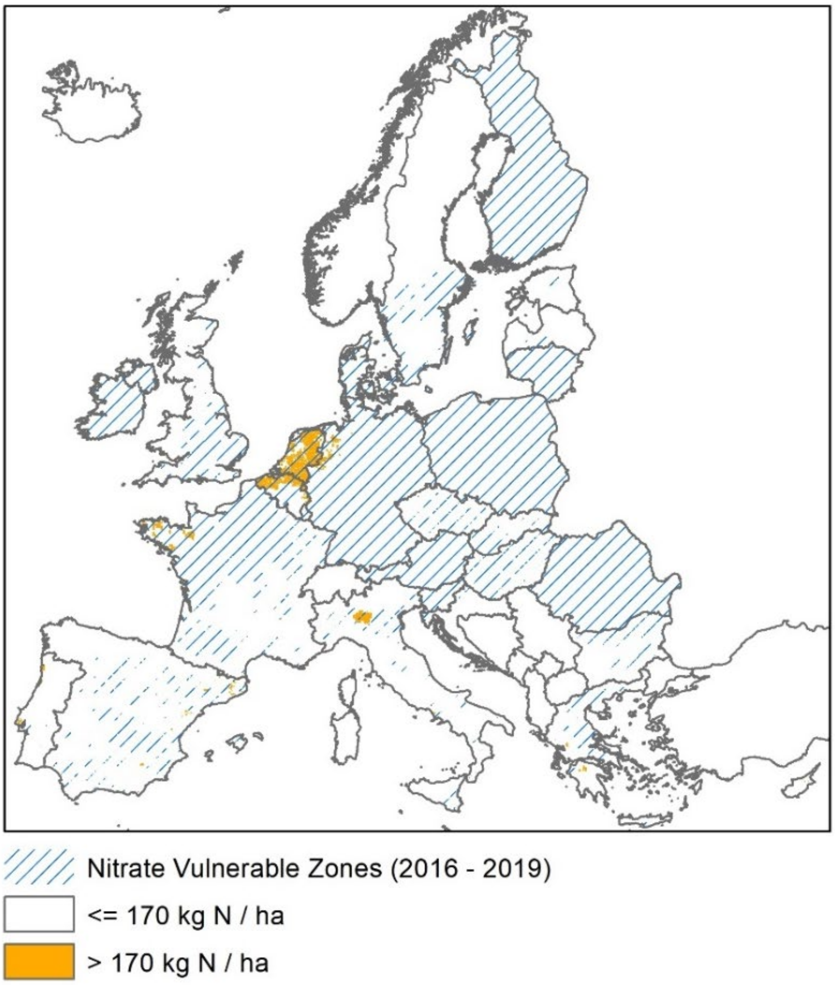
Source: JRC analysis

The data clearly illustrates those areas where livestock densities are very high, including larger parts of the Netherlands, northwest Germany, Bretagne (France), as well as the Po valley. In these areas the production of manure is high, and often more than what surrounding agricultural areas could utilize without surpassing the limit to manure application of 170 kg N/ha per year. This is illustrated by the assessment with the CAPRI (Common Agricultural Policy Regionalised Impact) model below.

4.1.2 Nitrogen budget based on disaggregated CAPRI model data (2000-2018)

Several assessments on the impacts of agricultural, environmental and climate change policies in Europe are investigated using the CAPRI (Common Agricultural Policy Regionalised Impact) model. CAPRI is a partial-equilibrium model that runs at a regional (i.e., NUTS2) level and is used for ex-ante impact assessments of agricultural and international trade policies including regional analyses of Common Market Organisations. Recently, the underlying agri-environmental variables have been spatially disaggregated to homogeneous units called Farm Structure Units for a time series from 2000 to 2018, see publication by Koeble et al. (2025). This harmonized and spatially consistent data set includes crop types (area, production, yield), livestock types (totals and densities) and various parameters associated with nitrogen application and losses. Trading of manure between regions/countries is taken into account. The disaggregated CAPRI model results (livestock, manure, mineral fertilizer, yield etc.) for the timeseries 2000 – 2018 can provide the basis for the assessment of 'hot spots' of concentrated livestock production and excess nitrogen inputs from manure. Figure 4.2 shows the nitrogen inputs from manure applied to agricultural area (net of losses in animal housing and manure storage) and from animals grazing above 170 kg / ha. The spatial resolution of the results corresponds to an area of 12 km^2 on the average (3 years average 2016-2018) against the background of Nitrate Vulnerable Zones. Data on local and regional manure trade or transport are scarce. Hence, the distribution model is based on certain assumptions (i.e. location and share of the grazing animals, maximum distance of manure transport). Nevertheless, it depicts hotspot areas where intensive livestock production poses a risk of application of manure in excess of the 170 kg N / ha limit. For the three regions (Belgium-Flanders, Italy-Lombardy/Piemont and the Netherlands) with larger continuous areas of manure N limits exceedance, derogations have been in place between 2016 and 2018 (derogations in place during 2016-2018: BE-Flanders (9/2015 - 12/2018), DK (5/2017 - 12/2018), NL (5/2014- 12/2017 and 5/2018 - 12/2019), IT-Lombardy/Piemont (6/2016 - 12/2019), IE (2/2014-12/2017 and 2/2018 - 12/2021, no derogation in Brittany, Catalonia and Germany).

Figure 4.2. Manure N application to the field (excluding N lost in housing and storage) and from grazing animals. Average 2016-2018.



Source: JRC analysis

However, in the view of a potential amendment to the Nitrates Directive, which may exclude the accounting of processed manure from the 170kg N/ ha limit because of similar environmental behaviour as mineral fertilizer, limits based on a full nitrogen balance at farm level may be more appropriate. A nitrogen balance, including both, synthetic fertilizer N and manure N inputs to the agricultural area, can have the advantage of

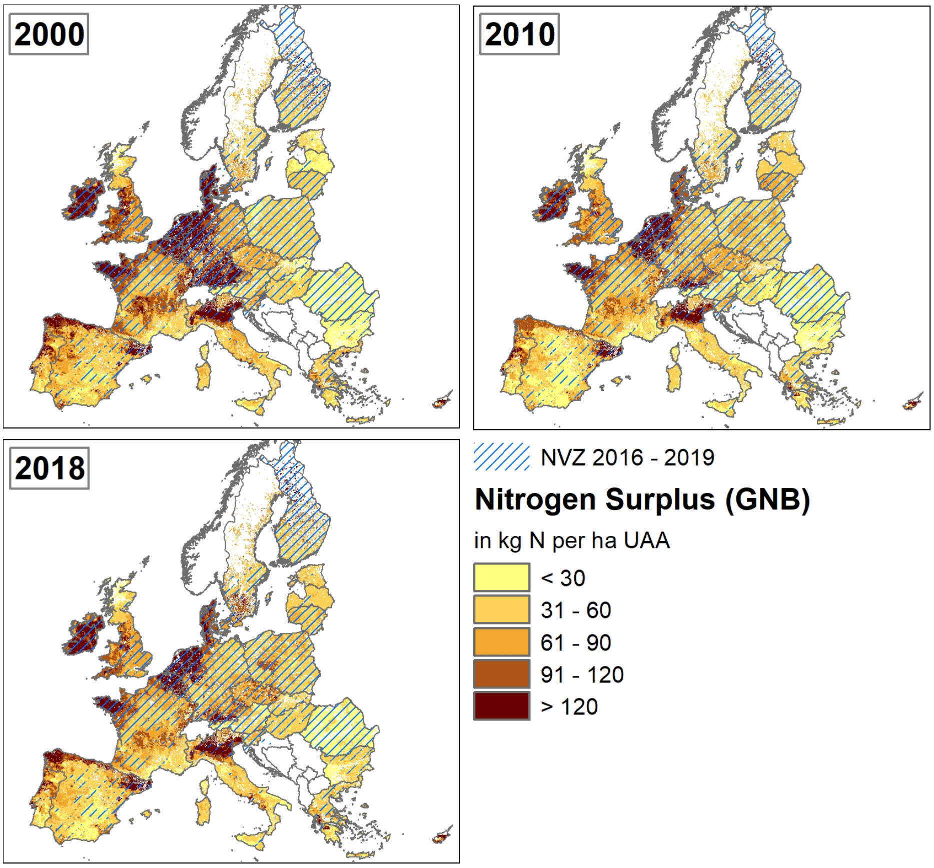
- function as a verification tool to ascertain whether processed manure applied on top of the 170 kg N / ha limit is being utilized as a bona fide one-to-one substitute for mineral fertilizers at farm level. If mineral N does not decrease by an amount equivalent to RENURE N applied, it effectively results in a relaxation of the 170 kg N / ha limit.

- providing a more relevant measure of the environmental impact of N fertilization, regarding nitrogen loss to aquatic systems and to the atmosphere

- incentivise farmers to improve nitrogen-use-efficiency (Regelink et al. 2024);

Figure 4.3 provides the temporal development (2000, 2010 and 2018) of the surplus of nitrogen potentially emitted to the air, leached or run-off to waters or add to nutrient accumulation or depletion in the soil based on the Gross Nitrogen Balance (OECD, 2007; Eurostat, 2013) based on CAPRI disaggregated data. The data suggests that the nitrogen surplus has decreased in wider areas (e.g. Germany, France) with high N surplus between 2000 and 2010, possibly related to restrictions under the Nitrates Directive. Between 2010 and 2018 the situation is rather stable in most areas, with N surplus levels remaining rather high (>=120 kg N / ha) mainly in the regions with high livestock densities (see Figure 4.1). The average nitrogen (N) surplus level in the EU26 (EU member states excluding Croatia) decreased from ~70 kg N ha^{-1} of utilized agricultural area (UAA) by 8 kg N ha^{-1} between 2000 and 2010 due to a decrease in mineral fertilizer input, while the export of N with the harvest increased. However, surplus increased again by 5 kg N ha^{-1} UAA from 2010 to 2018, according to this study. This increase is primarily attributed to the fact that, after 2010, the N input from both mineral fertilizers and manure rose. Particularly the input from mineral fertilizer increased at a faster rate than the increase in N export from the field via harvested products and crop residues. Thus, over the whole period 2000 – 2018 the average surplus in the EU decreased only slightly by ~2-3 kg N ha^{-1} UAA.

Figure 4.3. Nitrogen surplus 2000, 2010 and 2018 based on the Gross Nitrogen Balance (GNB). The N surplus corresponds to N input from mineral fertilizer and manure (taking into account imports and exports) + N input from crop residues + N input from atmospheric deposition + N input from biological fixation minus N removal by harvested products.



Source: JRC analysis

4.2 Effects of reducing manure and mineral nitrogen application

4.2.1 Reducing livestock density – Scenario analysis with CAPRI model

A relevant analysis on the potential economic, environmental, and climate impacts of lower livestock units (LSU) per hectare of utilised agricultural area (UAA) was provided in the context of the 2022 Medium-term EU Agricultural Outlook (EC, 2022, Chapter 7). Chapter 7 explores an environmental scenario where livestock density (LSU/ha UAA) is reduced following requirements of the Nitrates and Habitats Directives. The 2030 CAPRI medium-term baseline is used to benchmark the scenario outcomes. The details of the full analysis can be found in Chapter 7 and will not be discussed here. The text below is directly based and partly copied from EC (2022).

Importantly, the choice of reducing livestock density below 2 LSU/ha UAA (Table 4.1) was motivated by the fact that this livestock density more or less corresponds to the Nitrates Directive limit of 170 kg N manure/ha. This reduction was simulated at (NUTS2) region level (with possibility of compensation between regions) and at 10x10 km grid level (with restricted compensation between grids or regions). Overall, the impacts of lowering livestock density on animal numbers and production are diverse across sectors and EU regions. Impacts are highest on pigs followed by nondairy cattle and lowest for dairy cattle. Absolute reduction of livestock units is high in regions where densities are currently high (dubbed 'hotspot regions' in the report), but since CAPRI covers also price effects, these reductions are partially offset at EU level through increases in other EU regions.

Table 4.1. Scenarios defined for the environmental assessment of the 2022 Medium-term EU Agricultural Outlook that are compared against the 2030 CAPRI medium-term baseline (for details see EC, 2022).

Scenario	**Assumptions**
Scenario S1a	Livestock density ≤ 2 LSU/ha UAA at NUTS2 region level. This is the least ambitious scenario, which will have a rather strong effect in some regions that currently have a high livestock density. It can also induce strong compensation effects in other regions. This scenario provides the smallest effect of the tested lower livestock density.
Scenario S1b	Livestock density ≤ 2 LSU/ha UAA at 10 km x10 km grid level. Compared to 1a, this scenario considers livestock density values at a higher geographical resolution. Overall, scenarios at grid level are likely to be closer to reaching the maximum potential impact of the proposed measure.
Scenario S2a	Livestock density ≤ 1.4 LSU/ha UAA at regional level.
Scenario S2b	Livestock density ≤ 1.4 LSU/ha UAA at grid level.

Source: EC, 2022

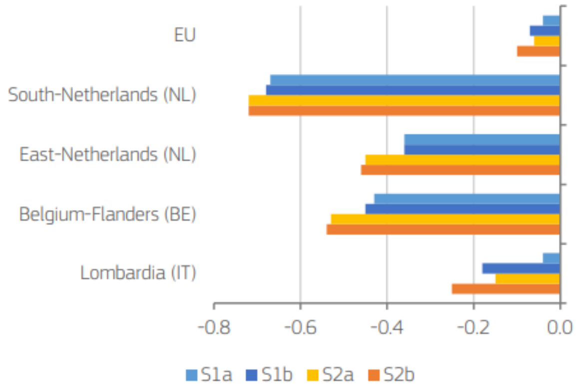
Since the scenarios are characterised by lower livestock densities in the EU, the EU Nitrogen surplus, ammonia emissions and nitrates losses to water are lowered as well. The EU average Nitrogen surplus would fall by 3 - 8 % under S1 and by 6 -12 % under S2. At regional level, the Nitrogen surplus would decrease in some regions but increase in other regions, mainly under scenarios S1a and S2a. Importantly, in hotspot regions, the reduction can reach over 100 kg N/ha or over 50 %.

On average across the EU, the scenarios are expected to lead to a reduction in nitrates leaching and runoff by 4-7 % under S1 and 6-10 % under S2 (see Figure 4.4, Graph 7.5 from the report). However, in hotspot regions, for example in regions in the Netherlands and Belgium, they can achieve reductions over 50%. As the analysis did not take into account manure trade (exports) from these regions, the leaching and runoff reduction effect could be overestimated in these hotspot regions.

In conclusion, the benefits of reducing livestock density below 2 LSU/ha has a significant impact on reducing nitrates leaching and runoff, specifically for hotspot regions with highest livestock densities.

The overproduction of manure in current livestock hotspots is a key driver of nitrate (and phosphorus) pollution, and the 170 kgN/ha limit is a measure to cap livestock density or manure production at the farm or regional level, thereby reducing nitrate pollution. The main problem is the nitrogen import via imported animal feeds, which is necessary because in some hotspot regions the feed requirements for livestock production exceed the capacity for plant production in the region (which more or less corresponds with the capacity to absorb nitrogen excreted). This leads to a permanent oversupply of nitrogen, which cannot be permanently buffered by soils. So, the 170 kg N/ha acts in a similar way as a livestock density limit, helping to avoid an unbalanced nitrogen cycle. However, plant yields vary between regions and so a unique level can be challenged. Moreover, there is the possibility of manure trade. A balanced livestock production between EU regions would avoid wasted and polluting manure disposal in livestock regions and also emissions from mineral fertilizer application in specialized crop regions.

Figure 4.4. Nitrates leaching and runoff reduction in the EU and in selected 'hotspots' (copied from EC, 2022).



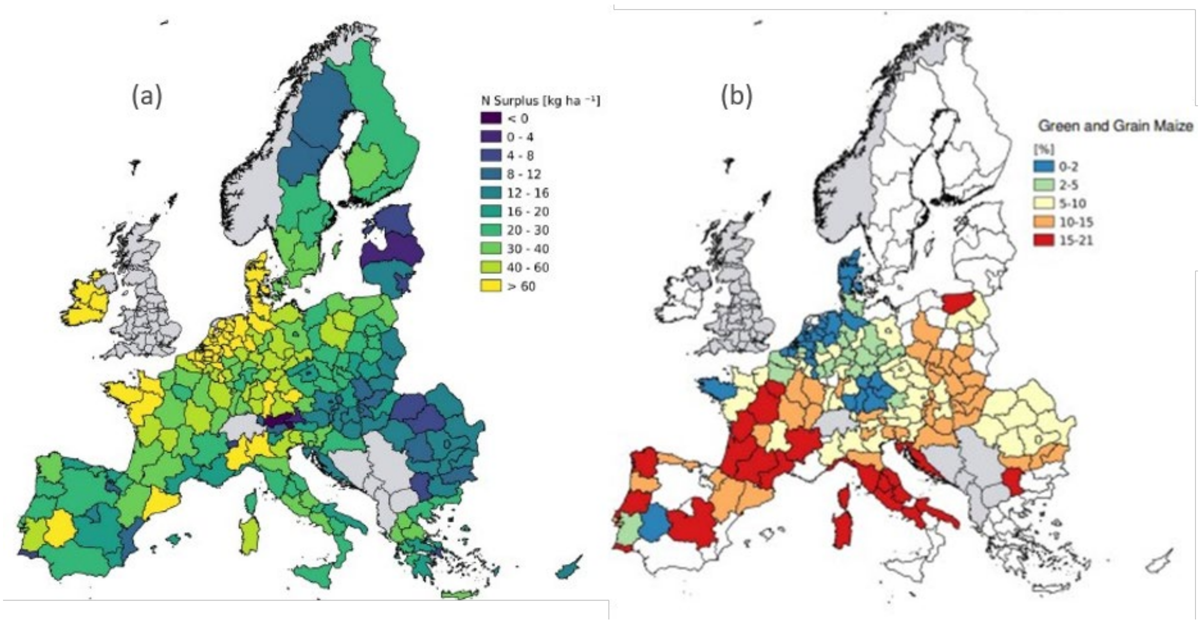
Sources: Scenario simulation based on CAPRI model, EC (2022)

4.2.2 Reducing mineral fertilizers – Scenario analysis with DayCent model

In a recent analysis (Pacifico et al., 2024), we used the DayCent model to evaluate the impact of an abrupt drop in mineral fertilizer applications on crop yields. DayCent is a process-based biogeochemical ecosystem model that has a detailed representation of the daily dynamics of Carbon (C), Nitrogen (N), and Phosphorus (P) partitioning between soil, plants, and the atmosphere (see e.g. Hartman et al., 2020). Three scenarios were simulated whereby mineral N fertilization across the EU is abruptly reduced by respectively 5, 15 and 25 %, and simulated yields were compared to baseline yields (2019–2022). Several factors including current surplus use of N fertilisers and share of organic N fertilisers lead to substantial variation among regions and crop types, for details see Pacifico et al. (2024).

An insight relevant for this assessment relates to the absence of a significant maize yield reduction in several countries even when mineral N was reduced by 25% (Figure 4.5). Using DayCent, even with a 25% drop in mineral nitrogen (N) fertilization, maize yield is not significantly reduced in some highly fertilized EU countries and regions, e.g. Netherlands, Belgium, and Denmark. The reason is the high N surplus, as well as the large share of organic manure fertilization, which was kept at the N directive ceiling of 170 kg N/ha/year. In reality, manure fertilization will have been higher in several regions (also see CAPRI data above). This is surely the case in the various Member States with derogations to the requirements under the N directive, which was not accounted for in these simulations.

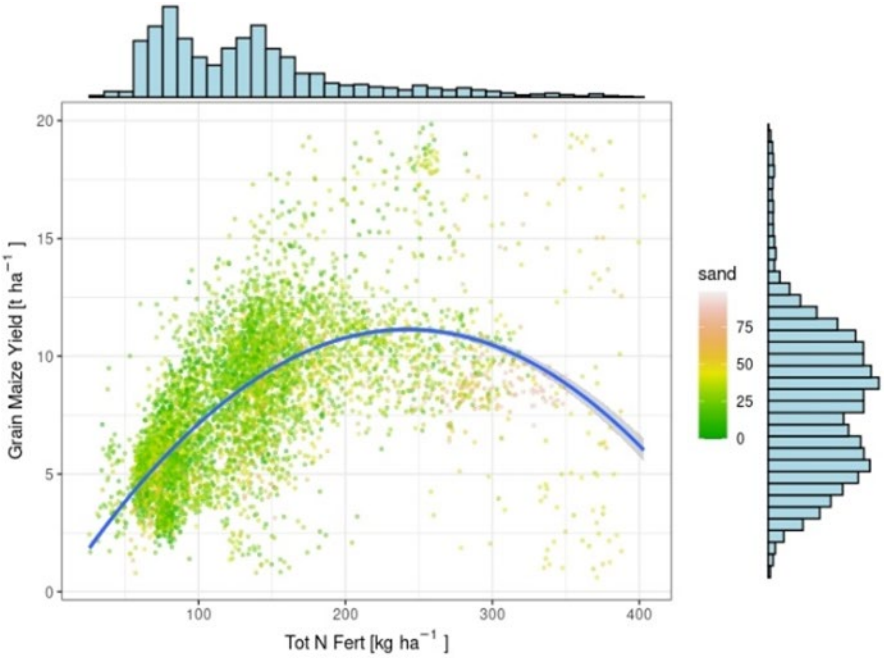
Figure 4.5. (a) Annual (2019-2022 average) N surplus at NUTS2 level in DayCent for the EU; (b) Reduction of the sum of green and grain maize following a 25% reduction scenario in mineral nitrogen fertilization at NUTS2 level simulated with DayCent. The map shows the percentage reduction in yield between the baseline simulation and the scenario. Non-EU countries have grey background.



Source: Pacifico et al., (2024)

This confirms the soil nitrogen surplus as calculated with CAPRI above and illustrates that in these regions nitrogen is applied in excess to crop requirements without any further yield gain. Indeed, typical N response curves, widely found in scientific literature (e.g. van Grinsven et al., 2022), and as derived from the simulations, show how yield increases with fertilization only up to maximum level beyond which increasing fertilization does not increase yield anymore (Figure 4.6).

Figure 4.6. Crop yield response curve to total N fertilization for grain maize simulated with DayCent, including 2nd degree polynomial fitting curves, histograms and % sand content.



Source: Pacifico et al., (2024)

Therefore, maintaining the 170 kg N/ha/year limit for manure fertilization is appropriate with regard to guaranteeing crop yield levels and securing food production. The analysis indicates that there is a large regional crop dependent variation in current manure application rates. Importantly, high manure application rates often coincide with areas with a considerable soil nitrogen surplus (see Figure 4.3 and Figure 4.5 derived from two independent methods).

5 Opportunities and future challenges

This section examines the potential for reducing nutrient losses through best farm practices and nutrient recycling, in the context of the Nitrates Directive. Furthermore, we discuss the challenges of nitrogen reductions needed to align with planetary boundaries and protect the environment from excess nutrients.

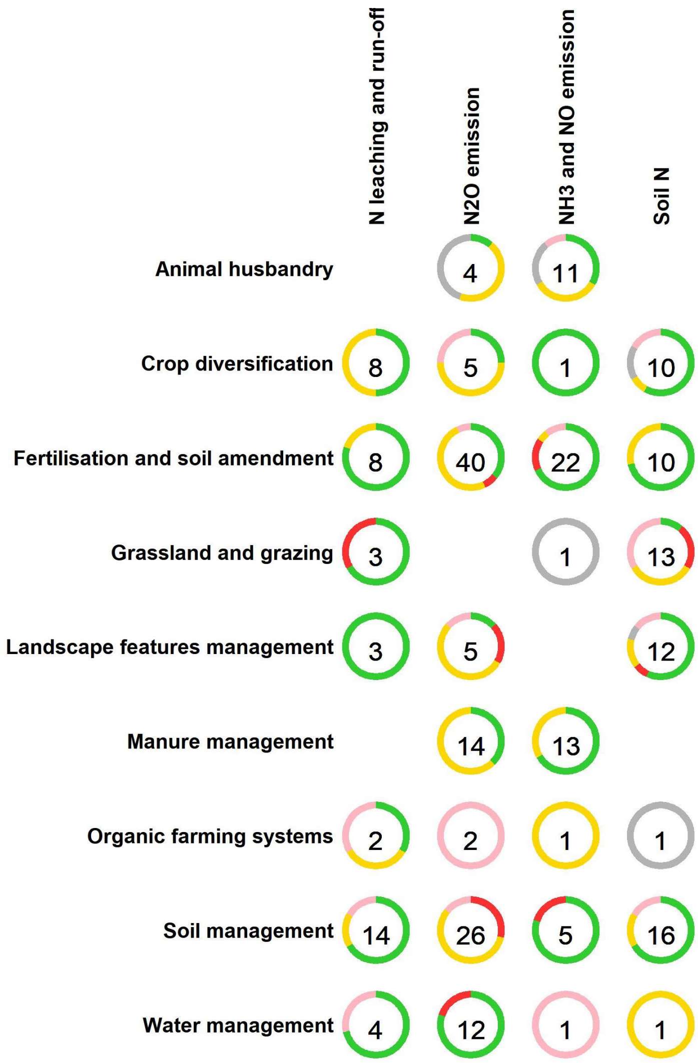
5.1 Farm practices and environmental impacts

Scientific evidence based on a large number of studies show that specific farming practices are able to reduce nitrogen leaching and run-off to waters. The implementation of such practices especially in Nitrates Vulnerable Zones can be beneficial to lower water pollution from agricultural sources.

Here we summarize evidence from a systematic review of meta-analyses to assess the effect of farming practices (FP) on environment, climate and production (Schievano et al, 2024) with a focus on farming practices with an impact on nitrogen management. We analysed the data included in the JRC-Farming-Practices data collection (https://data.jrc.ec.europa.eu/collection/id-00399), evaluating through systematic screening 570 meta-analyses published since 2000 and categorized according to farming practices. Results from this evidence library have been already used to assess the environmental ambition of the proposed CAP SPs in 2021 and have been used by the 'European Evaluation Helpdesk' to estimate the climate change mitigation of the CAP Strategic Plans.

In relation to nitrogen losses, the dataset includes the effect of farming practices on several impacts such as air pollutant emissions (NH3 and NO emission), nutrient leaching and run-off (N leaching and run-off), GHG emissions (N2O emission), and soil nutrients (soil nitrogen). In addition, several other impacts are covered, that can help to check for eventual synergies and trade-off among impacts. In the Figure 5.1 we show the effect of classes of farming practices on the main nitrogen losses based on results extracted from meta-analyses. More detailed results on specific farming practices are reported in the JRC Farming practices Evidence Library (https://datam.jrc.ec.europa.eu/datam/mashup/JRC_FP_EVIDENCE_LIBRARY).

Figure 5.1. Results for classes of farming practices on the main nitrogen losses from the systematic review of meta-analyses. Donut plots show the distribution of results with a significant positive (green) or negative (red) effects, a non-significant effect (yellow) or non-statistically-tested results (grey). Numbers represent the count of available meta-analyses.



Source: JRC analysis based on data from Schievano et al. (2024)

Farming practices with a positive impact on reducing nitrogen leaching and run-off include the use of cover crops and catch crops, crop residues retention, the establishment of grassy and woody features (as field margins hedgerows, lines of trees, buffer strips along water courses), increased use of slow-release fertilizers, the substitution of fertilisers inputs with leguminous service crops (including trees) used as green manure, and organic farming systems. Implementing such practices would reduce the reliance on N input generally, from either mineral or organic sources. Increasingly efficient and diversified farming systems that optimize nutrient management are also beneficial for the environment (e.g. soil erosion, air-pollutants emission) and climate (e.g. reduce GHG emission and store carbon). Substituting mineral N with locally-produced manure-N using efficient fertilization practices (e.g. using RENURE fertilisers and appropriate soil application techniques) and catch- or leguminous- service crops (including trees) are generally a good options in such systems, provided that mineral-N inputs are indeed reduced.

Considerations regarding water and nutrient management go hand in hand. A science for policy brief based on the JRC-Farming-Practices data collection focussed on the effects of water-management-related farming practices in agriculture, including several impacts (i.e. water use efficiency, water consumption, soil water retention, water quality, and nutrient leaching and run-off). The analysis identified 10 farming practices that are potentially beneficial for reducing water use quantity and 13 for improving water quality, with a total of 15 farming practices that can enhance water management in agriculture (Bosco et al. 2024, Figure 5.2).

Figure 5.2 Farming practices with a positive effect on water quantity and water quality (marked in blue), based on the systematic review of meta-analyses.

Farming practice	Water quantity			Water quality	
	Water use efficiency	Water consumption	Soil water retention	Water quality	Nutrient leaching and run-off
Agroforestry systems			■		
Cover and catch crops	■		■		■
Crop residue management	■	■	■		■
Enhanced-efficiency fertilisers					■
Grassland management			■		■
Landscape features (buffer strips and small wetlands)			■	■	■
Low-ammonia-emission techniques for mineral fertilisers					■
Mulching	■	■	■		■
No tillage and reduced tillage		■	■		
Organic farming systems					■
Organic fertilisation					■
Soil amendment with biochar	■		■		■
Wetland conservation and restoration				■	
Water-saving irrigation practices in flooded lands	■	■			■
Water-saving irrigation practices in non-flooded lands	■	■			■

Source: Bosco et al. (2024)

5.2 Opportunities for nutrient recycling

Manure processing is a common practice in some Member States for different reasons. A study by Foged et al. (2011) indicates that about 8% of the livestock manure production in the EU equal to 108 million ton, and containing 556,000 tons N and 139,000 tons P, was processed at that time. Most processing techniques involve anaerobic digestion and composting, but further processing could be performed to obtain more mineral-like fertilisers (Huygens et al., 2020). Materials such as mineral concentrates and scrubbing salts can be obtained following additional separation and concentration techniques such as reverse osmosis and scrubbing-stripping techniques (Camargo-Valero et al., 2015; Velthof, 2015; Huygens et al., 2020). Such materials may contain up to 10-20% of Total Nitrogen (% of dry matter), up to 25% dry matter, and have a relatively low organic carbon content (0-20%) (Huygens et al., 2020).

Manure processing may be performed for hygienisation, odor or volume reduction, energy production, the improvement of nutrient recycling, or a combination thereof. The processing mostly occurs in EU regions characterised by a nutrient excess, such as certain parts of Germany, Belgium, Italy and the Netherlands (Foged et al., 2011; Ehlert et al., 2012), suggesting that a better manure management is an important reason to undertake manure processing. After all, the processing can enhance the efficiency of transporting and storing manure, allowing it to be distributed and applied over greater distances and extended periods, in turn increasing the overall nutrient efficiency and reducing nutrient losses in regions of a high nutrient density.

The Nitrates Directive allows manure processing and nutrient recycling, but any output material applied in NVZ is still subject to the application limit of 170 kg N/ha, unless a derogation applies. Hence, the Nitrates Directive is not a legal impediment for manure recycling as long as the total application of processed and unprocessed manure does not exceed that application limit. Note that manure that is not applied locally, and further processed in e.g. a digestion or composting plant is also a waste material, and thus subject to waste provisions, unless it meets national End-of-Waste conditions or is contained in an EU Fertilising Product[3].

The recent amendment of the Nitrates Directive[4] allows Member States to authorise the use of certain types of processed livestock manure (qualified as RENURE, REcovered Nitrogen from manURE) up to an extra limit of 80 kg N ha^{-1} yr^{-1} on fields, in addition to the existing manure ceiling of 170 kg N ha^{-1}. Since the manure-derived materials are not fully equivalent to mineral fertilisers, provisions in relation to chemical composition, the presence of pathogens and specific metals in the materials have been included, as well as application conditions to limit gaseous emissions and labelling requirements. Finally, the application of these processed manure N-fertilisers should not lead to an increase of the livestock density or do not put risk the attainment of environmental quality standards, e.g. for nitrates in water bodies.

These safeguards are provided to prevent a (further) intensification of the livestock agriculture in the EU. However, the N application limit in Nitrates Vulnerable Zones still applies for other materials (e.g. those rich in organic matter, including digestates) that do not meet the criteria set out in the Commission's proposal to amend the Nitrates Directive. This is intentionally as the JRC 2020 study (Huygens et al., 2020) indicated that applications above the limit may induce further adverse impacts that may compromise environmental protection, or do not show a consistent quality that can be straightforwardly and cost-efficiently be monitored. Such recycled materials may show N dynamics after land application that are more comparable to unprocessed manure. As indicated in other sections of this report, the application limit of 170 kg N per ha of manure may be required or is even beyond the maximum manure application that agricultural ecosystems can withstand without exceeding the planetary boundaries for nutrient pollution. Finally, it is recalled that waste prevention is the preferred option under the waste legislation. Hence, when possible, and from an overall sustainability perspective, the prevention of generation of excess manure to volumes that can be applied locally and aligned to good agricultural and nutrient management practices are preferred over excess manure recycling, particularly for the long-term perspective. In conclusion, the Directive enables manure processing and nutrient recycling, but provisions are put in place to ensure that the overall objectives of environmental protection from excess nutrients are not compromised.

[3] Compliant with Annex I-IV of Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products.

[4] Commission Directive (EU) 2026/288 of 9 February 2026 amending Council Directive 91/676/EEC as regards the use of certain fertilising materials from livestock manure.

5.3 Future challenges: climate, planetary boundaries, agriculture and water resilience

Reducing nutrient waste and losses to the environment is feasible through best practice farming, balanced livestock and crop systems and nutrient recycling. Nevertheless, a range of challenges must be overcome to attain the nitrogen reductions required for water, soil and air quality objectives and to fulfil the aims of the Nitrates Directive.

- **Staying within the planetary boundary** – De Vries et al. (2021) identified thresholds for N losses based on the spatially explicit ecosystem sensitivity for N losses by runoff to surface water, NH3 emissions and N losses via leaching to groundwater. Using these thresholds, 74 %, 66 % and 18 % of all agricultural land in the EU+UK has too high N inputs for surface water, air and groundwater quality, respectively (de Vries et al., 2021). To keep N losses within the spatially explicit environmental boundaries, the average N input across EU agricultural land should reduce from 145 kg N ha^{-1} $year^{-1}$ (year 2010) to 83 kg N ha^{-1} $year^{-1}$ (De Vries et al., 2021). The spatial variability of critical N inputs is large, but is for most areas below 170 kg N ha^{-1} $year^{-1}$ when considering N runoff to surface water to avoid eutrophication (De Vries et al., 2021).

- **Climate driven shifts in nutrient pathways** – Warmer temperatures, altered precipitation patterns and more frequent extreme events modify N leaching, runoff and gaseous losses, complicating the prediction and management of impacts on rivers, lakes and coastal waters. Adaptation may require new crop varieties, altered sowing dates and adjusted water and fertiliser use strategies.

- **Balanced management of nitrogen and phosphorus** – Both nutrients must be managed jointly to avoid disproportionate effects on aquatic ecosystems; measures that target only N may inadvertently exacerbate phosphorus related eutrophication.

- **Manure recycling without secondary pollution** – Circular use of livestock waste should be coupled with technologies that limit N emissions to air, water and soils, ensuring that processed manure substitutes mineral fertilizer, rather than bypass the Nitrates Directive limit.

- **Robust data collection and monitoring** – Continuous, high resolution data on fertiliser application, soil N stocks and water quality are crucial for detecting system changes and implementing targeted interventions.

- **Economic and social acceptability** – Reductions in nitrogen inputs can affect yields and farm profitability, and require behavioural changes among consumers; policies must therefore incorporate cost effectiveness and stakeholder engagement.

- **Policy coherence across the EU** – The Nitrates Directive must be aligned with the Water Framework Directive, Groundwater Directive and other sectoral policies to avoid contradictory requirements and to harness synergies for nutrient management.

An integrated approach that combines a sound scientific understanding of nutrient fluxes, long term monitoring networks and coherent policy frameworks is essential. Addressing these interlinked challenges will enable the EU to meet its nitrogen reduction targets while respecting planetary boundaries and protecting aquatic ecosystems.

6 Conclusions

The Nitrates Directive has had positive effects in reducing nitrogen input to waters and promoting better management practices to reduce environmental impacts (Section 2.3). The Directive has also contributed to reduce ammonia emissions with benefits for air quality (PM2.5) and nitrogen deposition on terrestrial and aquatic ecosystems.

However, the threshold of 50 mg of nitrate per litre foreseen by the Directive is too high for protecting the ecological quality of water bodies in Nitrates Vulnerable Zones. Focus should be placed on harmonising effort across Member States to ensure both nitrogen (N) and phosphorus (P) are monitored and that thresholds are appropriate to meet EU water policy objectives (Water Framework Directive) (Section 3.1). Similarly, the limit of 170 kg of nitrogen in manure application per hectare appears excessive in regions with elevated soil P concentrations, for ensuring soil health as defined in the Soil Monitoring Law (Section 3.2). The manure application limit in the Nitrates Directive might be required for achieving air quality objectives and specifically ammonia emissions reduction targets set in the National Emission reduction Commitments Directive and FitFor55 policy package (Section 3.3).

An integrated approach to crop nutrient management is needed to reduce nitrogen pressure on environment and waterways. Unprecedented detail in livestock densities from the 2020 agriculture census illustrate that livestock densities remain high (>3 LSU/ha) in countries such as the Netherlands, Belgium, Denmark, Bretagne (France), the Po valley (Italy), and parts of Germany and Spain. In these areas the production of manure is logically high, and often likely more than what surrounding agricultural areas could utilize without surpassing the limit to manure application of 170 kg N/ha per year (Section 4.1). Independent spatially disaggregated N budget and manure data from CAPRI model confirms these hotspots with manure application in excess of the 170 kg N/ha limit. Importantly, areas with high livestock densities often coincide with areas that display a considerable soil nitrogen surplus (Section 4.2). The overproduction of manure in current livestock hotspots is a key driver of nitrate (and phosphorus) pollution, and the 170 kgN/ha limit is a measure to cap livestock density (with the risk of being by-passed if using processed manure). As advocated also by other studies, a balance and integration between crop and livestock production would be beneficial for reducing environmental impacts (within EU and outside EU), facilitate manure recycling and reduce EU dependency on mineral fertilisers from non-EU countries (Section 4.2).

There is room for reducing nitrogen input without large losses. Environmental scenario analysis carried out with CAPRI model for the 2022 EU Medium-term Agricultural Outlook concluded that the benefits of reducing livestock density below 2 LSU/ha (corresponding to the 170 kg N manure/ha limit) significantly reduces nitrates leaching and runoff especially for hotspot regions characterised by high livestock densities. Simulations with the DayCent biogeochemical modelling framework illustrate that a reduction of mineral N by 25% did not results in significantly lower maize yields in areas with excess N applications (from both mineral and organic sources) (Section 4.3).

There are opportunities for reducing nitrogen losses and increase recycling. Adoption of farming practices (permanent soil cover, cover/catch crops, buffer strips, woody features, hedgerows, agroforestry) that can reduce nitrate leaching and soil/nutrients run-off should be encouraged (Section 5.1). The recent RENURE Act, amending the Nitrates Drective, enables manure processing and nutrient recycling. However, in line with the circular economy objectives, provisions are established to ensure that the overarching goals of environmental protection from excess nutrients are not compromised (Section 5.2).

Key challenges to achieve the Nitrates Directive goals include keeping nitrogen inputs within planetary-boundary limits, coping with climate-driven changes in nutrient losses, and ensuring balanced management of nitrogen and phosphorus to avoid aggravating eutrophication. Circular manure use must be coupled with technologies that prevent secondary pollution, while monitoring of fertiliser application, soil and water quality is essential for targeted action. Economic viability and social acceptance of lower nitrogen use, together with coherent alignment of the Nitrates Directive with other EU policies, are also critical for a sustainable nutrient management approach (Section 5.3).

An integrated approach to nutrient management is needed to balance agricultural production with environmental protection. Adopting sustainable farming practices and manure recycling can reduce nitrogen losses and support EU green transition and climate neutrality goals, ensuring water resilience and sustainable food production.

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List of abbreviations and definitions

Abbreviations	Definitions
ND	Nitrates Directive
NVZ	Nitrates Vulnerable Zone
TN	Total Nitrogen
N	Nitrogen
NO3	Nitrate
NH3	Ammonia
P	Phosphorus
TP	Total Phosphorus
PO4	Phosphate
WFD	Water Framework Directive
MSFD	Marine Strategy Framework Directive
UWWTD	Urban Waste Water Treatment Directive

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