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EVALUATION

Evaluation of the Nitrates Directive
(Council Directive 91/676/EEC)

{SWD(2026) 235 final}

ANNEX XI. ENVIRONMENTAL EFFECTS OF THE MEASURES IN ANNEX II AND ANNEX III OF THE NITRATES DIRECTIVE

This annex provides background on the effects of the measures set out in the annexes of the Nitrates Directive and which the Member States should include in their action programmes for nitrates vulnerable zones.

The annex is based on analysis performed by the Soil Service of Belgium during 2024 under a contract[1] with the European Commission for the provision of scientific assistance for the implementation of the Nitrates Directive.

In addition to the strict focus on measures set out in the annexes of the directive, this annex also covers some additional types of measures of relevance for the directive's objectives (buffer strips, nature-based solutions). Member States are free to add such additional elements.

For the analysis, the various types of measures are grouped in 5 clusters: page

1. CLUSTER I – PROHIBITION PERIODS, WINTERLY CONDITIONS, MANURE STORAGE

The risk of nitrate and phosphorus pollution of groundwater and surface waters from agriculture depends on the presence of

- sources of N and P,
- vulnerability of groundwater and surface waters, and
- transfer pathways of N and P from sources to groundwater and surface waters.

Land fertilized with animal manures, fertilizers, composts, and crop residues are one of the possible N and P sources. Regarding the state of the land, this potential source implies a different risk regarding pollution.

Possible transfer pathways are

- surface/subsurface runoff of N and P to surface waters,
- downward leaching of nitrate-N, which directly leads to groundwater pollution by nitrates,
- the emission of gaseous N compounds into the atmosphere and the subsequent deposition of atmospheric N compounds to surface waters and lands, and
- direct application or discharge of fertilizer and manures into surface waters.

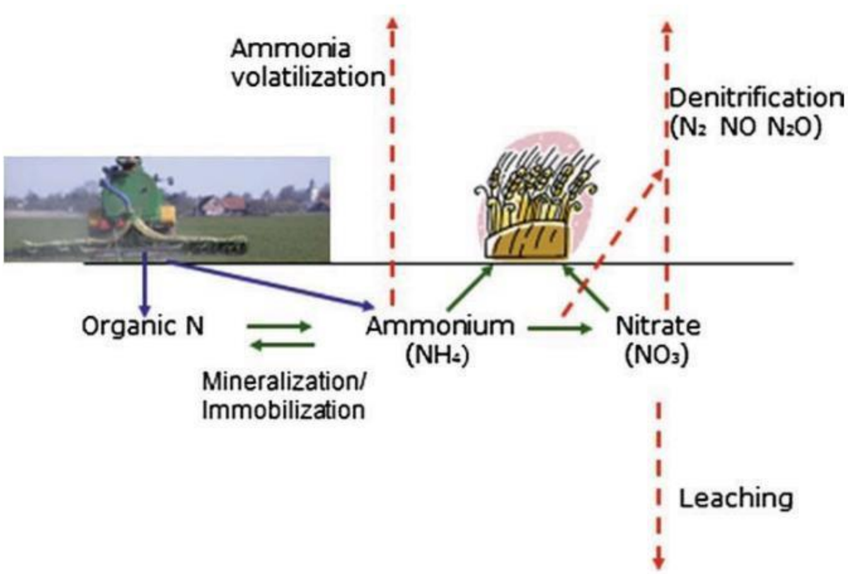
[1] Service contract No. 09.0201/2022/881726/SER/ENV.D.1

1.1. Periods when the land application of certain types of fertilizer is prohibited.

1.1.1. Background

The application of N fertilizer, especially manure, in autumn and winter is particularly linked to the risk of nitrate (NO_3^-)-leaching (Di & Cameron, 2002; Cameron et al., 2013). Nitrate leaching occurs when there is an accumulation of NO_3^- in the soil profile that coincides with or is followed by a period of high drainage. NO_3^--accumulation can occur through (excessive) nitrogen application rates or N applications at the wrong time of the year, not synchronized to plant demand. A decrease in plant growth rate leads to an increase in the organic N-compounds recirculating in the phloem and repressing N absorption by the roots. So the actual N absorption rate of a plant is co-regulated by both soil N supply and its own growth rate potential (Gastal et al., 2015). Leaching occurs when inputs of water exceed the level that can be retained according the water holding capacity of the soil and the level of water use by evapotranspiration. Therefore, autumn applications are likely to increase the risk of nitrate leaching losses, because crop N uptake during the autumn/winter period is generally low and there is sufficient over-winter rainfall to transport manure derived nitrate beyond crop rooting depth. Closed periods are specific times of the year when the application of inorganic and/or organic fertilisers is prohibited, with the objective of reducing the loss of NO_3^- via leaching and run-off. Nitrogen in manures is present in both mineral and organic forms (Figure 1). Mineral-N in manures consists mainly of ammonium (NH_4^+) ions and NH_3 in solution. After aerobic decomposition, manures can also contain mineral-N as NO_3^-. Nitrogen in chemical fertilizers is either in the ammonium or nitrate form, while urea is a chemical fertilizer with nitrogen in an organic form. Nitrate is the nitrogen form that is most susceptible for leaching, due to poor retention by the soil. In soil, ammonium is mineralised to NO_3^-. The organic forms of nitrogen are also aerobically decomposed, first to ammonium, and subsequently to nitrate, but the mineralisation of organic N is a slower process compared to the nitrification of the mineral ammonium. Therefore, the content of mineral nitrogen in the manures determines the immediate risk for leaching. For manures, differences in mineral nitrogen content will therefore result in different leaching risks. For chemical fertilizers, high nitrate containing fertilizers pose more leaching risk compared to ammonium-based fertilizers or urea (Hina, 2024).

Figure 1 The most important N transformations and losses after application of animal manure to soil. (Webb et al., 2013)

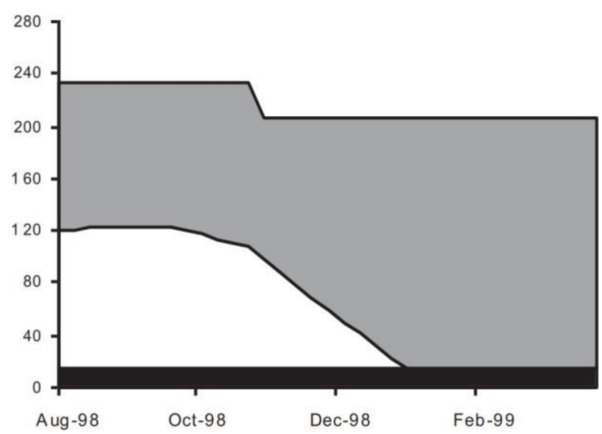


It can be concluded that the leaching risk of applied manures or chemical fertilizers depends on the content of mineral nitrogen or nitrate, the time period between application and sowing/planting, the weather conditions and hydrological status of the soil between application and sowing/planting, and the success and type of crop.

1.1.2. *Effectiveness and efficiency in terms of preventing and reducing nutrient losses.*

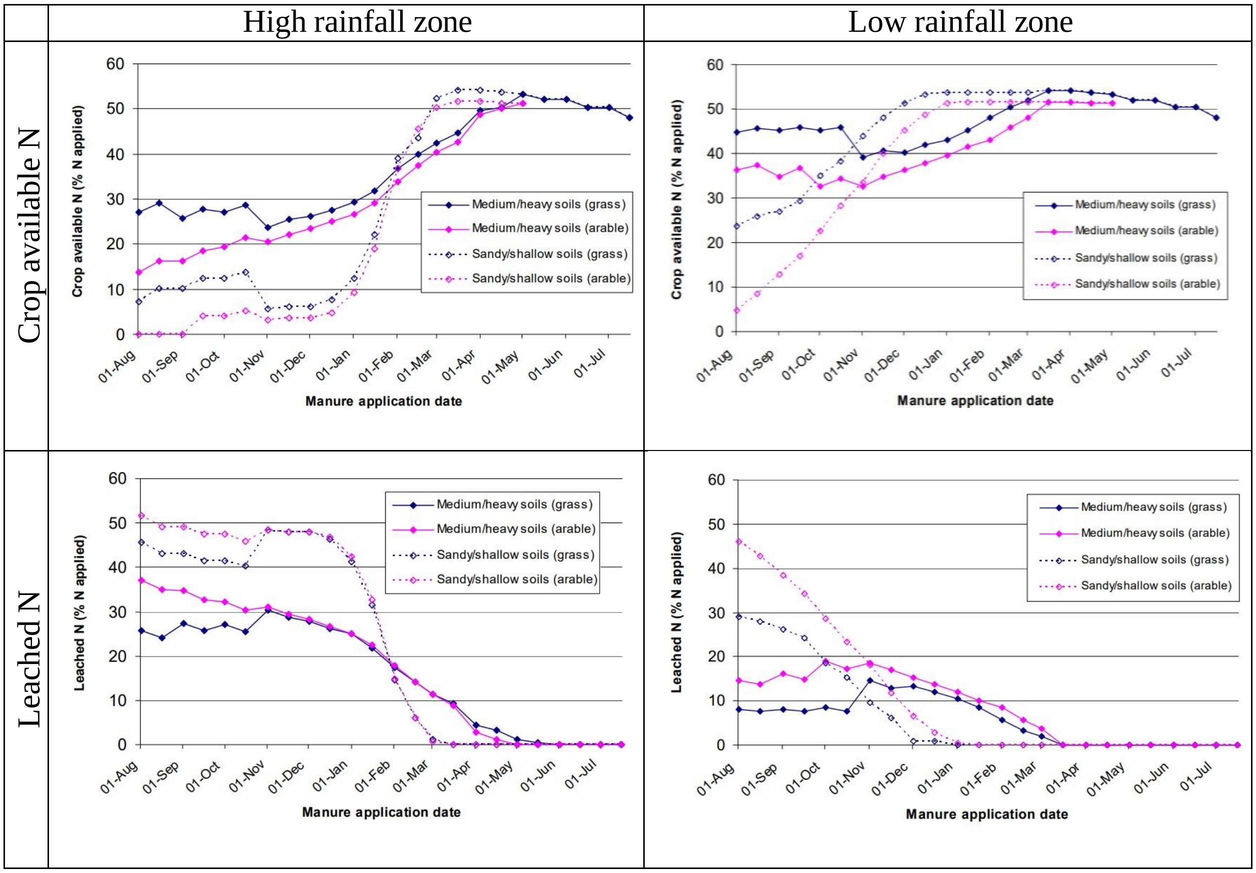
The 'closed spreading periods' or 'prohibition periods' are designed to minimise nitrate leaching (and other nutrient) losses. Times of the year when heavy rainfall is expected, temperatures are low, crop growth is limited and/or soils are left bare (limiting evapotranspiration), are particularly vulnerable to both leaching losses and the generation of overland flow (run-off losses) and coincide often with winter months. To a certain extent the measure aims for a better coordination of demand and supply. Synchronizing N supply with crop demand is essential in order to ensure adequate quantity of uptake and utilization and optimum yield (Fageria and Baligar, 2005). This synchronization will improve the N use efficiency (NUE) which is desirable in order to maintain or ameliorate environmental quality. Webb et al. (2013) mention the timing of manure applications to reduce the risk of NO_3^- leaching as a well-documented approach to increase manure-N efficiency. They refer to Webb et al. (2001) who demonstrated the impact of differences in the time of application for UK conditions (Figure 2). They also refer to Webb and Archer (1993) and their references confirming that autumn applications of manure cause increased leaching in areas with moderate or high precipitation during autumn and winter.

Figure 2 The proportion of N lost by NO_3 − leaching from application of broiler manure to a CL soil in Eastern England at times of application between early August and late March in a year of average hydrologically effective rainfall for the location. The white area of the graph represents the manure-N lost by leaching. The black area represents NH_3–N emissions. The gray area represents the manure-N remaining in the soil. The units of the y-axis are kg N ha^{-1} (Webb et al., 2001).



Nicholson et al. (2011) reviewed and synthesized some UK research projects and confirmed that autumn slurry applications present the greatest risk of nitrate-N losses to drainage waters. They mentioned the MANNER-NPK-model which can predict manure N efficiencies of high-readily available N manures applied at 2-week intervals throughout the year to grassland and arable crops in the different soil type and agro- climatic zones. These results illustrated how changing the timing of manure application can affect the amount of manure N taken up by the crop (Figure 3-top). By consequence, the outputs from the MANNER- NPK-model were also used to predict nitrate leaching losses from the different high-readily available N manure applications. The results illustrated how changing the timing of a slurry application can affect the amount of N lost through nitrate leaching (Figure 3-Bottom), and by consequence the effectiveness of a period when fertilisation is prohibited.

Figure 3 MANNER-NPK predicted crop available N (% total N applied) (top) and nitrate-N leaching (% total N applied) losses (bottom) at different application timings (pig slurry broadcast applied and not soil incorporated). High rainfall zone (median rainfall = 1200 mm/annum).- left; Low rainfall zone (median rainfall = 650 mm/annum).- right (Nicholson et al., 2011).



The authors evaluated the impact of extending the closed spreading periods (Table 1) for high-readily available N manures. Crop N uptake increased up to 5 % of total N applied for pig slurry and N leaching losses reduced with up to 3 % of total N applied for cattle and pig slurry (Table 2). For poultry manure the enhanced efficiency coincided with a 4 % of total N applied reduction of ammonia losses. Due to the closed spreading periods, the authors found a decrease in N leaching of 37.5% for cattle slurry and 30 % for pig slurry, compared to the business-as-usual manure spreading.

Table 1 'Closed periods' for spreading manures with readily available N contents greater than 30% of total N, applied for two scenarios by cropping and soil type (Nicholson et al 2011)

Scenario	Grassland	Tillage
BASELINE (2007)		
Sandy/shallow soils	15 Sept – 1 Nov	1 Aug – 1 Nov
All other soils	None	None
CURRENT NVZ-AP		
Sandy/shallow soils	1 Sept – 31 Dec	1 Aug – 31 Dec
All other soils	15 Oct – 15 Jan	1 Oct – 15 Jan

Table 2 Impact of 'closed spreading periods' and increased slurry storage on crop N uptake, nitrate-N leaching losses, ammonia- N emissions and total P losses in NVZ areas. (Nicholson et al., 2011)

	Crop N uptake		Nitrate-N leaching losses		Ammonia-N losses to air		Total P losses to water	
	Total (kt)	% total N applied	Total (kt)	% total N applied	Total (kt)	% total N applied	Total (tonnes)	% total P applied
Baseline[1]								
Cattle slurry	11.6	27	3.3	8	5.5	13	162	1.8
Pig slurry	4.4	44	1.0	10	1.6	16	40	1.7
Poultry manure	12.4	30	3.4	8	7.9	19	26	0.2
Current NVZ-AP[2]								
Cattle slurry	12.7	30	2.2	5	5.5	13	152	1.7
Pig slurry	4.8	49	0.7	7	1.4	14	38	1.6
Poultry manure	13.9	34	3.4	8	6.2	15	26	0.2

[1]Assumes 20% of cattle slurry, 75% of pig slurry and 50% of poultry manure was incorporated by plough within 24 hours.

[2]Assumes 30% of pig slurry and 4% of cattle slurry was applied by trailing hose to arable land or shallow injected to grassland. Of the remainder, 30% of cattle slurry, 80% of pig slurry and 80% of poultry manure applied to arable land was incorporated by plough within 24 hours.

Shore et al. (2016) concluded after a 4-year study in Ireland that the disproportionately high nutrient losses identified during the closed period supported the notion of restricting nutrient applications during this period. They also referred to the soil moisture deficit as a risk determining factor. Soils with a soil moisture deficit of less than 0 mm are more likely to transport available nutrients to water bodies in surface runoff and drainflow rapidly and, therefore, constitute high risk. Shore et al. (2016) noted that in almost all of the study sites the soil moisture deficit was about zero nearly all of the time in the closed period. The results demonstrate that imposing periods when the land application of certain types of fertilizer is prohibited, is effective in reducing N losses.

Figure 4 Example of scoring the coverage of the leaching risk by a closed period in the Atlantic Central environmental zone as described by Tzilivakis et al. (2021).

Environmental zone: Atlantic Central

Closed period of 1 November to 15 January:

Risk during the closed period	1(Nov) + 1 (Dec) + 0.5 (1/2 Half January) = 2.5
Maximum leaching risk in Atlantic Central:	1+0.5+1+1+1+1= 5.5
Coverage score for leaching risk:	(2.5 /5.5)*100= **45 %**

Leaching risk periods for environmental zones based on Alterra et al. (2011)

Zone	Crop	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Alpine North	All crops	1	1	0	0	0	0	0.5	1	1	1	1	1
Alpine South	All crops	1	1	0	0	0	0	0	0.5	1	1	1	1
Atlantic Central	All crops	1	0	0	0	0	0	0	0.5	1	1	1	1
Atlantic North	All crops	1	1	0	0	0	0	0	0.5	1	1	1	1
Boreal	All crops	1	1	0	0	0	0	0.5	1	1	1	1	1
Continental	Arable	1	0	0	0	0	0	0	1	1	1	1	1
Continental	Grass	1	0	0	0	0	0	0	0.5	1	1	1	1
Lusitanian	All crops	1	0	0	0	0	0	0	0.5	1	1	1	1
Mediterranean Mountain	All crops	1	0	0	0	0	0	0	0.5	1	1	1	1
Mediterranean North	Arable - early summer	1	0	0	0	0	0	0	0	1	1	1	1
Mediterranean North	Arable - late summer	0	0	0	0	1	1	1	0	0	0	0	0
Mediterranean North	Non-irrigated grass/perm crops	1	0	0	0	0	0	0	0	0	0	1	1
Mediterranean South	Arable - early summer	1	0	0	0	0	0	0	0	0.5	1	1	1
Mediterranean South	Arable - late summer	0	0	0	0	1	1	1	0	0	0	0	0
Mediterranean South	Non-irrigated grass/perm crops	1	0	0	0	0	0	0	0	0.5	0.5	1	1
Nemoral	All crops	1	1	0	0	0	0	0	0.5	1	1	1	1
Pannonic-Pontic	Arable	1	0	0	0	0	0	0	1	1	1	1	1
Pannonic-Pontic	Grass	1	0	0	0	0	0	0	0.5	1	1	1	1

Key: 0 = low; 0.5 = medium; 1 = high

Tzilivakis et al. (2021) evaluated the effectiveness of the fertiliser closed periods under the Nitrates Directive. He regarded the overlap of the closed period and the risk of leaching and run-off, the latter based on Alterra et al. (2011, part D). The effectiveness depends on the length and timing of the closed period. Each closed period within each NAP was scored for coverage of the risk periods. The coverage of the risk is determined by summing up the risk scores the closed period covers (to the nearest half month) and expressing this as a percentage of the maximum. An example of a closed period in the Atlantic Central environmental zone is given in Figure 4. Their snapshot (measures in place in 2019-20) showed that in 7 Member States more than 50 % of agricultural NVZ area had more than 61 % coverage of the leaching risk periods and in 11 Member States more than 50 % of agricultural NVZ area had more than 61 % coverage of run-off risk periods.

Leaching risks are highest when rainfall is expected in combination with a period where nitrogen uptake by plants is restricted. This is most likely in cold and rainy periods, i.e., late autumn and winter. However, in dry and hot periods, plant growth is also restricted. In that respect a closed period for fertilizer application could possibly prevent leaching risks in such conditions during a precipitation event. Several Spanish regional action plans refer to respecting critical periods of fertilizer application (region of Andalucia, Extremadura) related to the vegetative state or maximum period between application and sowing/planting. In Spain, closed periods during summer occur in different regional action plans, mostly for cereals, likely due to less growth during the dry period. Greece nor Cyprus mention prohibition periods in dryer periods.

1.1.3. Environmental co-benefits and drawbacks

Closed periods have the potential to have other environmental benefits. Reducing the risk of run-off of organic material (containing pathogens and substances with a high Biochemical Oxygen Demand (BOD)) is a potential secondary benefit of closed periods (Tzilivakis et al., 2021). Prohibiting the application of organic fertilisers at times when overland flow is most likely reduces the chance of organic materials (containing and pathogenic organisms and substances with a high BOD) being transported into surface waters (Pandey et al., 2014).

Reducing the risk of N_2O emissions is also a potential secondary benefit of closed periods (Tzilivakis et al., 2021). The application of N containing fertilisers at times when soil conditions may be prone to waterlogging may result in increased denitrification and emissions of N_2O. These conditions or moments can be covered by the closed periods and as such the closed periods can reduce the risk of N_2O emissions.

A co-benefit could exist, however to a lesser extent, for NH3 emissions. The co-benefits or the combined risk reduction (whether or not N2O or NH3 emissions) depends on the positioning of the closed period, the coincidence of the respective risk periods. These will be regionally different.

A potential drawback of too strict deadlines for manure application is reduced flexibility for farmers to adapt to the current soil conditions. The effect of this will depend on climate zones and soil types. Dryer areas in Europe are less susceptible to soil compaction since wet conditions occur less frequently.

1.1.4. Minimum/threshold or optimum level of implementation

The efficiency of the measure depends on climate conditions, soil type or soil characteristics and land use (cultivated crop and crop uptake). Figure 3 (above) demonstrated that the effect was most pronounced for the shallow soils under high rainfall conditions. The length of the prohibition period is also an important parameter influencing efficiency. Nicholson et al. (2011), however, demonstrated the delicate balance of the length of the prohibition period. Extending the closed periods mentioned in Table 1 by 1 month in spring was predicted to further reduce nitrate losses. However, extending the closed period by 2 months was predicted to increase nitrate losses, compared with the 1-month extension. Moreover, the authors mentioned that a further extension of the 'closed periods' was likely to increase the potential for methane and N_2O emissions during manure storage, because of the requirement to store slurry and poultry manure for longer.

The cost of the closed periods relates to the cost associated with extra storage capacity. The economic benefits can be quantified in terms of reduced chemical fertiliser use, resulting from the improved manure nutrient efficiency, and reductions in ecosystem damage costs resulting from abated ammonia-N and nitrous oxide emissions, nitrate leaching and P losses. The report of Nicholson et al. (2011) mentioned a cost-benefit ratio of 1,4:1 for the closed periods mentioned in Table 1 for the 'current NVZ-AP'. The ratio increased by extending the storage periods, indicating that the additional costs of extending the current NVZ-AP storage periods were not reflected in proportional reductions in manufactured fertiliser use and ecosystem damage costs.

According to Alterra et al. (2011, part D) relevant factors for determining the prohibition period are the length of the growing season, the rainfall surplus outside the growing season, the temperature outside the growing season, the soil type and drainage, and the slope. Differentiation of the closed period for soil type, climate (rainfall) and land use is appropriate.

The principle of a prohibition period is climate robust. The periods as such can be adapted (shifted, extended or shortened) to changing climatic conditions. Member States could revise the prohibition period at regular intervals, e.g. every 5 years considering the climate conditions of the past 5 or 10 years. Climate change, however, increases the risk of more extreme events (e.g. droughts, rains, …) and more unpredictable events. These cannot be countered by a fixed prohibition period.

The effectiveness of fixed prohibition periods will likely be lower due to unexpected/unpredictable weather events. Prohibition periods could be supplemented with warning services or short-term prohibitions based on (longer term) weather forecasts. Such weather forecasts could take into account factors such as:

- Incidence of frost and snow
- Incidence of longer drought periods
- Risk of volatilisation
- Risk of surface runoff due to intense rainfall events
- Incidence of heavy rainfall, risk of flooding
- Incidence of wind in combination with high temperatures and/or drought

In case of short-term prohibitions, the feasibility for the farmers must not be forgotten, this demands a high degree of flexibility. Prohibition periods expressed in function of the vegetative state of crops or in terms of a maximum period between application and sowing/planting can from that point of view be more "climate robust" than a period defined in terms of fixed dates.

1.2. Restrictions for application on water saturated, flooded, frozen or snow-covered ground.

1.2.1. Background

The restriction for application on water saturated, flooded, frozen or snow-covered ground targets the fertilized land as a potential source of risk because of certain defined states. The relevant risks include surface runoff, leaching and emission of gaseous N.

The term surface runoff covers overland flow, run-off and shallow lateral flow. Waterlogging refers to the saturation of soil with water either temporarily or permanently. Soil is regarded as waterlogged when it is saturated with water to an extent that its air phase is restricted and anaerobic conditions prevail.

1.2.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses.

There is not much scientific literature about nutrient losses through surface runoff and leaching of fertilizer and manure nutrients from water saturated, flooded, frozen or snow-covered soils in Europe. An important reason could be that it is evident that such practices will lead to high nutrient losses. Wherever the evapotranspiration of crops and the water holding capacity of soils are exceeded by natural or anthropogenic additions of water, there is a risk that nutrients will be lost. An imbalance in demand and supply of water on soils tends to occur:

- on soils that are regularly waterlogged due to an impermeable subsoil or a high water table
- when considerable amounts of snow/ice start to melt
- on soils that are impermeable due to their frozen state

Kerebel et al. (2013) stated that the risk of pollution is partly related to the soil water status during and after spreading. **Waterlogged or saturated soils** can trigger leaching, runoff or denitrification with associated gaseous losses. Wet soils have greater risk of runoff. An increase in soil moisture diminishes the hydraulic gradient and decreases infiltration into the soil.

The "infiltration capacity" or the potential infiltration rate of a soil is the maximal rate at which the soil surface can absorb water. Therefore, when the water supply exceeds the infiltration capacity, only part of it infiltrates and the remaining part remains on the soil surface or runs off according to local topography. Soil infiltration rate is a key variable in soil particle and sediment delivery processes, and it has an impact on fertilization efficiency, environmental quality, and water conservation (Assouline, 2013).

Motarjemi et al. (2023) studied the impact of different drainage conditions in Denmark. They found that N losses of poorly drained soils were 14 % higher compared to those of well-drained soils. Greater losses of nitrogen through leaching from drainage and other pathways were measured at the poorly drained soils (109 kg N ha^{-}1 ya^{-1}) compared to the well-drained soils (95 kg N ha^{-1} ya^{-1}). At the same time, denitrification in topsoil was also higher under poor drainage conditions (see also section 1.2.3 below).

Soils with **frozen** soil layers have low water infiltration rate and hydrological conductivity. These elements, even when the frozen soil layer is below the soil surface, can trigger superficial run-off and enhance the risk of overland flow. Srinivasan et al. (2006) referred to Hensler et al. (1970), who applied fresh dairy manure on frozen soils for two consecutive winters and found that the runoff losses were variable. During the first year of observation, more N and P losses were noted in runoff than occurred during the second year. They observed that the dry winter conditions during the second year resulted in less nutrient losses. Williams et al. (2012) mentioned the infiltration of water into the soil varied according to soil temperature during precipitation. They observed a 56 % increase in runoff volume for frozen soils compared to non-frozen soils and referred at that point to Zuzel and Pikul (1987) and Gray et al. (2001) who also mentioned an increase in runoff volume on frozen soils.

Young et al. (2022) measured nutrient losses after dairy manure was applied on top of an established **snowpack**. The results indicated that applying manure on top of snow resulted in high nutrient losses when runoff occurred, regardless of manure solids content. They concluded that liquid manure application on top of an existing snowpack resulted in high N and P losses in snowmelt runoff and was potentially a high-risk surface water quality condition depending on runoff hydrology and stream proximity.

Williams et al. (2011) examined manure application under winter conditions using laboratory methods (Table 3). They studied the influence of manure position within the snowpack on surface and subsurface losses of N and P. For one of the treatments, manure was applied on frozen ground, so this study can give evidence for fertilisation on frozen ground and snow-covered ground and the possible loss. Manure application on frozen ground or on top of a snow cover resulted in higher N and P losses. (All soils were at field capacity at the time of soil freezing, influencing the volume of surface runoff.)

Table 3 Summary of nitrogen and phosphorus losses in surface runoff and subsurface leachate (Williams et al., 2011).

	Runoff and Leachate Losses ($\mu g\ cm^{-2}$)					
	NH_4-N		NO_3-N		Total N	
Treatment	Snowmelt	Rainfall[b]	Snowmelt	Rainfall[b]	Snowmelt	Rainfall[b]
Control (C)	n/a	0.9 ±0.4	n/a	0.5 ±0.1	n/a	4.3 ±1.3 b
Manure on frozen soil (MF)	n/a	8.9 ±7.1	n/a	0.4 ±0.3	n/a	39.2 ±33.1 a
Snow-covered control (SC)	0.8 ±0.3	0.8 ±0.3	<0.1	0.3 ±0.6	3.6 ±1.2 c	7.5 ±5.4 b
Manure on top of snow (MTS)	76.4 ±22.1	3.3 ±1.6	<0.1	0.3 ±0.3	254.5 ±14.8 b	12.1 ±4.6 ab
Manure between snow (MBS)	113.8 ±46.1	2.7 ±0.5	<0.1	0.3 ±0.2	230.6 ±56.8 b	11.1 ±1.3 ab
Manure under snow (MUS)	205.3 ±36.6	5.9 ±2.7	<0.1	<0.1	362.4 ±66.5 a	18.3 ±5.1 ab

	DRP		Total P	
Treatment	Snowmelt	Rainfall[b]	Snowmelt	Rainfall[b]
Control (C)	n/a	0.1 ±0.1	n/a	0.8 ±0.2 b
Manure on frozen soil (MF)	n/a	1.0 ±0.7	n/a	4.0 ±2.9 ab
Snow-covered control (SC)	0.1 ±0.1	0.1 ±0.1	2.2 ±0.8 c	1.6 ±1.1 ab
Manure on top of snow (MTS)	2.4 ±0.7	1.1 ±0.3	10.9 ±0.8 a	2.9 ±0.7 ab
Manure between snow (MBS)	1.5 ±0.6	1.2 ±0.3	8.5 ±1.5 b	2.8 ±0.3 ab
Manure under snow (MUS)	0.7 ±0.2	2.8 ±0.7	9.1 ±1.5 ab	4.5 ±1.3 a

[a] Values are means ± standard deviation (n/a denotes not applicable).
[b] Rainfall values are sums of surface runoff and subsurface leachate losses.

Alterra (2011-partB) confirmed the necessity of this measure and stated that the application of fertilizers and manures to water-saturated, flooded, frozen or snow-covered ground is associated with very high risk for surface run-off/leaching of N and P, possibly resulting in water pollution.

Moreover, they point out that such applications of fertilizers and manures are not effective as in these situations there are no growing crops and, consequently, no demand for nutrients (Alterra, 2011-part B).

1.2.3. Environmental co-benefits and drawbacks

Waterlogging can create anaerobic conditions which can cause denitrification. Denitrification is the process of anaerobic bacteria in soil converting nitrate-N into gaseous forms like nitric oxide, nitrous oxide and di-nitrogen. Nitrous oxide (N_2O) is a potent greenhouse gas. The primary N loss mechanism from saturated, fine-textured soils may be denitrification. Many studies have observed denitrification increasing when water filled pore space is above 60%. Denitrification will only occur if microbes have access to a food source in the form of soil organic matter, crop residues, or manure. Adding N fertilizers to soils that are already saturated may cause gaseous N losses. In contrast, waiting for the upper few centimeters of soils to dry reduces the risk of losing applied N through denitrification. Soil water content and soil N availability were found to be co-required for high N_2O emissions (Abdalla et al., 2011).

Fertilisation on waterlogged or water saturated soils is not generally expected of farmers, since they rather prefer frozen soils for their carrying capacity. However, the prohibition will certainly contribute to the maintenance of soil structure, avoiding heavy machinery being operated on wet soil.

1.2.4. Minimum/threshold or optimum level of implementation

Although this restriction is an effective measure, the restriction will not be equally relevant everywhere, as pedo-climatic zones vary in rainfall, monthly temperature and periods with frost and snow and also in the relative areas used for agriculture. Alterra et al. (2011-part B) provided an indication of environmental zones with a serious risk of nutrient emissions due to thawing. The indication for the risk of nutrient emissions due to thawing was derived from databases giving information on frost and precipitation: without frost there will be no thawing and generation of water.

The cost of this measure relates to the cost associated with sufficient storage capacity. This cost coincides, however, with the storage cost related to the closed periods, so this measure may not be experienced as a significant extra cost when considered in relation to the closed periods as a whole.

The measure as such is clear, effective, efficient and climate robust. The importance of the measure can change depending on climate change. Global climate change causes waterlogging events to be more frequent, severe, and unpredictable. The change of importance can differ between regions.

The measure can be experienced by farmers as contradictory to good agricultural practices, as they often appreciate frozen soils for their carrying capacity allowing land spreading of manures without negatively affecting the soil structure. European scientific analysis, even in expected low risk situations, could be convincing and boost willingness to comply with this measure. A clear and well underpinned rationale contributes to the respect of the measure.

Williams et al. (2011) referred to Zuzel and Pikul observing that high soil moisture contents at the time of soil freezing can significantly decrease the final infiltration rate (Zuzel and Pikul, 1987 in Williams 2011). From that point of view, fertilisation on frozen soil could be considered under certain conditions. However, the conditions would be very specific and not easy to define. Control and enforcement would be very difficult.

1.3. The capacity and construction of storage vessels for livestock manures.

1.3.1. Background

During specific periods of the year nutrients in manure cannot be utilized efficiently and pose a significant risk of water pollution. During these periods, manure must be stored.

1.3.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses.

Sommer and Knudsen (2021) referred to the so-called Harmony Rules in the late 1980s in Denmark, which introduced mandatory stipulations on manure storage capacity, application of manure during the growing season, and limitation on the number of animals per hectare agricultural land. Very high local surpluses of N and phosphorus were prevented. The authors stated that the ban on direct discharge of manure and the ban of application of manure in periods without or with limited plant nutrient uptake, were the two most important regulations that were implemented. Also in Denmark, Petersen et al. (2021), who analyzed three decades of trends in NO_3^--concentration in five agricultural catchments, revealed the importance of the manure storage capacity for decreasing stream NO_3^--concentrations.The mandatory storage capacity, being in line with these regulations, can be assessed as an effective measure, in combination with the prohibition periods. It is an effective, but also a costly measure. Construction of storages so as to prevent leakage and ensure impermeability is essential in order to avoid point losses and leaching. The risk of airborne emissions should also be considered.

1.3.3. Environmental co-benefits and drawbacks

Environmental co-benefits

Storage of manure allows to apply nutrients at the right time for crop uptake. The regulation of nutrient application to crops via manure application, by storing it during periods in which soil application should be avoided, allows to optimize crop production while decreasing nutrient leaching.

Environmental drawbacks

The storage of manure has other environmental consequences. Nutrient losses from manure storage facilities are mainly linked to emissions of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4), the latter two being greenhouse gases. The level of emissions from manure storage facilities depends on the type of manure storage (manure as a solid, liquid, or slurry) and whether or not the manure storage facility is covered (Table 4).

Table 4 Manure N loss (%) from different storage systems (Yang et al., 2011).

Storage systems *m*	Housing N loss (%)^{z}	Storage N loss (%)^{y}	Total N loss (%)	References
Liquid system				
1. Unlined lagoon	N/A	70	70	Lagoon, Table 3–8 (USEPA 2004)
2. Lined lagoon	N/A	50	50	70% of unlined lagoon
3. Open tank	5	24	28.5	Flush barn, Table 3–8 (USEPA 2004)
4. Tank below slatted floor	5	17	21.5	70% of open tank
5. Sealed covered tank	5	7	11.6	Tanks, Table 3–8 (USEPA 2004)
6. Other	5	17	21.5	The same as tank below slatted floor
Solid system				
1. An open pile on the ground without a roof	5	20	25	Stockpile, Table 3–8 (USEPA 2004)
2. An open pile on the ground with a roof	5	15	20	Solid fraction manure-cattle, Table 2 (Hutchings et al. 2001)
3. Manure pack in barns, pens or corrals	5	10	15	75% of an open pile on ground without a roof
4. An open pad without runoff containment	5	15	20	Solid fraction manure-cattle, Table 2 (Hutchings et al. 2001)
5. An open pad with runoff containment	5	8	13	Outdoor confinement areas, Table 3–8 (USEPA 2004)
6. A covered storage pad	5	4	9	50% of an open pad without run off containment
7. Other^{x}	5	15	20	The same as an open pad without run off containment

[z]5% N loss from housing is used in this paper due to the lack of measured data.
[y]Storage N loss rates were based on literature.
[x]Compost was grouped with "other manure" in the survey.

The ammonia emission rate is a function of the ammonia N concentration in the manure, ambient temperature, manure pH, manure moisture content, the exposed manure surface area and wind velocity near the surface. As many of these factors involve the storage surface, a common ammonia reduction strategy is to use a storage cover. According to Brusselman et al. (2016) reduction of the ammonia emission during manure storage is based on the following principles:

- Reduce the free surface of the manure storage;
- Reduce the ammonia source strength (by lowering the pH and the ammonium concentration);
- Leave the manure undisturbed as much as possible

So, lowering the pH of the slurry can be another way to reduce ammonia emissions from storage. With a system decreasing slurry pH to 5.5 by adding 4-6 kg concentrated sulfuric acid per ton of manure, the ammonia emissions were reduced by 70% (Pedersen, 2004). Finally, considering the three possible sources of emissions: manure management from livestock, manure applied to soils and excreta deposited by grazing livestock, manure management almost always predominates as the main emitter of ammonia, followed by manure applied to soils (Malherbe et al., 2022). When manure is stored as a liquid, it decomposes anaerobically. The combination of anaerobic conditions and degradable organic matter results in the production of a significant quantity of methane. The temperature and the retention time in the storage unit greatly affect the amount of methane produced. When manure is stored as a solid, it tends to decompose under more aerobic conditions and less methane is produced.

Direct N_2O emissions occur via combined nitrification and denitrification of nitrogen contained in the manure. The emission of N_2O from manure during storage and treatment depends on the nitrogen and carbon content of manure, and on the duration of the storage and type of treatment. Nitrification is likely to occur in stored animal manures provided there is a sufficient supply of oxygen, it does not occur under anaerobic conditions. Nitrites and nitrates are transformed to N_2O and dinitrogen (N_2) by denitrification, an anaerobic process. Nawara et al. (2021) concluded based on a literature study that an aerobic storage of solid and liquid manure would reduce the greenhouse gas emissions. Aeration would decrease methane emission, increase the N_2O emission but result in a net decrease of total greenhouse gas emission.

1.3.4. Minimum/threshold or optimum level of implementation

This measure is an easily controlled measure, supportive of the prohibition periods. Up to date N-excretion values for livestock with a clearly defined calculation and correct quantification of the required storage capacity will support the commitment of the farmers for this measure. The measure demands an investment of the farmers. Regarding the situation or state of art at the average farms and the efforts that generally still have to be made, a financial support measure can be considered. It will increase the ability and motivation of the farmers.

Adequate requirements for the construction of manure storages can reduce losses by volatile emissions. A minimum of prescriptions is necessary and should be included in legislation. The prescriptions enhance the efficiency of this measure to reduce N losses.

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2. CLUSTER II - SLOPING GROUND, WATER COURSES

2.1. Restrictions for application on steeply sloping ground

2.1.1. Background

A sizeable portion of agricultural activities takes place on sloping lands, with estimated 0.62 million hectare on slopes > 25% and 1.78 million ha on slopes between 15 and 25% within nitrate leaching vulnerable zones in the EU-27 (Velthof et al., 2014). On these lands, nutrient loss to surface waters can occur due to both overland-flow and soil erosion. In Flanders, about 25% of the agricultural area is vulnerable to erosion (Departement Leefmilieu, Natuur en Energie, 2011). The Nutrient Emission Model of the Flemish Environmental Agency (NEMO; Van Opstal et al., 2014) estimates the total yearly N-losses due to runoff to be 600 ton N, which equals to 2.5% of the total N losses from agricultural fields to the surface waters. In the Mediterranean area, soil erosion is one of the biggest environmental threats. The areas subjected to soil erosion in the Mediterranean area cover 1,309,000 km², equal to 15% of the land of Mediterranean countries (Cosentino et al., 2015). Globally, 20% of cropland is estimated to be affected by soil erosion (Prăvălie et al., 2021).

The impact of slope on average risk of soil erosion can be calculated with the S-factor within the Revised Universal Soil Loss Equation (RUSLE), which shows an increased risk of soil erosion starting from a slope of 9% (McCool et al., 1987; McCool et al., 1989). Zhang et al., (2015) found a similar relation between sediment yield and slope for farmland and fallow land in Southern China, concluding that starting from a slope of 7-15% erosion on farmland and fallow land became most severe, whereas forests and orchards showed smaller degrees of erosion at these slopes. Based on calculations of the RUSLE model, Alterra Wageningen UR (2011, the Netherlands) considered a full ban on fertiliser application on land with slopes of 8-15% or more to be judicious from a precautionary perspective, with the lower value of 8% referring to arable land, which is more vulnerable to soil erosion, and the upper value of 15% referring to permanent grasslands. Deviation from this rule could be made possible for direct injection of fertiliser and slurries into the soil. In the EU member states in general, restrictions for the spreading of fertilizers on sloping ground are applied to slopes as from >10% to >30%. Practical restrictions vary between member states, regions, etc. and include the following types of measures:

- total prohibition of any fertilizer application on any ground with slopes higher than a limit value,
- prohibition of fertilizer application on sloping ground along watercourses (with varying distances and in some cases depending on the presence of a grassy or perennial unfertilized buffer strip),
- limitation of the applied N-doses on sloping ground and/or obligation of splitting the fertilizer in several doses,
- prohibition of fertilizer application in certain periods (autumn-winter),
- obligation of measures to prevent runoff and erosion, e.g. injection or incorporation of fertilisers within 12 or 24h, terraces, intercropping, conservation tillage, tillage perpendicular to the line of maximum slope.

In addition, restrictions can be different depending on the application technique, the application of measures to prevent runoff and erosion, the applied fertilizer type (mineral or organic, liquid or solid, rapid release N fertilizers, etc.), the crop type (permanent grassland vs. arable crops), specific geomorphological and soil conditions, predicted rainfall, and the size of the field.

2.1.2. Assessment of the effectiveness and efficiency in terms of preventing and reducing nutrient losses

Generally, nutrient losses through erosion on sloping ground can be reduced by restrictions regarding the application of nutrients on the one hand, or by measures to avoid/reduce runoff and erosion on the other hand. Two main strategies to avoid erosion can be distinguished: (1) make the soil more resilient to erosion and (2) break the force of rain and runoff. This translates into some basic rules that can be used to limit erosion: maintain organic matter content and pH; avoid soil compaction so that water can penetrate the soil properly; keep the soil covered as much as possible with crops, e.g. green cover crops or crops with good (and fast) ground cover; keep crop residues/green cover residues on the surface by not ploughing; leave more clods in the seedbed; avoid preferential lines along which water flows more quickly, e.g. by installing "speed bumps" between ridges and erasing tire tracks.

In the following paragraphs, the effectiveness and efficiency of the most common practical measures to reduce nutrient losses on sloping grounds are reviewed.

2.1.2.1. *Limit fertilizer application on slopes*

Nutrient pollution caused by agricultural fertilizers can be directly controlled by limiting the amount of fertilizer applied. However, it should be kept in mind that this does not directly affect the pollution caused by the nutrients that are naturally present in the soil (Wang et al., 2023). Therefore, in many cases efforts still have to be made to reduce soil and water erosion even when reducing fertilizer application on the slopes. On the other hand, the application of organic fertilizer or manure can have a positive effect in reducing soil erosion by changing the physicochemical properties of the soil, such as the presence of macro aggregates, the soil organic carbon content, and exchangeable cations (Cui et al., 2023). Therefore, more emphasis should be put on reducing mineral fertilizer application than on organic fertilizer application.

As for the quantitative impact of reducing fertilizer application rates on slopes, results from peer-reviewed scientific studies within the EU are scarce. Rodríguez Sousa et al. (2022) found that on olive groves in Portugal, conventionally managed intensive and highly intensive groves used greater amounts of NPK fertilizers, which ultimately led to a higher risk of diffuse pollution. Intensive and highly intensive farms showed both higher erosion (up to 48 t/ha/year) and nitrate concentration (11-16 ppm) than integrated and organic farms (erosion up to 20 ton/ha/year and nitrate concentration up to 5-6 ppm), with part of this difference being attributed to the higher amounts of NPK fertilizer application on intensive farms. In this case, the organic farms had on average 60% less risk on diffuse pollution.

2.1.2.2. Effect of reduced tillage

Both non-inversion tillage and no tillage (no-till) have well documented effects on the reduction of soil erosion, often in combination with mulching of the soil (e.g. in Volker, 2012; Choudhary et al., 1997; Leys et al., 2007; Chalise et al., 2019; Keesstra et al., 2016; Tebrürrge & Düring, 1999). These reductions can in principle also result in reduced nitrogen and phosphorus losses, however, this reduction is certainly not guaranteed. Stevens and Quinton (2009) conclude that the effect of no-tillage on leaching losses is unclear, with no-tillage assumed to be the most extreme application of reduced tillage. Cooper et al. (2017) reported no effect of shallow non-inversion tillage and direct drilling on soil water NO_3-N and P concentrations compared to conventional ploughing. Furthermore, even when erosion and/or run-off of N and P is reduced, care should be taken, especially for nitrogen, that these nutrients are not lost due to other pathways such as deep percolation or subsurface runoff (Xia et al., 2014).

2.1.2.3. Contour tillage

With contour tillage, tilling and sometimes planting is carried out along the contour lines of the land instead of up and down the slopes. In this way, water runoff is slowed down and infiltration is increased. This practice is effective on slopes with gradients between 2% and 10% (NRCS, 2007). Contour cultivation, especially in combination with the addition of organic matter and the mulching of straw, has been shown to reduce runoff depths as well as soil erodibility, resulting in a reduction of total N-losses by 33 to 63% (Zhang et al., 2016).

2.1.2.4. Contour vegetative barriers

Contour vegetative barriers are grass strips, hedges, or other vegetative barriers that are located within fields and act as barriers to overland flow (Stevens and Quinton, 2009). The effect on sediment losses is well established, with the effectiveness of contour grass strips reported as 15-79%, depending on both grass type and flow rate (Ligdi and Morgan 1995; Dabney et al., 1995). The effect on nutrient losses are expected to be similar to riparian buffer zones (Stevens and Quinton, 2009). Xia et al. (2014) reported significant effects on P but not on N using *Vetiveria zizanioides* and alfalfa contour hedgerows, with reductions of 31-47% in sediment P losses, 28-45% in surface runoff total P losses.

2.1.2.5. Mulching and use of crop residues

Crop residues can be retained on the soil surfaces to act as mulch and help in controlling wind and water erosion. Steven and Quinton (2009) report an average reduction in soil loss of 78% (with range of 40- 100%) from the use of crop residues. Crop residues will release nutrients as they are decomposed, and the fate of these nutrients is determined by the interplay of processes such as immobilization, mineralization, and plant uptake. Therefore, the use of crop residues has only been moderately successful in the reduction of nitrate and phosphate leaching (Thomsen, 2005; Schreiber, 1999; Stenberg et al., 1999; Sharley and Smith, 1991). On the other hand, crop residues are generally more successful in reducing N and P in overland flow (Mostaghimi et al., 1992; Tiscareno-Lopez, 20024; Torbert et al., 1999; Andraski et al., 2003).

2.1.2.6. Perennial crops

Perennial crops, such as perennial grass, can improve soil structure and stability, soil quality, and biodiversity in comparison to annual crops (Zegada-Lizarazu et al., 2010). This can be especially advantageous in regions where soil erosion is a serious concern. Cosentino et al. (2015) reported higher N losses through leaching with annually tilled crops (0.23-2.21 kg N/ha) compared with perennial crops (0.11-0.66 kg N/ha).

2.1.2.7. Cover crops

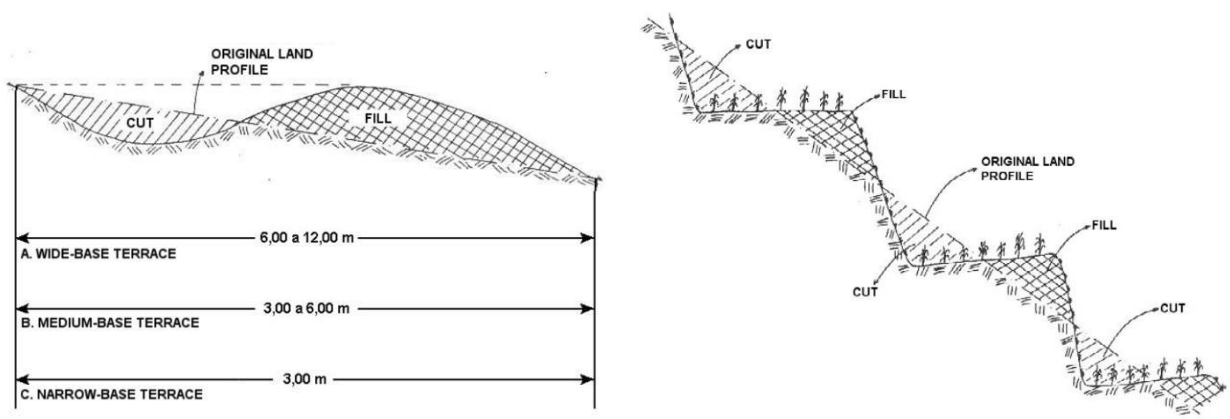
Cover crops or catch crops are effective in the reduction of soil erosion and nutrient losses through leaching. Even on non-sloping soils, cover crops reduce nitrate leaching directly by intercepting nitrogen that would otherwise be lost from the plant-soil system. The quantitative impact of these reductions, although widely reported, is very mixed, with a review of literature showing reductions between 0 and 98%, with an average annual value of 48% (Stevens and Quinton, 2009). One example from the UK reports a decrease in nitrate leaching losses in soil water of 75-97% by using an oilseed radish cover crop relative to fallow, but had no impact upon phosphorus (P) losses. Corresponding reductions in riverine NO_3-N concentrations however were not observed (Cooper et al., 2017). On the other hand, the long-term frequent use of cover crops can also increase the risk of nitrate leaching, by increasing the mineralizable nitrogen in the soil. Thus, in years when the catch crop develops poorly, or if the use of catch crops is discontinued, the potential for nitrate leaching is enhanced (Aronson & Torstensson, 1998). Cover crops can also reduce nitrate losses by impacting overland flow. In theory, increased infiltration and evapotranspiration should reduce the volume of overland flow, however in practice both increases and decreases in the nitrate concentration of runoff are reported. When comparing cover crops with bare fallow treatments, a review of Sharpley and Smith (1991) reports the effect of cover crops as ranging from an increase of 31% to a reduction of 87% in nitrate concentration of the runoff. These differences are attributed to site-specific climatic, soil, and crop factors.

2.1.2.8. Terracing

Terracing is one of the oldest measures taken to reduce soil and water losses on sloping ground and are a distinctive feature of the agricultural landscape in Europe (Stanchi et al. 2012). There is a wide variety in the types of terraces, which can be classified according to different criteria (Dorren & Rey, 2004):

1) their main function: *retention* vs. *diversion* of the rainfall runoff
2) the construction process: *channel* (only excavation of soil from uphill side) vs. *ridge* (excavation from both sides of the embankment)
3) the size of the terrace base: *narrow* (soil movement < 3m) vs. *medium* (3-6m) vs. *wide* (6-12m, Figure 5)
4) the terrace shape: *common/normal* (as shown in Figure 5, left) vs. *bench* (Figure 5, right)

Figure 5 Left: diagram of a common terrace form, with illustration of the classification according to difference base sizes. Right: diagram of a typical bench terrace (FAO, 2000).



This classification is relatively detailed, and most publications refer mostly to bench terraces, contour terraces, and parallel terraces; with contour terraces following the contour lines of the terrain, and parallel terraces being constructed parallel to each other (Brown et al. 2021).

The advantages and disadvantages of terracing are summarized in Table 5.

Table 5 Overview of advantages and disadvantages of terracing, based on reviews by Brown et al. (2021), Dorren & Rey (2004), Deng et al. (2021) and Stanchi et al. (2012)

Advantages	Disadvantages
Reduction of runoff and soil loss due to water erosion	Rainwater saturation of the soil can lead to higher storm runoff
Can act as a major store of catchment nutrients	Disturbance of the soil can lead to lower soil quality and fertility in the first few years after construction
Increased soil moisture content and crop yields	Labour cost for construction and maintenance
Conservation of plant biodiversity on a local scale	Large risk of land degradation if management and maintenance are not carried out (sufficiently)

As with many of the measures used to combat soil and water erosion, there is a large amount of research and broad agreement on the effect of the measure on soil and water erosion itself, but the effects on soil nutrient cycling are much less studied, as noted specifically for terracing by the review of Stanchi et al. (2012). These processes are generally more complex, because the nutrients can take different pathways from the soil to the surface- and groundwater and may undergo chemical changes along the way. Generally, and theoretically, however, it can be expected that terracing reduces the risk of N and P losses to surface- and groundwater, because the increased retention of catchment nutrients in the soil will be coupled with increased crop uptake, which will remove (most of) the nutrients from the soil and water cycle. However, this underlines the importance of soil cover during the year, for example with cover crops during the off-season. Additionally, when coarse material is used as filling material for the terraces, rapid water drainage can occur, leading to water stress and increased nutrient losses (Stanchi et al., 2012). This again emphasizes the importance of the design and management of the terraces.

Quantitative figures for the effect on N and P are scarce; however, the effects that are found are in agreement with the expectations stated above: higher nutrient availability on the terraces, and reduction of losses. We have summarized these studies in Table 6.

Table 6 Quantitative effects of terracing on N and P, literature overview

Source	Location	Effect of terraces
Chen et al. (2021)	China (diverse regions)	Increase in soil total nitrogen (TN) of 5, 6 and 43%
Qiang et al. (2020)	Loess plateau of China	Increase in soil organic carbon (SOC) of 50% and soil TN of 75%
Wacha & Giley (2023)	USA (general)	40% decrease in nitrogen (N) and phosphorus (P) loads

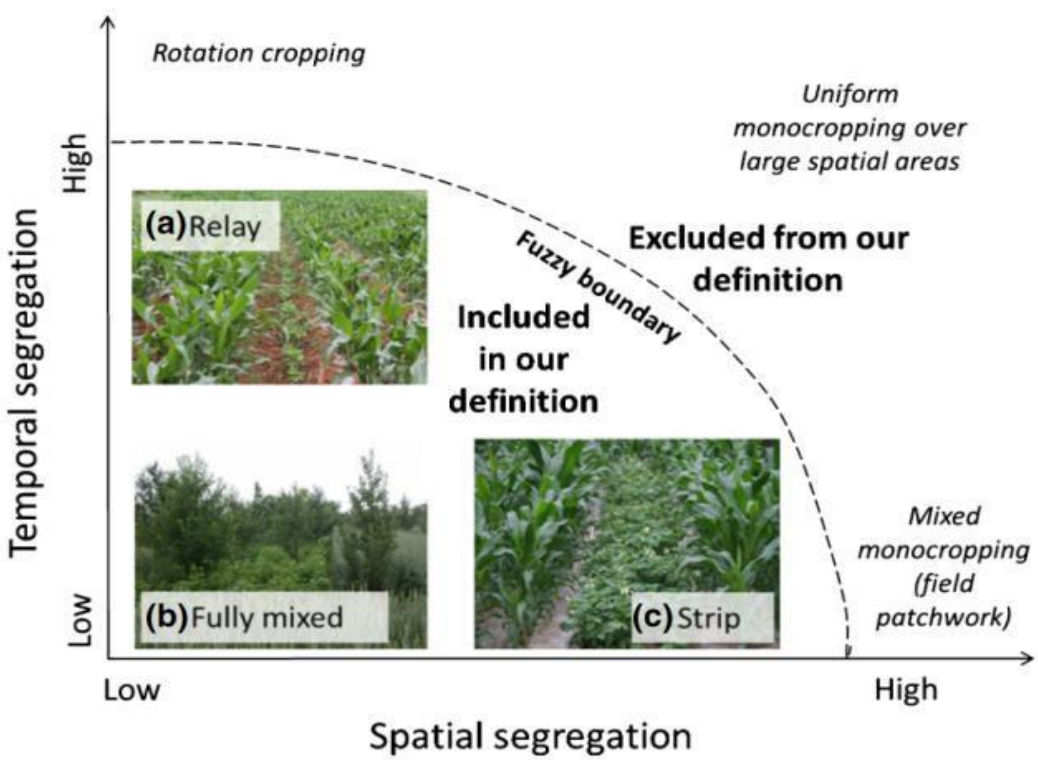
Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources, & Iowa State University College of Agriculture and Life Sciences (2013	Iowa, USA	77% decrease in P losses
Bezu & Tezera (2019)	Tigray, Ethiopia	Increase in plant-extractable P of 32% and SOC of 200%

2.1.2.9. Intercropping

Intercropping is the practice of growing multiple crop species at the same time in the same place (Figure 6). Different types of intercropping are distinguished:

- Mixed intercropping: two or more crop species are grown together at the same time in a field without using any particular spatial configuration.
- Strip or row intercropping: two or more crop species are grown in separate, but adjacent, rows at the same time.
- Temporal intercropping: a fast-growing crop is sown together with a slow-growing crop, so that the fast-growing crop is harvested before the slow-growing crop starts to mature.
- Relay cropping: a second crop is sown during the growth, often near the onset of reproductive development or fruiting, of the first crop, so that the first crop is harvested to make room for the full development of the second (overlap with the technique of under-sowing of a grass cover crop, e.g. in maize).

Figure 6 Definition of intercropping based on the temporal and spatial segregation (source: Brooker et al. 2015)



The primary goal of applying intercropping is to obtain complementarity between the different crops: available resources are partitioned and more completely utilized and competition is minimized as a result of different resource acquisition traits: differences in rooting depth, phenology, vegetative architecture, etc. Moreover, one crop species can provide limiting resources or improve environmental conditions for the other species, e.g. a legume crop providing nitrogen or a taller crop providing physical support for the other crop (Bybee-Finley & Ryan, 2018). In practice, farmers also use intercropping to reduce soil erosion, prevent nutrient losses and suppress weeds, as well as provide other ecosystem services such as moisture management, pest reduction, and to increase aggregate yields.

Advantages and disadvantages when compared with monocropping systems are summarized in Table 7.

Table 7 Overview of advantages and disadvantages of intercropping (Brooker et al., 2015; Bybee-Finley & Ryan, 2018; Jørgensen et al., 2000; Blessing et al., 2021; Marques et al. 2010; Jensen et al., 2020)

Advantages	**Disadvantages**
More complete exploitation of resources (solar radiation, water, soil nutrients, fertilizer)	Not always suitable for mechanization in an intensive farming system
Improved biodiversity, e.g. by providing a habitat for a variety of insects and soil organisms that would not be present in a single-crop environment	Undesirable when a single standard product is required
Pest reduction by increasing predator biodiversity by increasing barriers against biological dispersal of pest organisms through the crop: trap cropping, repellant intercrops, push-pull cropping, etc.	Lack of economies of scale for labour and time management
Weed control	Aggregate yield is not always increased
Reduction of use of N-fertilizer (in case of intercropping with legume crops)	Competition for resources (water) in certain conditions
Pollution mitigation; reduction of nutrient losses	
Input of organic carbon to the soil; improved soil structure	
Similar and even increased aggregate yields	
More stable aggregate yields, decreased risk of crop failure	
Erosion control by continuous soil cover (in time and in space)	

Regarding the reduction of nutrient losses to the environment, intercropping provides a more efficient use of nutrients in the soil and therefore can be useful in reducing nitrate leaching. For instance, a 3-year study in China on strip intercropping with sweet corn and soybean showed a 40% reduction in soil mineral N at the time of harvest compared to the corn monoculture (Tang et al., 2017). During a 2-year experiment in Czech Republic, mixed intercropping of winter wheat and white clover reduced mineral nitrogen losses by 20% compared to sole winter wheat (Kintl et al., 2018).

Finally, in erosion-sensitive crops, such as maize and perennial row crops such as vineyards and orchards, intercropping can provide an efficient soil cover between crop rows, offering a solution against erosion on sloped fields. This effect is reported in several studies regarding the use of intercrops in row crops to reduce surface runoff and soil erosion (Table 8).

Table 8 Erosion reduction through intercropping

Source	Location	Crop	Soil type	Slope	Reduction of surface runoff	Reduction of soil losses
Biddocu et al. (2017)	Italy	vineyard	clay – clay-loam	15%	50-66%	72-89%
Gomez et al. (2009)	Spain	olive orchard	sandy loam	11%	64%	98%
Kincl et al. (2022)	Czech Republic	maize	Haplic Luvisol	6.6-10.0%	0-66.2%	18-80%
Kabelka et al. (2021)	Canada	hop	Luvic Cambisol	9-17%		9-82%
Marques et al. (2010)	Spain	vineyard	sandy loam	14%	0%	93%
Novara et al. (2011)	Sicily	vineyard	Vertic Haploxerept	15.9%		68%
Wall et al. (1991)	Canada	maize	Luvisol	5%	45-87%	46-78%
Zuazo et al. (2008) (review)	Mediterranean	olive&almond orchards		30-35%	18-78%	51-94%

2.1.3. Environmental co-benefits

- Reducing nitrous oxide (N_2O) losses to atmosphere when using less N fertilizer;
- Reducing loss of fertile soil;
- Reducing risk of mudslides during heavy rainfall.

2.1.4. Minimum/threshold or optimum level of implementation

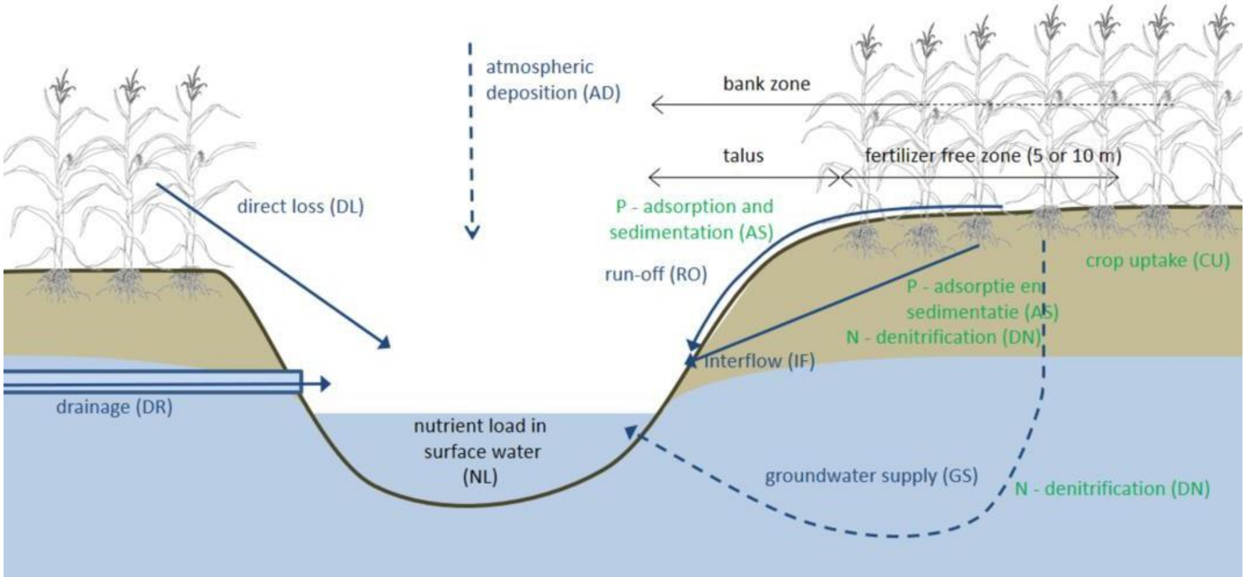
As illustrated in the discussion of the effectiveness of the different measures used to combat nutrient losses on steeply sloping ground, the effectiveness of these measures is in many cases reported as a very wide range. For some measures, not enough research is available to even safely determine this range. While the effect of these measures on reducing soil and water erosion is generally quite well understood and can be modelled using equations such as the RUSLE, the subsequent effect on N and P losses is dependent on relatively complex processes such as immobilization, mineralization, and plant uptake. These processes are in turn dependent on site-specific parameters such as climate, soil type, slope, crop type, and land management. Therefore, the impact on N and P losses is much more variable in results from field studies and is much more difficult to model. Based on this review of existing literature, the use of cover crops and limiting the mineral fertilizer application on slopes show the most well-documented impact on limiting nutrient losses on steeply sloping ground.

2.2. Restrictions for application near watercourses (buffer strips)

2.2.1. Background

As shown in Figure 7, nutrients can end up unintentionally in surface water through various transport routes: deposition during fertilizer application (direct losses), atmospheric deposition, run-off and interflow (subsurface drainage), leaching (groundwater supply), and drainage. During transport as well as in the surface water itself, nutrients are subject to various processes resulting in their retention, conversion and/or removal, including crop uptake, physical processes (adsorption, sedimentation), chemical and microbiological processes (denitrification).

Figure 7 Nutrient loads in surface water: transport routes, processes and factors (Tits et al., 2018).



Nutrient transport via atmospheric deposition and drainage (through drainage pipes) will not be further discussed, since buffer strips along watercourses have little or no influence on these routes.

For Flanders, Tits et al. (2018) calculated that the total amounts of nutrients ending up in watercourses via direct deposition during fertilisation are relatively small, compared to the nutrient loads via other transport routes (runoff, drainage). However, they represent peak loads in surface water that temporarily cause high nutrient concentrations and are therefore important for surface water quality.

In order to reduce nutrient losses to surface water via these transport routes, buffer strips can be implemented along watercourses. A distinction can be made between fertiliser-free buffer strips (i.e. unfertilised zones in which the same crops are grown as in the rest of the field) and vegetated buffer strips (i.e. unfertilised zones in which a permanent cover such as grass is grown). Vegetated buffer strips specifically designed to decrease nutrient transport from agricultural fields to surface water are further discussed under Cluster V (buffers, nature-based solutions).

In the EU, restrictions for the application of fertilizers near surface water apply in most of the Member States. The distance to the waterbody that must be respected in specific conditions vary from 2 to 100 m, depending on a number of factors, such as the slope of the field or of the zone along the watercourse, the application equipment and/or the incorporation of the fertilizer, the surface area and/or width of the field, the type of water body (lake, watercourse, etc.), the type of fertilizer (mineral, organic, liquid, solid, etc.), the presence of a (permanent) plant cover, hedges or wooded bands, the amount of fertilizer applied and the type of crop (row crops or other crops).

2.2.2. Assessment on the effectiveness and efficiency in terms of preventing and reducing nutrient losses

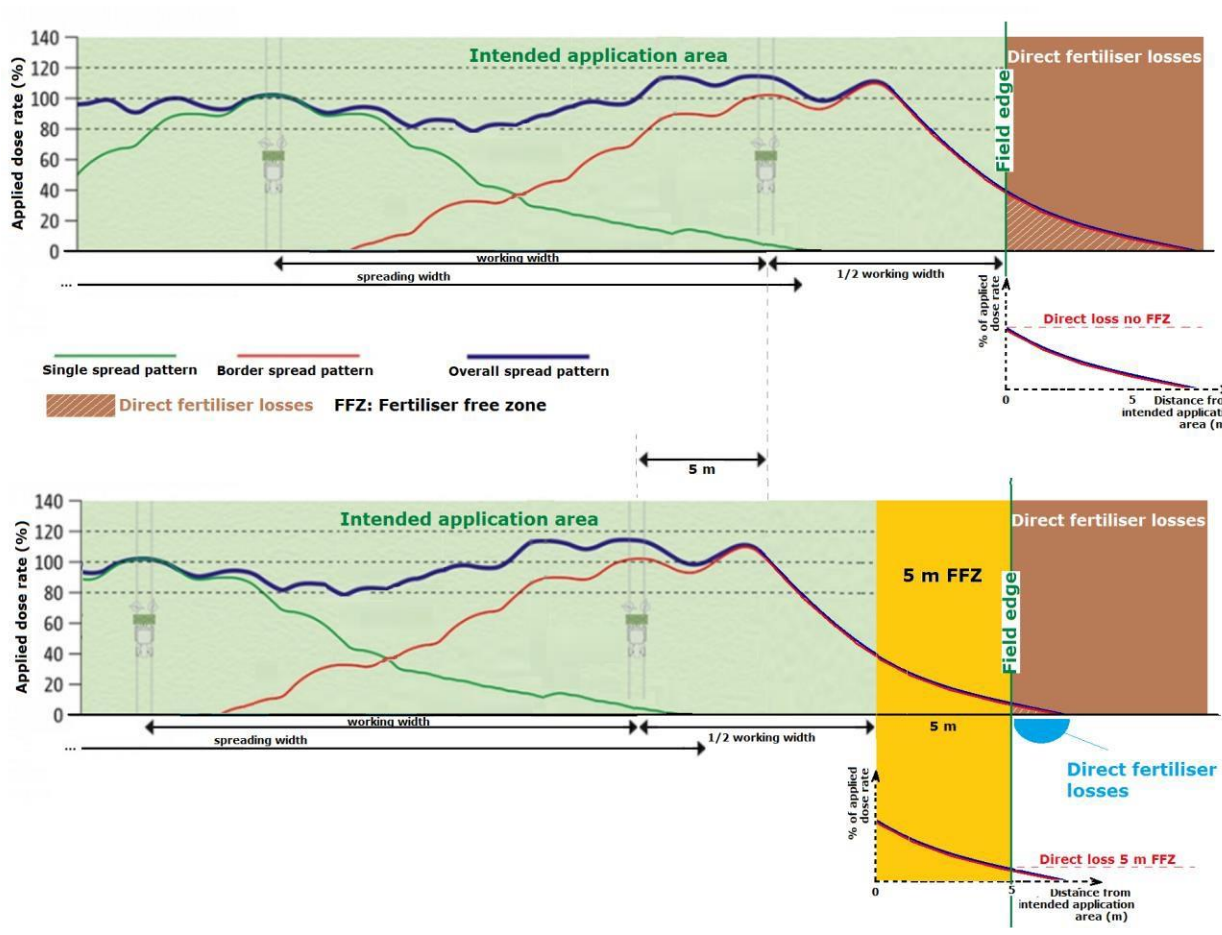
A fertilizer-free zone bordering surface waters can both prevent or reduce direct deposition of fertilizer in the water. In addition, the reduction of nutrient input near the watercourse reduces the risk of nutrient loss due to overland-flow and soil erosion of the nutrients applied close to the water body.

2.2.2.1. Direct deposition of nutrients during fertilisation

Direct deposition of nutrients in the watercourse during fertilisation can be reduced directly by the implementation of a fertiliser-free zone bordering the surface water (Figure 8). In a study, Tits et al. (2018) calculated that for Flanders, direct deposition of fertilizer in the surface water decreased by a little over 50%, when a fertilizer-free zone of 5 m along all "classified" streams is imposed. This would increase to a little over 80%, when this would also be imposed along all "non-classified" streams. Finally, direct deposition of fertilizer would decrease with 97% when the fertilizer-free zone would increase to 10 m. This measure, however, resulted in a loss of economic revenue for the farmers, due to lower crop yields.

Other efficient measures to reduce or even prevent direct deposition of nutrients in the watercourse during fertilization are the use of precision techniques such as row fertilization, injection of manure, use of border spreaders, etc.

Figure 8 Spread pattern and direct fertilizer losses without (top) of with (bottom) fertilizer free zone (FFZ) (Tits et al., 2018).



2.2.2.2. Surface runoff and subsurface drainage

Tits et al. (2018) showed that the potential loss from N and P applied on the 5-meter strip next to the water body on steeply sloping grounds is 10 to 12 times as large as the direct N deposition in the watercourse and could account for 1% of the total nitrogen transfer from agriculture to surface waters in Flanders (2.6% for P). Surface runoff is influenced by slope (see above), soil texture, soil drainage class and (winter) ground cover. Regarding surface runoff, an unfertilized buffer strip will have an effect on the travel time of the nutrients, because nutrients are deposited at a greater distance from the watercourse and therefore have a longer travel time to migrate to the watercourse. An increase in travel time means that there are more opportunities for other processes to take place during nutrient transport, e.g., denitrification or crop uptake if a crop is present in the unfertilized buffer strip. The increase in travel time is highly dependent on the soil physical properties (mainly moisture balance) and the local slope next to the watercourse. The mechanisms and efficiencies involved here are discussed more in detail in Cluster V.

2.2.2.3. Economic feasibility

Unfertilized buffer strips will likely result in the loss of economic revenue for the farmers, due to lower crop yields in the case of unfertilized strips or due to the loss of arable area in the case of vegetated unfertilised strips. The loss depends on the width of the strip.

2.2.3. Environmental co-benefits (and drawbacks)

2.2.3.1. Co-benefits

- Improved biodiversity
- Increased carbon sequestration
- Creation of attractive landscape elements

2.2.3.2. Environmental drawbacks

- Increased travel time of nitrogen to migrate to the watercourse can lead to increased ammonia volatilization and nitrous oxide emissions. At the same time, ammonia and nitrous-oxide emissions would be reduced because no fertilizer is applied in the buffer strip.

2.2.4. Minimum/threshold or optimum level of implementation

Since the effects of buffer strips on direct deposition of nutrients in the watercourses as well as on surface and subsurface runoff and drainage depend on numerous factors, such as application technique, crop, soil type, slope, etc., thresholds or optimum levels of implementation are difficult to identify.

Thresholds for buffer strip widths in respect to subsurface nitrate removal (groundwater nitrate) are discussed in Cluster V.

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3. CLUSTER III - BALANCED FERTILIZATION, APPLICATION METHODS, FERTILIZER PLANNING AND RECORD KEEPING

3.1. Limitations on fertilisation application rates, including reduced application rates

3.1.1. Background

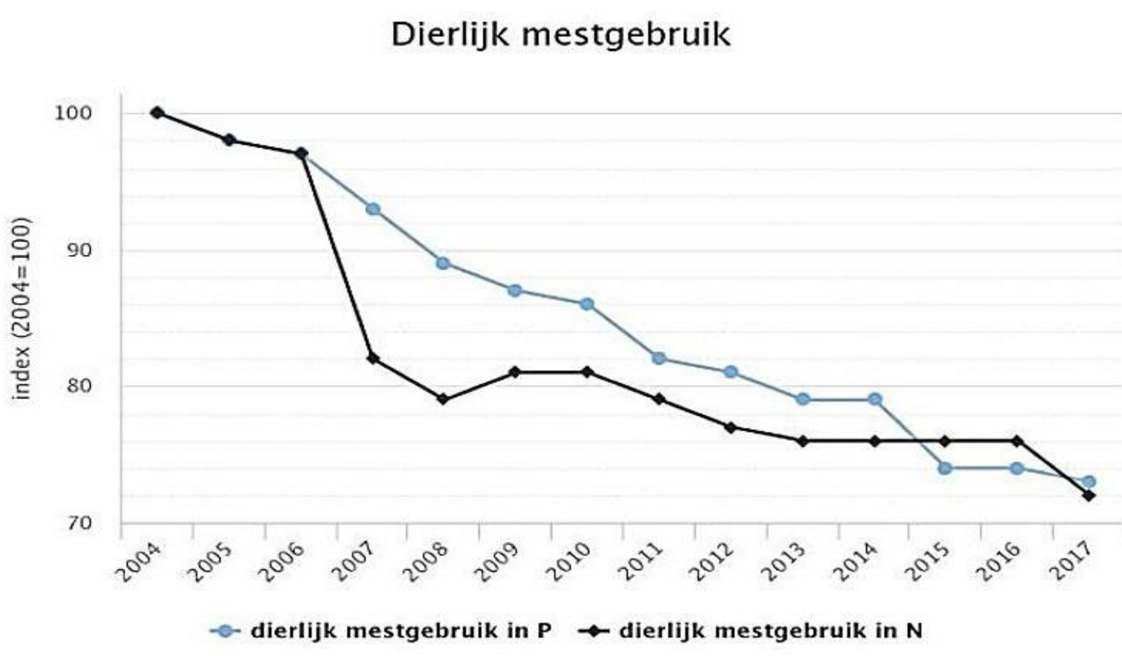
This measure could be simply categorized as a form of supply and demand. Supply should be aligned with crop demand. An oversupply of a nutrient, more than what the crop needs, can sometimes and to a very limited extent lead to higher uptake by plants. But most of the oversupply will accumulate in the soil and will be lost to the environment at a certain moment. From that point of view, limitations of fertiliser application rates are required to protect water and the environment against the loss of the surplus of nutrients. The application rates should be based on the crop requirements, considering all available nitrogen forms.

3.1.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses.

The **limitations on fertilisation application rates** expressed in fertilisation standards, set a clear and explicit limit or line. Without a clear and legally defined limit, fertilisation would certainly be higher. In that respect, this measure has the potential to limit the nitrogen input and to be one of the determinant factors in the reduction of the N-surplus. The N-surplus is an important indicator to evaluate the environmental impact.

The impact of an application standard on the use of a fertiliser and the input of nutrients can be demonstrated in Flanders. Nawara et al. (2021) concluded that a reduced application standards led to a reduced application of manure (Figure 9). Since 2007, the nitrogen application standard for organic manure was limited to 170 kg N ha^{-1}. Before 2007, the application standard was 200-250 kg N ha^{-1} for most common crops.

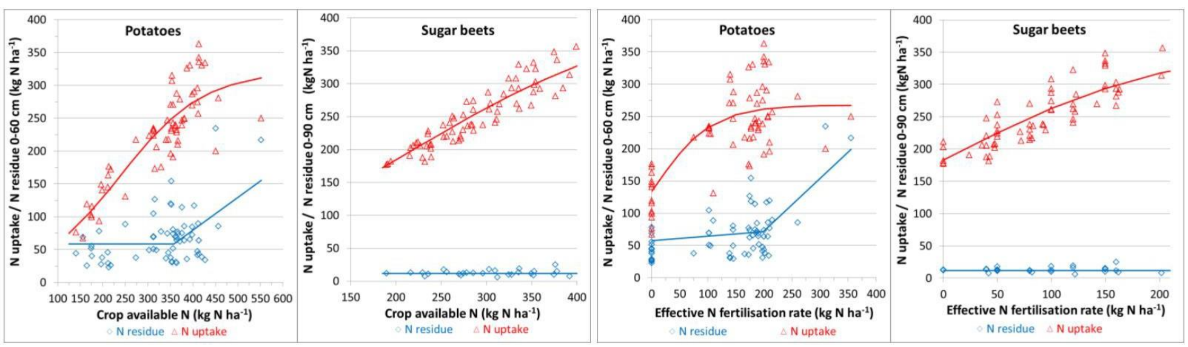
Figure 9 Use of manure since 2004 until 2017 (reference 2004), manure-N= black, manure-P= blue (Nawara et al. (2021) after Vlaamse Milieumaatschappij, 2019).



Van Grinsven et al. (2012) pointed to the limitation of “how much nitrogen in fertiliser and manure can be applied to agricultural land” as a major instrument to reduce nitrogen pollution in waters, together with the restrictions on where and when nitrogen in fertiliser and manure can be applied to agricultural land. They considered that the application limit for nitrogen from animal manure was the most important restriction following from the Nitrate Directive. They also referred to Mikkelsen et al. (2010) who concluded that in Denmark the reduction of N application standards alone contributed 32% of the total reduction of the soil N surplus between 1998 and 2004.

D’Haene et al. (2015) also states that limiting nitrogen (N) fertiliser application rates is one of the best N management measures to minimise N losses. Based on N-dose-response curves and residual soil mineral nitrogen in function of crop available N and applied effective N fertilisation rate for potatoes and sugar beets, they concluded that maximum fertilisation rates can be derived which limit the potential risk of NO_3-leaching during winter (Figure 10). For crops as sugar beets, the effect of a fertilisation standard is less pronounced since the residual soil mineral nitrogen is rather irresponsive for N-fertilisation. The range of N fertilisation evaluated, however, was only the range not negatively affecting tuber quality, an excessive N amount increases impurities and decreases extractable sugar content.

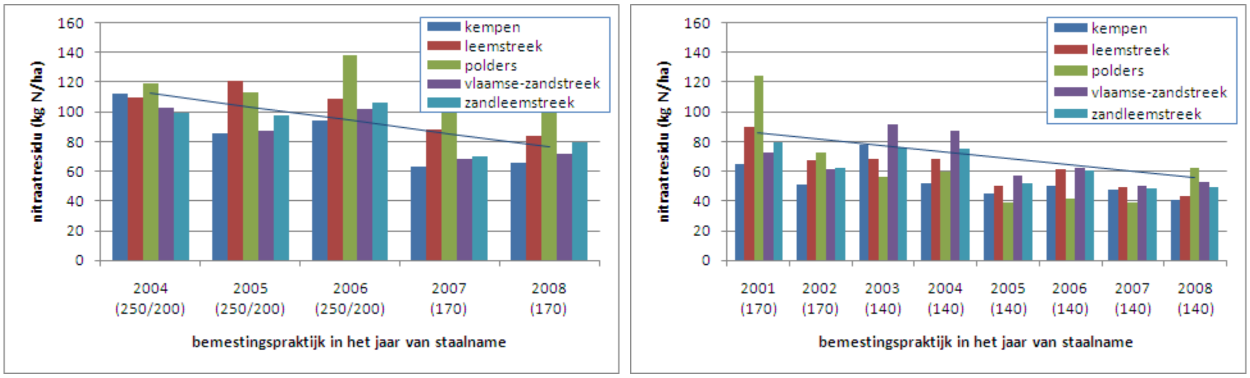
Figure 10 Nitrogen (N) dose-response curves (red triangle) and residual soil mineral nitrogen (rooting depth) (blue circle) in function of crop available N (left) and applied effective N fertilisation rate (right) for potatoes and sugar beets (D’Haene et al., 2015).



Also Van Overtveld et al. (2010) could relate fertilisation practice and by extent the application standards to the residual nitrate-N in Flanders (Figure 11). Fertilisation declined over the years due to stricter fertilisation standards, increasing attention for fertilisation and stricter application periods.

The applicable fertilization standards (standards for total N, manure-N, other organic N and mineral N) appeared all to have a significant effect on the total nitrate residue. However, for some the effect was positive, for others negative. The authors mentioned that fertilization standards were correlated with the year (because of changing standards over the years) and strongly depended on the cultivated crop. The effects found are usually a combination of crop effects and fertilization practices. (Van Overtveld et al., 2010).

Figure 11 Average residual nitrate-N per fertilisation practice/year and agricultural region (Van Overtveld et al., 2010).



Granlund et al. (2007) demonstrated the effect of a decreased N input on N leaching for the Finnish situation using the nutrient leaching model ICECREAM. The model results indicated that it is possible to decrease N leaching by adjusting and decreasing the fertiliser dose, without limiting crop growth and yield. The decrease in N leaching, however, was not linearly related to the reduction in N dose, due to strong annual and regional variation. The effectiveness of limitations on fertilisation application rates depends on the level of the application standards. High application standards based on unrealistically high yield figures will result in nutrient losses and have a negative effect on the environment.

The concrete values need to be properly substantiated and differentiated, then it will be an effective measure. Crops differ in their need and in their capacity to store nutrients (‘the plateau’ of the nutrient uptake curve). Sandy soils are more vulnerable to leaching than loamy and clay soils. Sandy soils are often characterized by a lower water holding capacity, a limited cation exchange capacity (CEC) and a limited rate of denitrification after leaching. For manure, the timing of the application determines the potential uptake still to be realised, the N needed, and it affects the possible supply of N via mineralization.

Reduced application rates can be beneficial as stated by Granlund et al. (2007). Nawara et al. (2021) noted that lowering N fertilization decreases nutrient leaching but also pointed to the risk of yield losses. The latter is economically disadvantageous and may be resisted by farmers. The possibility and effect of reduced application rates also depends on the ‘starting value’ on the defined application standards. The application standards are ideally based on a dose response curve. The economically optimal rate is strongly dependent on the economic situation (including the market prices of fertilisers and crops), can be highly variable, is not known at the beginning of the season, and does not necessarily coincide with agriculturally optimal dose. Depending on the crop and the starting point for the application standards, a further reduction might not necessarily result in a reduced residual amount of nitrate-N, prone to leaching (Figure 10), but could rather affect the yield. In literature, results on the extent to which N leaching is influenced by increasing N fertiliser application rates can be contradictory, so the response of N leaching to increasing N application and the dose response curve should be determined for a given environment, i.e., soil-climate-crop.

Nawara et al. (2021) pointed to the possibility of enhancing nitrogen use efficiency, for example by splitting fertilization in several doses. They also stressed the importance of thorough and long- term field research and a solid scientific and theoretical basis for the application standards.

Regulation of the use of animal manures is important since there are positive relationships between regional livestock densities and N and P concentrations in groundwater and surface water in Europe (Velthof et al., 2009). Animal manure is more difficult to manage than mineral fertilisers. Van der Straeten et al. (2011) referred to the impossibility to predict the exact nutrient availability and uptake as the reason for the precautionary fertilization standard for NVZ's of 170 kg manure-N ha^{-1} $year^{-1}$.

As pointed out in van Grinsven et al. (2015), the fertiliser equivalent (FE) of organic manures is important. Only a correct defined FE can lead to a correct fertilization. Underestimating FE will result in a higher dose as foreseen, maybe in a higher dose than the application standard allows and a higher risk for losses. Overestimating will result in shortage fertilization and loss of production and revenue for farmers. One of the determining factors for the extent to which the N supply from manures and crop demand are synchronised and N supply is utilised, is the ratio of mineral-N (Nm) and organically bound N (Norg) in the manure. One of the major factors determining the Nm:Norg ratio is the housing and storage. N-content depends on animal type, production level, feeding regimes, growth stage, etc. Therefore, values need to be defined regionally or nationally and reviewed at regular basis since production levels and feeding strategies evolve.

3.1.3. Environmental co-benefits and drawbacks

By reducing and regulating the input of nutrients, the surplus of nitrogen and nitrate leaching are reduced. The reduced input implies reduced emissions of ammonia (NH_3) and nitrous oxide (N_2O), the former contributing to air pollution and the latter a potent greenhouse gas.

To the extent the reduction in inputs is achieved through lower use of chemical N fertilisers, this also would reduce the demand for the production of such fertilisers, which is associated with emissions of carbon-dioxide contributing to global warming.

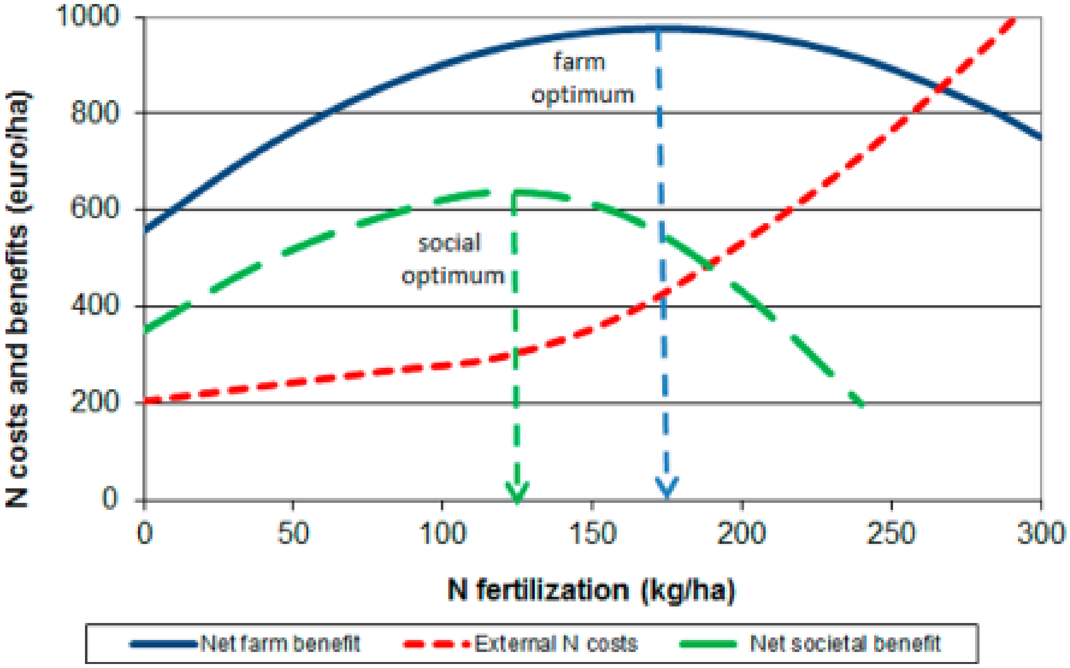
3.1.4. Minimum/threshold or optimum level of implementation

Limitations on fertilisation application rates are an effective measure to reduce the nutrient input, to limit the surplus and to reduce nutrient losses. It will be most effective and efficient if the concrete values are properly targeted and differentiated according to crop, soil type and other relevant factors. The consequence of having a legal system of application standards based on fertiliser equivalent nitrogen is the need of fixed statutory values for the fertiliser equivalency of livestock manures. So, the effectiveness and efficiency of the application standards also depends on the correct valuation of the fertiliser equivalency.

The cost efficiency of the measure depends on the farm type, the standard for mineral or organic fertilisers and the economic situation. If the economically optimal dose exceeds the application standard, the application standards imply yield loss and loss of revenue for the farmer. For arable farms, without livestock production, the application standards will generally limit the input of mineral fertilisers, which reduces costs for the purchasing of such fertilisers. On livestock farms the application limit for manure can imply a need to export manure from the farm and replace it by mineral fertilisers, both of which can be costly for the farmer.

As pointed out in van Grinsven et al (2013), there can be a mismatch between the private economic optimum of the farmer on the one hand and the social economic optimum for society on the other, due to the externality costs associated with fertilizer use, which fall on society as a whole (see Figure 12).

Figure 12 Mismatch between the economic optimum for the farmer and for society (van Grinsven et al (2013)



The application rates need to be evaluated regularly in a changing climate. A panel of Flemish experts assessed that climate change would impact the effectiveness of the nitrogen application standard negatively. Changing climate conditions, extreme events and longer lasting periods of drought or rain will impact the availability and uptake of nutrients. The effect of the changing climate will be more pronounced at higher doses, creating more uncertainty (Nawara et al., 2011).

3.2. Requirements on application methods to enhance efficiency

3.2.1. Background

Application of manure or fertilisers entails a risk of contamination of soil, water, and air. Not all of the applied nutrient can be recovered by the plants. The low recovery of N is responsible for environmental pollution. Nutrient use efficiency (NUE) is the ratio of the amount of N harvested trough the crop or taken up by the crop relative to the N inputs. A whole series of parameters can influence nutrient use efficiency. For manure, the situation is even more complex than for mineral fertilisers. The nitrogen applied in organic fertilisers will never equal the effect of the same amount of nitrogen applied in mineral fertilisers. This fact is what is referred to by the terms nitrogen fertiliser replacement value (NFRV), N equivalency of manure, or N efficiency of manure. Even when handled according to the best practices, manures have a NFRV smaller than 100%. As for the NUE, measures can be taken to increase the NFRV. This review will cover the measures regarding the application of fertilisers and manure (other factors e.g. affecting the composition of the manure could also influence the NFRV but will not be discussed here).

Ammonium (NH_4^+) contained in manure and fertilisers can be lost as ammonia gas (NH_3). Ammonia is emitted wherever manure is exposed to the atmosphere, also after manure application to fields. Urea-based fertilisers also result in ammonia emissions. Urea is rapidly hydrolysed by the enzyme urease to ammonium carbonate ($(NH_4)_2CO_3$) and ammonium ions provide the main source of ammonia. Ammonia volatilisation occurs when NH_3 in solution is exposed to the atmosphere. The extent to which NH_3 is emitted depends on the chemical composition of the solution (including the concentration of NH_3), the temperature of the solution, the surface area exposed to the atmosphere and the resistance to NH_3 transport in the atmosphere. Surface application of fertiliser involves higher risk.

Ammonia emissions from livestock manures during and after field application depend on:

- properties of the manure, including viscosity, total ammoniacal-N (TAN) content, C content and pH;
- soil properties such as pH, cation exchange capacity, calcium content, water content, buffer capacity and porosity;
- meteorological conditions including precipitation, solar radiation, temperature, humidity and wind speed;
- the method and rate of application of livestock manures, including, for arable land, the time between application and incorporation, and method of incorporation;
- the height and density of any crop present

Ammonia affects air quality but also later results in the deposition of nitrogen compounds on land and water bodies, effectively adding a (more or less) thin layer of fertilizer, which can cause eutrophication of natural ecosystems. Such deposition from the air can also be sufficiently significant that it should be taken into account in fertilizer planning on farms.

Manure and fertiliser applications raise soil nitrate (NO_3) levels. Mineral fertiliser may include nitrate directly (e.g. calcium-ammonium-nitrate (CAN)). Ammonium is converted to nitrate by micro-organisms in the soil through a process called "nitrification". Over time, organic nitrogen in manure is also gradually released through microbial activity (mineralization) in the form of ammonium, in turn making it available for conversion to nitrate (or release to the air as ammonia). The release of nitrates in the soil should ideally coincide with the demand for nitrates by plants, to limit the risk of losses. This indicates the importance of timing to enhance nutrient use efficiency. In summary, timing of application, application rate, and application method are all crucial for nitrogen use efficiency (Hanif, 2023).

3.2.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses.

Hutchings et al. (2010) subscribed the potential to reduce NH_3 emissions of different **application techniques**. They stated that emissions following application of slurry may be reduced if the slurry is applied in narrow bands (trailing hose), if the slurry is placed beneath the crop canopy (trailing shoe) or placed below the soil surface (injection) (Table 9). It was mentioned that incorporation of slurry and solid manures into tillage land can reduce NH_3 emissions by up to 90 %. The reduction in emission varies according to method of incorporation, interval between manure applications and incorporation and type of manure.

Table 9 Abatement techniques for slurry and solid manure application to land mentioned in Hutchings et al. (2010) (according to UNECE, 2007).

Abatement measure	Type of manure	Land use	Emission reduction (%)	Limits to applicability
Trailing hose	Slurry	Grassland, arable land	30 Emission reduction may be less if applied on grass > 10 cm. Poor reductions on bare land in some situations	Slope of land (< 15 % for tankers; < 25 % for umbilical systems); not for slurry that is viscous or has a large straw content
Trailing shoe	Slurry	Mainly grassland	60**	Slope (< 15 % for tankers; < 25 % for umbilical systems); not viscous slurry, size and shape of the field, grass height should be > 8 cm, difficult when crop residues present
Shallow injection (open slot)	Slurry	Grassland	70**	Slope < 10 %, greater limitations for soil type and conditions, not viscous slurry.
Deep injection (closed slot)	Slurry	Mainly grassland, arable land	80	Slope < 10 %, greater limitations for soil type and conditions, not viscous slurry.
Broadcast application and incorporation by plough in one process	Slurry	Arable land	80	Only for land that can be easily cultivated
Broadcast application and immediate incorporation by plough	Slurry	Arable land	80–90	Only for land that can be easily cultivated
Immediate incorporation by disc			60–80	
Broadcast application and incorporation by plough within 12 h	Slurry	Arable land	30	(according to § 10)
Immediate incorporation by plough	FYM (cattle, pigs)		90	
Immediate incorporation by plough	Poultry manure		95	
Incorporation by plough within 12 h	Solid manure	Arable land	50 for cattle and pig 70 for poultry	
Incorporation by plough within 24 h	Solid manure	Arable land	35 for cattle and pig 55 for poultry	

Notes:
1. */ Emissions reductions are agreed as likely to be achievable across the UNECE.
2. ** revised to incorporate conclusions of recent review.

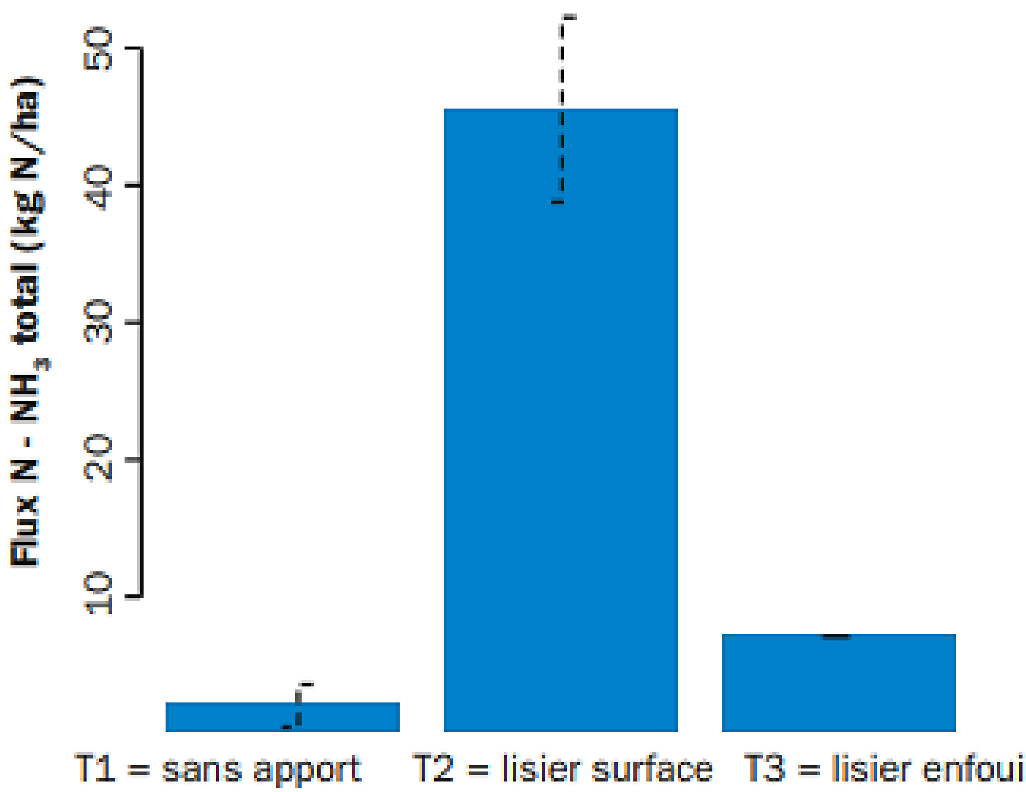
Huijsmans et al. (2008) described low emission application in the Netherlands. They concluded, both for grassland and arable land, that ammonia losses were reduced significantly by low emission application techniques (Table 10), increasing the NFRV of the manure.

Table 10 Ammonia loss (kg NH_3-N per 100 kg total ammoniacal-N applied (TAN) of cattle slurry applied to grassland and arable land, as affected by the application technique (Huijsmans et al., 2008).

Land use	Technique	Loss (kg N/100 kg TAN)
Grassland	Surface spreading	74
	Trailing shoes	26
	Sod injection	19
Arable	Surface spreading	69
	Shallow incorporation (e.g. rigid tine)	22
	Thorough incorporation (e.g. injection, rotavator)	2

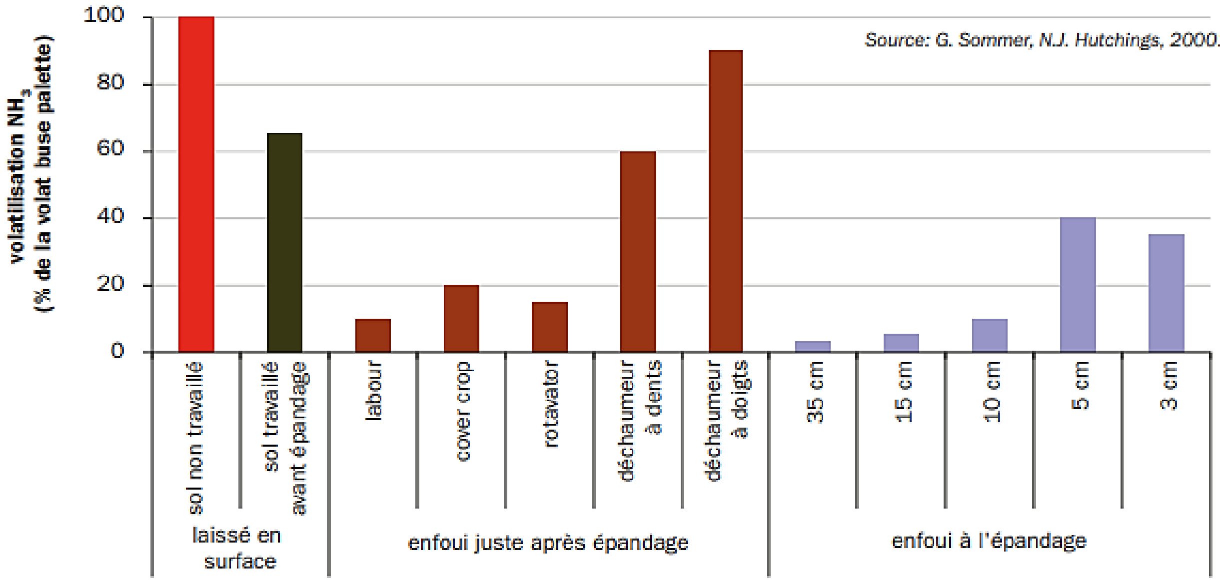
In French field trials a considerable reduction of N emissions was measured by application of pig slurry (Figure 13). If pig slurry was not incorporated 46 kg N of the 71 kg ammonium N was lost, i.e., 64 %. Immediate incorporation reduced the losses to 7 kg N, about 10 % of the applied ammonium N.

Figure 13 Amount of ammonia nitrogen lost by volatilisation on a testimony object without application of manure en two objects with application of pig slurry. Results of the CASDAR VOLAT NH3 project, Essai de Bignan (56), 2011 referred to in Levasseur et al., 2019.



Levasseur et al. (2019) confirmed the importance of the depth of incorporation, which was also mentioned by Huijsmans et al. (2008). They stated that an incorporation technique realising a homogeneous application at a minimal depth of 5-8 cm should be pursued, referring to Sommer and Hutchings (2001) (Figure 14).

Figure 14 Effect of different tillage tools on ammonia volatisation after a broadcast application of pig slurry, in percentage of volatilisation without tillage. Mentioned in Levasseur et al. (2019) referring to Sommer and Hutchings (2001).



Martin and Mathias (2013) analysed the potential of 10 actions to reduce ammonia emissions on French farms. They concluded that incorporation and application techniques showed the highest potential to reduce ammonia losses in France (Figure 15). It needs to be noted, however, that the potential of these measures is partly due to the fact that they cover all animal types and all types of farms.

As mentioned, ammonia losses are most likely in the case of urea fertilisers and manure. The low emission application techniques mentioned for manure application are applicable for mineral fertilisers containing urea. They need incorporation for the same reasons.

In case of row crops, the effect of precise placement of fertilisers on the yield of several crops was demonstrated by Nkebiwe et al. (2016) based on a meta-analysis (Figure 16). Several field studies showed that fertiliser placement resulted in 3.7% higher yield than broadcast and higher nutrient concentrations in different plant parts by 3.7% and nutrient content in above-ground biomass by 11.9% than fertiliser broadcast (Nkebiwe et al., 2016).

Figure 15 Reduction potential (kt. NH3) by 100 % application per practice or practice modality compared to 2010 (Martin and Mathias, 2013).

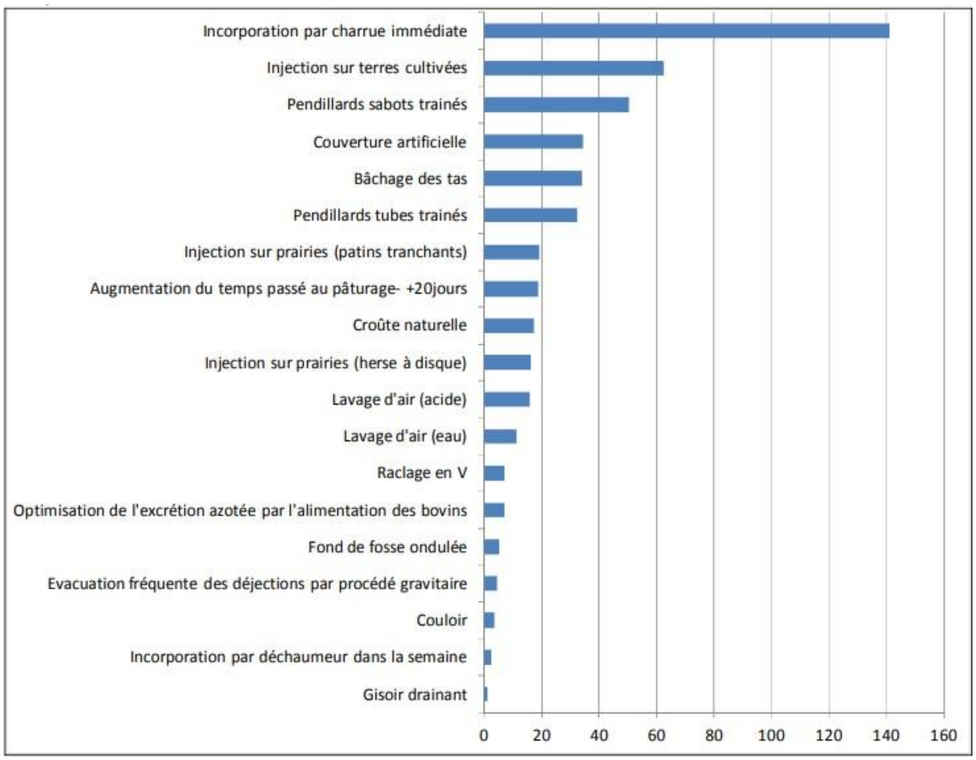
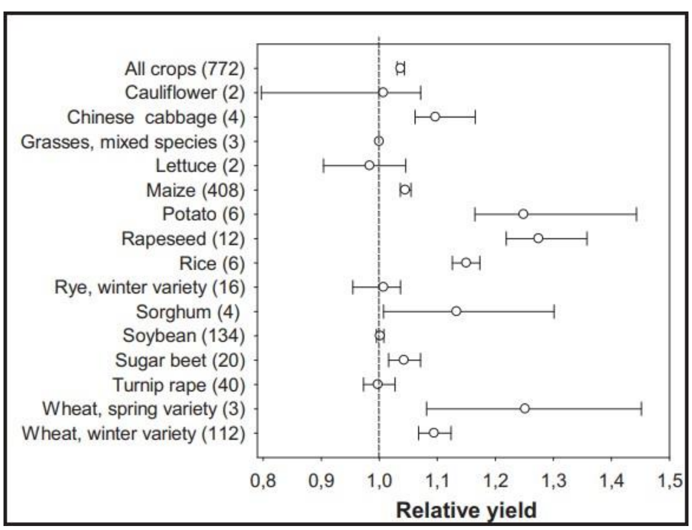


Figure 16 Relative yield of fertiliser placement by crop type. Y-axis: categories and number of datasets per category in brackets; X- axis: relative value of fertiliser placement to fertiliser broadcast; error bars, 95 % confidence intervals; Placement ≠ broadcast, if error bars do not include 1.; Relative values of a pair of categories are different from each other if their 95% confidence intervals do not overlap (Nkebiwe et al., 2016).

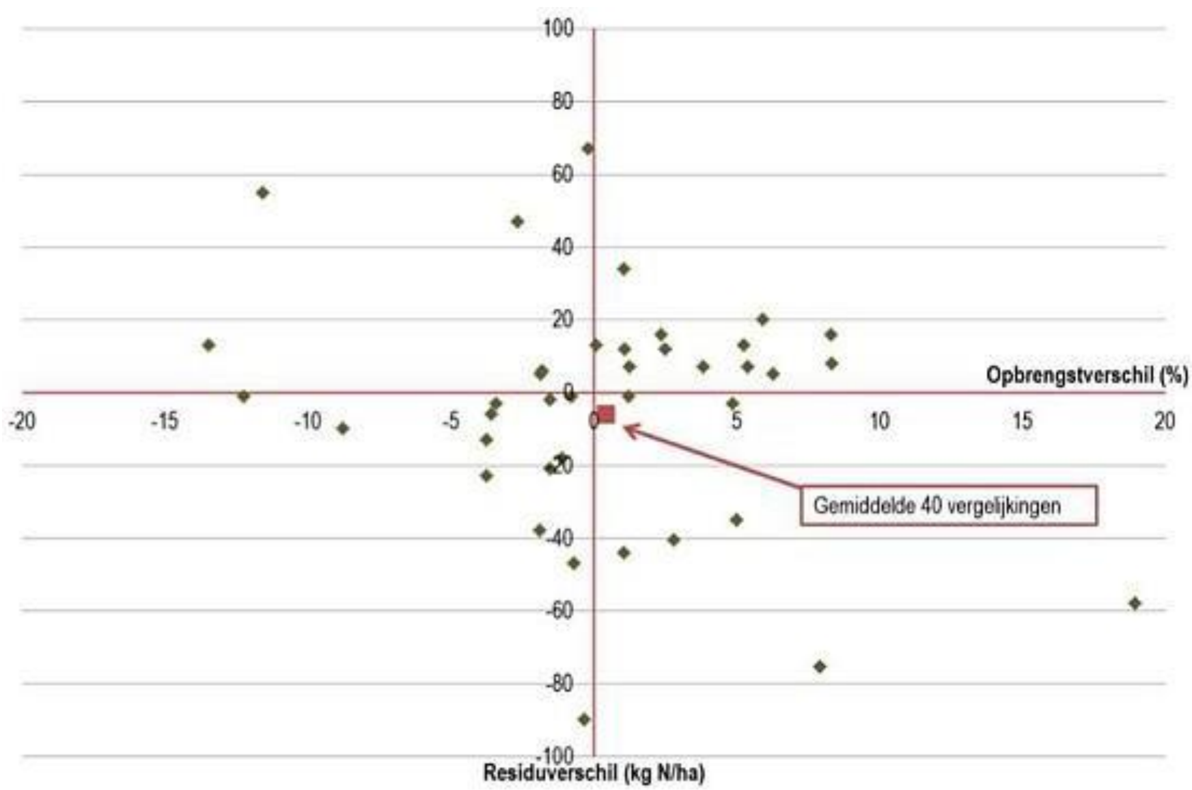


For potatoes, a crop with a relatively high risk of nitrogen losses, the positive effect on yield could not always be confirmed. However, Van Geel (2015) has pointed to the advantage of row application as a low-emission application technique. Maidl et al. (2002) found a positive effect on N recovery of nitrogen placement in the ridge, when the total N amount was applied at planting. In France, Cohan and Laurent (2012) demonstrated an equal yield with less nitrogen input.

In maize, row application is more common. In Flemish field trials it was demonstrated that equal or higher dry matter yields could be realised by row application at 75% of the broadcast dose (Odeurs et al., 2014). As a low emission technique with higher N-recovery in the crop and a possible lower N-input, row application could be an effective measure to limit the surplus and concomitant losses.

Regarding the **time of application**, plant-available N (especially nitrate containing fertilisers) should be applied close to the rapid growth stage of the crop, when N uptake is maximal. Synchronicity of N supply with crop demand is essential in order to ensure adequate uptake (Fageria and Balligar, 2005). In potato, Maidl et al. (2002) found for a broadcast application that fertiliser N recovery was higher when fertilisation was split in several doses as compared with a single application at planting. In Flemish field trials it was demonstrated that split fertilisation can reduce the residual amount of nitrate-N without yield loss (Figure 17). The timely administration of the last fertilisation dose is also important in this case (De Blauwer et al., 2014).

Figure 17 Effect of split fertilisation on the residual nitrate-N and total yield (difference=verschil= split fertilisation versus one application) (De Blauwer et al., 2014).



In maize, the indications on split fertilisation vary in the literature. In Dutch and Flemish research, merely based on yield considerations, it is currently stated that split fertilisation is not to be preferred in maize. In France, on the contrary, Soënen and Bouthier (2015) proposed a split fertilisation for maize to enhance nutrient use efficiency. Davies et al. (2020), in a study from Minnesota, found a higher recovery efficiency of the applied N with split fertilisation. Considering treatments PP100, Sp100 and TSp100, a clear beneficial effect on N-recovery can be noticed (Table 11).

Table 11 Recovery efficiency of applied N fertiliser as affected by treatment and year, averaged across locations (Davies et al., 2020).

Treatment †	2014		2015		2016	
	kg kg^{-1}					
C						
F100	0.433	a‡	0.504	bc	0.213	d
F125	0.403	a	0.423	cd	0.258	cd
PP100	0.383	a	0.363	d	0.418	abc
PP125	0.427	a	0.373	d	0.323	bcd
Sp75	0.388	a	0.644	ab	0.488	ab
Sp100	0.431	a	0.561	bc	0.482	ab
TSp50	0.468	a	0.543	bc	0.562	a
TSp75	0.386	a	0.707	a	0.423	abc
TSp100	0.496	a	0.643	ab	0.503	a

† C, control no N applied; F100, N applied in the fall at 100% of the recommended rate; F125, N applied in the fall at 125% of the recommended rate; PP100, N applied at preplant in the spring at 100% of the recommended rate; PP125, N applied at preplant in the spring at 125% of the recommended rate; Sp75, half of the N applied at preplant in the spring and half at V6 at 75% of the recommended rate; Sp100, half of the N applied at preplant in the spring and half at V6 at 100% of the recommended rate; TSp50, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 50% of the recommended rate; TSp75, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 75% of the recommended rate; TSp100, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 100% of the recommended rate.

‡Means within the column followed by different low case letters are significantly different at $p \leq 0.05$.

F100 en F125, the applications in fall, should not be considered.

Yield was sometimes lower after split fertilisation, but it can still be concluded that yield is generally not negatively influenced by split fertilisation (Table 5). Frerichs et al. (2022) showed in Germany the possibility of a differentiated nitrogen fertilisation strategy in spinach. The strategy of a base N fertilisation and a single top dressing could be further split resulting in a strategy of a base N fertilisation and two top dressings. They concluded that the nitrate concentration at risk of leaching could be considerably reduced by such further splitting of the top dressing. The timing of post-planting nitrogen applications is critical (De Blauwer et al., (2014), Soënen and Bouthier (2015)). One could summarize that the target species must be immature and growing to provide time for the nitrogen to be absorbed and metabolized in order to have the most efficient yield or quality impact.

Table 12 Maize grain yield as affected by treatment, location and year (Davies et al., 2020).

	Becker						Lamberton						Waseca					
Treatment†	2014		2015		2016		2014		2015		2016		2014		2015		2016	
	Mg ha^{-1}																	
C	6.0	e‡	4.9	e	4.9	g	9.5	e	5.0	d	6.3	d	4.5	g	6.9	d	6.1	d
F100	5.7	e	6.8	d	8.0	f	13.8	abc	13.5	a	13.9	ab	8.6	de	15.3	a	11.3	bc
F125	5.8	e	8.2	d	10.2	e	14.5	ab	14.2	a	15.5	a	9.8	bc	14.2	ab	12.1	abc
PP100	7.1	d	10.1	c	12.6	cd	14.4	ab	13.6	a	14.5	ab	10.3	ab	13.5	b	13.4	a
PP125	7.6	d	11.7	b	14.4	ab	14.7	a	14.2	a	14.9	ab	11.2	a	14.0	ab	13.6	a
Sp75	8.9	c	12.0	b	13.6	bc	13.2	bcd	12.0	bc	14.0	ab	9.2	cd	14.0	ab	12.9	ab
Sp100	10.0	bc	15.2	a	14.7	ab	13.9	abc	13.6	a	14.9	ab	9.9	abc	15.3	a	13.0	a
TSp50	9.8	bc	9.9	c	11.1	de	12.2	d	11.0	c	11.3	c	6.6	f	11.6	c	10.8	c
TSp75	10.5	b	12.9	b	13.9	bc	12.8	cd	13.1	ab	13.5	b	7.9	ef	14.1	ab	12.7	ab
TSp100	12.7	a	14.5	a	15.7	a	13.2	bcd	13.3	ab	14.7	ab	9.5	bcd	15.5	a	13.3	a

† C, control no N applied; F100, N applied in the fall at 100% of the recommended rate; F125, N applied in the fall at 125% of the recommended rate; PP100, N applied at preplant in the spring at 100% of the recommended rate; PP125, N applied at preplant in the spring at 125% of the recommended rate; Sp75, half of the N applied at preplant in the spring and half at V6 at 75% of the recommended rate; Sp100, half of the N applied at preplant in the spring and half at V6 at 100% of the recommended rate; TSp50, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 50% of the recommended rate; TSp75, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 75% of the recommended rate; TSp100, a third of the N applied at preplant in the spring, a third at V6 and a third at R1 at 100% of the recommended rate.

‡Means within the column followed by different lowercase letters are significantly different at $P \leq 0.05$.

According to Vogeler et al. (2020), the potential of **precision agriculture** technologies with site-specific N application rates depends on the N leaching response curve. They studied the marginal nitrate leaching in winter cereals in Denmark. The marginal nitrate leaching is defined as the increase in N leaching as a percentage of an additional unit of N applied. A linear marginal N leaching curve around the optimal rate indicates a situation in which potential N leaching reductions are small. With an exponential response of the marginal N leaching the use of precision agriculture will be more influential on N leaching across the field.

Bongiovanni and Lowenberg-Deboer (2004) did a meta-study in which they gathered data from studies about precision agriculture. The results related to variable N fertilization, as discussed by Bongiovanni and Lowenberg-Deboer (2004), are presented in Table 13.

Table 13 Review of studies cited by Bongiovanni en Lowenberg-Deboer (2004) considering variable N fertilisation as precision technique.

Crop	Input/Factor	Region	Methodology	Results of using VRT
Wang et al. (2003) Corn	N	Missouri	Used topsoil depth data to develop recommendations with a simulation model.	*VRT was more profitable than uniform rate in 75% of the cases, with a gain in profits up to \$37.14 ha^{-1} in one of the fields.
Roberts et al. (2001) Corn	N	Tennessee	EPIC simulation model to estimate N leaching.	*More N was applied with VRT, but less N was lost to the environment N leaching reduced by 2.24–4.48 kg ha^{-1}.
Delgado et al. (2001) Barley Potato	N	South Central Colorado	Field trials and use of information to estimate N leaching as a difference.	*N management practices can potentially be improved to reduce potential N losses and conserve water quality.
Kholsa et al. (2001) Corn	N	Colorado	Field trials and use of information to estimate N leaching as a difference.	*VRT-N has the highest NUE and lowest leaching compared to other treatments.
Whitley et al. (2000) Potato	N	Washington State	Field trials. Measured N leaching with probes.	*Surface soil had high NO_3–N flux. *Subsurface soil NO_3–N flux stable. *NO_3–N leaching was decreased in vulnerable zones due to a lower N rates in these zones.
Griepentrog and Kyhn (2000) Wheat Barley	N	Northern Germany	Field trials. Measured reduced chemical loading.	*VRA reduced N by 36% in low areas while maintaining the high yields.
English et al. (1999) Corn	N	West Tennessee	EPIC simulation model to estimate N leaching.	*VRT was more profitable than uniform rate and that it generated less nitrogen loss to the environment in most cases.
Rejesus and Hornbaker (1999)	N	Lake Decatur Illinois	EPIC simulation model to estimate N leaching.	*VRT-N has the potential to reduce the mean and variability of NO_3–N pollution, while improving profitability.
Thrikawala et al. (1999) Corn	N	Ontario, Canada	Simulation model (Barry *et al.*, 1993) to estimate N leaching.	*NO_3–N leaching reduced by 13%, average or between 4.2% and 36.3% in high and low fertility areas, respectively.
Watkins et al. (1998) Potatoes	N	Idaho	EPIC simulation model and dynamic programming to estimate N leaching.	*No environmental benefits. *No differences in N applied. *No differences in N losses.
Larson et al. (1997) Continuous Corn	N	Minnesota	LEACHM simulation model to estimate N leaching.	*No_3–N leaching was decreased in 50%, average, or from 60 to 29 kg ha^{-1}. *Decrease was 0 kg ha^{-1} in the loam, but 99 kg ha^{-1} in a loamy sand.
Leiva et al. (1997) Wheat Rapeseed Soybeans	Fertilizers Pesticides	Silsoe England	Measured chemical loading and estimated leaching with simulation.	*PA leads to savings in fertilizers and pesticides, decreasing risk of pollution and energy use, contributing to sustainability.
Hergert et al. (1996) Furrow Irrigated Corn	N	Nebraska	Measured after harvest residual soil NO_3–N and estimated N leaching.	*Improves N use efficiency. *Reduces leaching by minimizing high NO_3–N areas in the field.
Redulla et al. (1996) Irrigated Corn	N	Central Kansas	Measured after harvest residual soil NO_3–N and estimated N leaching.	*No differences in N use efficiency. *No differences in NO_3–N leaching.
Kitchen et al. (1994) Continuous Corn	N	Missouri	Measured grain production, unrecovered N in the crop, and post-harvest NO_3–N.	*The amount of unrecovered N decreased in the least productive soils with VRT. *Gross savings of \$10–\$12 ha^{-1} for using VRT.

Soto et al. (2019) published a study based on a survey of more than 500 farmers and contractors in Belgium, the Netherlands, Germany, France and Greece. The main focus was on arable land, more specifically wheat and potatoes, or cotton in Greece. The effect on yield, fuel and N fertiliser was estimated for a pessimistic, average and optimistic scenario for two precision agriculture techniques: machine guidance (MG) and variable rate nutrient technology (VNRT). Respectively 2.9 and 8.0 % applied N could be saved by machine guidance and variable rate nutrient technology (Table 14). Even without improved yields, the lower N input would lead to a reduced leaching potential.

Table 14 Estimation of the effect of precision agriculture techniques machine guidance (above) and variable rate nutrient technology (under) on yield, fuel use and applied N, according the survey of Soto et al. (2019) in Belgium, the Netherlands, France, Germany and Greece, focussing on wheat and potatoes or cotton in Greece.

	Pessimistic	Average	Optimistic
Yield	-4.0%	0.0%	4.0%
Fuel use	-2.4%	-5.4%	-8.3%
N applied	0.5%	-2.9%	-6.4%

	Pessimistic	Average	Optimistic
Yield	0.8%	4.1%	7.4%
Fuel use	0.6%	-5.4%	-6.3%
N applied	-4.6%	-8.0%	-11.7%

The literature referred to for the comparison in Table 14 is summarised below and it seems to confirms the potential of precision agriculture to improve N efficiency and reduce losses:

Machine guidance

- From the literature survey: Controlled Traffic Farming (CTF) can lead to improved fertiliser use efficiency, uptake of fertiliser can be improved by around 15%;
- Presentation Ulrike Kloeble (JRC workshop Sevilla, 2015): parallel tracking and auto- guidance can save 2-5% of N fertiliser and also 2-5% fuel use, based on results from the FutureFarm project;
- Wösten et al. (2016): Fuel consumption can be reduced by 5-10% by using guidance systems

Variable rate nutrient technology

- Koch et al. (2004): Site-specific fertilisation can reduce the total amount of fertiliser used, indicating an increase in Nitrogen Use Efficiency (NUE).
- Raun et al. (2001): average NUE increase of more than 15% in winter wheat in Oklahoma, USA.
- FutureFarm project: variable rate application can save 2-20 kg N per ha;
- Presentation Ulrich Adam (CEMA): 5-30% reduction in fertiliser use due to precision fertilization;
- Presentation from Wilfried Winiwarter (JRC workshop Sevilla, 2015): variable rate application can reduce fertiliser application on average by 24% (range 8-40%), based on several US studies.

The estimated effect of applying these two techniques at EU scale on leaching, NH_3 emissions and N fertiliser use is given in Table 15. The reduction in fertiliser use with the full application of both techniques would be in the order of 5 % of total N fertiliser use in the EU (the total is around 10 million tonnes of N annually). The relative reduction in leaching is in the same order of magnitude.

Table 15 Annual fertiliser, NH3 emission and N leaching and runoff reductions at EU-28 level for different precision agriculture adoption scenarios (Soto et al., 2019).

Adoption scenarios	**Fertilizer savings**	**NH_3 emissions**	**N leaching and runoff**
	kton N	**kton N**	**kton N**
Machine guidance >100 ha	72	3	15
Machine guidance >50 ha	88	4	19
Machine guidance full application	120	6	26
VRNT >100 ha	275	13	56
VRNT >50 ha	336	16	70
VRNT full application	457	22	98

3.2.3. Environmental co-benefits and drawbacks

Nawara et al. (2021) noted that the low-emission techniques often represent a trade-off between NH3 and N2O-emissions, as the deeper placement of manures will reduce ammonia losses but can increase N_2O emissions. Depending on the applied manure type and the application method, the increase of N_2O emissions will differ. Split applications in turn require additional rounds to spread fertilisers on the fields, resulting in higher fuel use and related emissions. Precision farming can have added benefits in reducing GHG emissions. Soto et al. (2019) indicated the possible impact of precision agriculture on greenhouse gas emissions (Table 16). They considered the direct soil N_2O emissions from lower fertiliser application, the indirect N_2O emissions resulting from N volatilisation and N leaching and the GHG emissions from fertiliser production and fuel use for field operations.

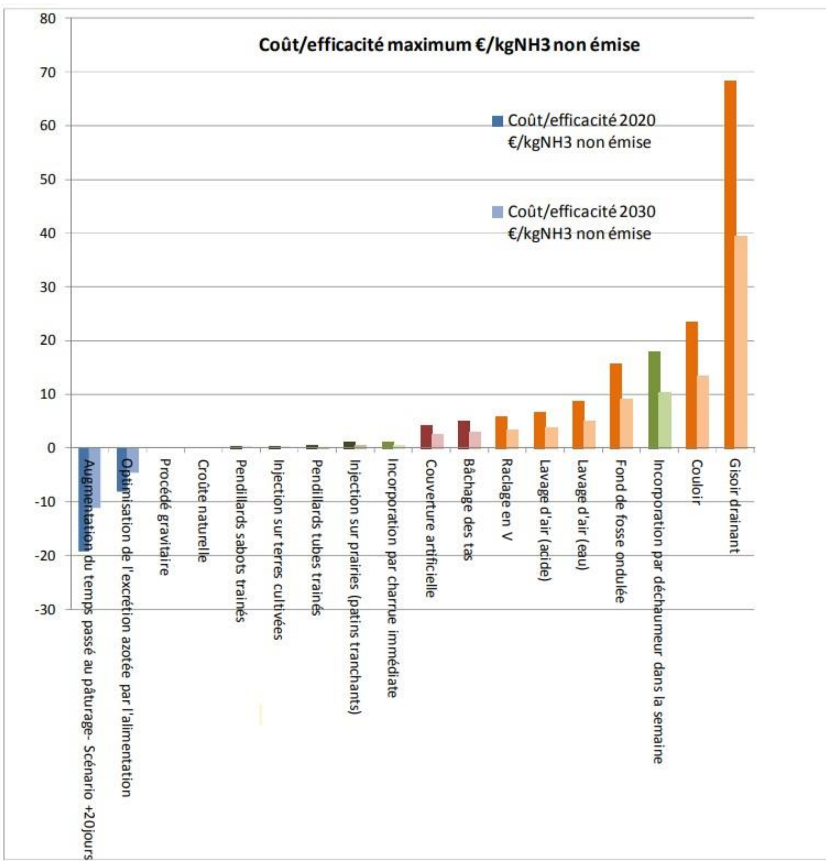
Table 16 Annual greenhouse gas savings at EU-28 level for the MG and VRNT optimistic, average and pessimistic scenarios, based on adoption on farms > 50 ha (Soto et al., 2019).

Savings	**MG**			**VRNT**		
[kton CO_2-eq]	**Pessimistic**	**Average**	**Optimistic**	**Pessimistic**	**Average**	**Optimistic**
Direct N_2O	-98	413	1258	904	1572	2299
Indirect N_2O	-21	87	263	190	327	471
Fuel use	363	891	1370	363	891	1370
Fertilizer production	-119	499	1520	1092	1899	2778
Total GHG savings	*125*	*1889*	*4410*	*2549*	*4689*	*6918*

3.2.4. Minimum/threshold or optimum level of implementation

The importance of low emission techniques could increase in a changing climate. More variable weather and extreme weather conditions are likely to limit the number of suitable moments for fertilisation. Every method that can limit the impact of the weather conditions on losses will therefore become more important and more effective in a changing climate. A panel of Flemish experts also did not consider it likely that there would be any negative impact of climate change on the effectiveness of this measure (Nawara et al., 2021). In a study in France, low emission application techniques were also estimated to be a cost-efficient measure with insignificant cost net cost per kg ammonia avoided (grey bars in Figure 18).

Figure 18 Cost-benefit ratio per practice or strategy, all animal categories included, to the horizons of 2020 and 2023, regarding an application rate of 100. (Blue: practices related to feeding, orange: practices related to the buildings, red: practices regarding storage, **gray: practices related to application**, green: post-application practices) (Martin and Mathias, 2013).



Split fertilisation does not necessarily require adapted machinery. Extra costs will concern eventual intermediate soil samples and the extra time that needs to be spent. It can be concluded that split fertilisation is a cost-efficient measure. The addition of an intermediate decision about the need (and level) of any further fertilizer applications makes this measure rather climate robust, even if unforeseen wet conditions may prevent timely treatment or dry conditions can prevent fertiliser from reaching crop roots. In dryer conditions, leaf fertilisation can be considered. However, this is not an option if conditions are too dry or in case of very high doses.

The different forms of precision agriculture (machine guidance, variable rate nutrient technology, variable irrigation) are often associated with high investment costs. The cost-benefit analysis of these techniques is therefore not always clear, also given that the environmental benefit is not directly felt by the farmer. The size of the farm or the relevant parcels and the diversity of the parcels will partly determine the feasibility and suitability of precision agriculture techniques. These techniques also demand a certain level of IT awareness, which can have a discouraging effect. In conclusion, while is has been demonstrated that precision agriculture can be effective in reducing nutrient losses, economic efficiency is not always clear, especially at the level of the farm. On the other hand, precision agriculture can be seen as a rather climate robust measure, since decisions can be made ad hoc in light of climatic factors.

3.3. Fertilisation plans, records on fertiliser use

3.3.1. Background

This point refers to ANNEX II (B) of the Nitrates Directive where is stated that the Member States may also include in their code(s) of good agricultural the establishment of fertiliser plans on a farm-by-farm basis and the keeping of records on fertiliser use. Focusing on the fertiliser plans and the keeping of records, it's useful to define some terms:

- Fertiliser plan: expected fertilisation practices
- Record keeping (fertilizer register): fertilisation practices actually carried out
- Nutrient balance: calculation of the nitrogen dose based on a nutrient balance.

The fertilisation plan is an overview at parcel level in which the farmer plans the use of animal manure and other fertilisers at the beginning of the fertilization season. The nutrient balance is not asked for in this point of the annex but is rather closely related to the fertiliser plan and could easily be confused with it. The separately effectiveness and efficiency of a fertiliser plan or a fertiliser register is not often scientifically defined or discussed. Nutrient management and fertilisation planning are rather discussed more generally in relation to the nutrient balance.

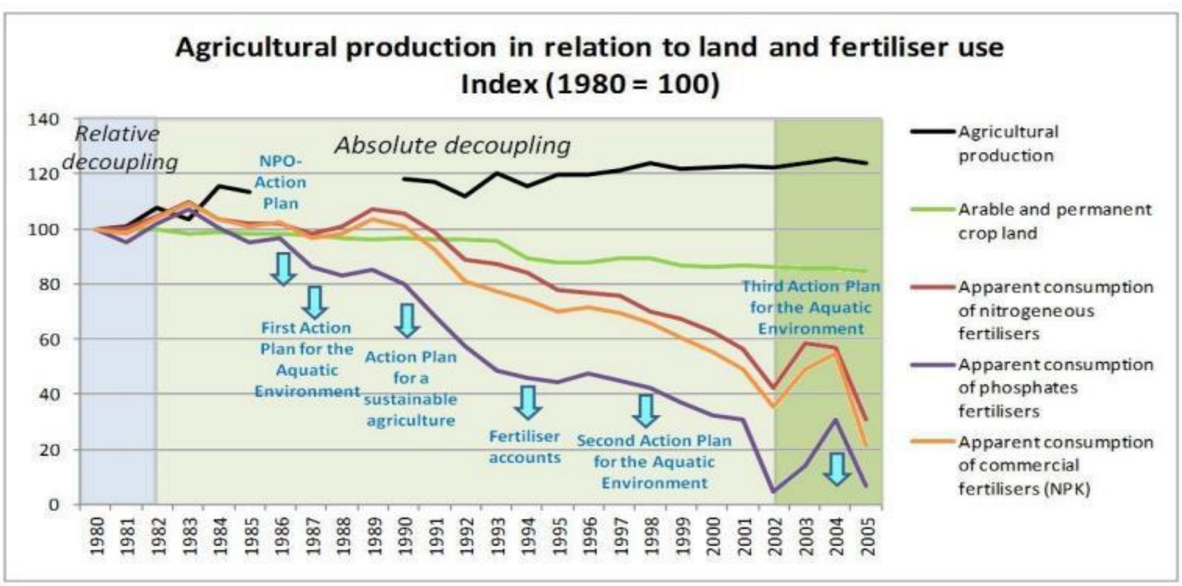
3.3.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses.

A fertiliser plan provides an overview of the planned use of nutrients on a farm. A fertiliser plan helps the farmers to plan the use of the different fertilisers, gives insight in the flux of fertilisers on the farm and draws the attention on how to apply each fertiliser best. It makes clear which fertiliser should be used for which crop and at which time. In fact, the plan helps to identify and fulfil the 4 R's of fertiliser application (i.e., right source, right rate, right place and right time). Where animals are present, it is necessary to estimate how much manure will be produced and be used and the amount and type (mineral, organic) of N it contains. The application of manure will have to be planned for each crop, as a result of which the need of additional mineral (or other) fertilisers will appear. At farm level, the possible overall shortages or surpluses of N will become apparent in the fertilizer planning. Alterra (2011) stated in a detailed analysis that in many cases, an improvement of the environmental performance of farms can already be achieved by using coarse default values for the various nutrient flows passing the farm gate. The fertilisation plan can be oriented after predetermined and generally applicable application limits or be based on an analysis of the specific need, determined for each crop or parcel at the farm. In either case the fertilisation plan should reduce over-fertilisation and by consequence reduce nitrate leaching.

Determining the input or the fertilisation dose on a specific nutrient balance per crop or at parcel level is the more elaborate and time-consuming exercise. In this case, the estimated need and nitrogen export is based on the production capacity of each crop realised at the farm. The N dose can be further specified by soil sampling at the start of the growing period. Fertilisation planning based on soil testing, a crop fertilisation plan and a post-hoc nitrogen balance, was evaluated as an effective tool for achieving balanced economic and environmental benefits in crop farming (Popluga et al., 2016). Valkama et al. (2013) indicated that with knowledge of field and crop properties fertilization planning could reduce N input by 20-75 kg/ha for cereals in Finland. The N-balance could be reduced by 10-40 kg/ha/year, indicating a possible reduction of nitrogen losses. Osmond et al. (2015), considering nutrient management planning as the primary tool to reduce nutrient losses from agricultural fields in America, pointed out that the effectiveness of this measure ultimately depends in large measure on its practical implementation by farmers.

The registration of the fertiliser use will encourage the farmers to respect the application standards and reduce nutrient losses. On the one hand, the records on fertiliser use can be used as a control system by governments to control compliance with application standard or N-budget at the farm and to get the closest possible indication of fertiliser use and overall N fluxes at national and regional level. On the other hand, farmers themselves can also use it as a control instrument. They can put the registered use of fertilisers against the fertiliser plan and the results in the field. In that way, it can be used for a post-hoc nitrogen balance and be an element in steering the farm towards more sustainable fertilization practices. The effectiveness of this measure may depend on its enforcement. With outside controls or mandatory registration, it can be a measure to induce farmers to respect the application standards, to reduce over-fertilisation and reduce nutrient losses. Without enforcement, a requirement of fertilizer planning is less likely on its own to help to avoid nutrient losses or lead to a sustainable fertilisation. In Denmark, farmers registered in the Register for Fertilizer Accounts are required to prepare a fertilizer plan, to calculate the nitrogen-quota for the farm and to submit a fertilizer account. The fertilizer plan must be prepared before the growth season begins and the fertilizer account must be submitted after the growing season is over. Fertilisers purchases and manure production and transfers are registered as well as the balance stored. Figure 19 shows that the introduction of the fertiliser account was part of the measures resulting in a reduced use of fertilisers.

Figure 19 Decoupling trends in agricultural use of fertilisers in Denmark. Agricultural production in relation to land and fertiliser use 1980-2005 (Tan and Mudgal, 2013).



In summary, the fertilisation plan encourages the farmer to think about the manure present on the farm, the needs of the different crops, the moment at which N is needed, the need of mineral N and the most suited application method. Since the fertilisation plan will be oriented to a certain application limit, being the application standard or the specifically determined N dose, farmers will have a clear view on the limits and over-fertilisation is discouraged and will be reduced. Reduction of over-fertilisation will result in reduced nitrogen losses. However, the fertilisation plan only reflects an intention. If not implemented, the measure will have no effect. A fertilizer register or record that reflects the actual fertilization practices on the farm can be a useful instrument for both government and the farmer. If planning is also based on soil sampling and determination of the amount of mineral nitrogen, especially nitrate-N, in the soil, the measure can be even more effective but also more costly.

3.3.3. Environmental co-benefits and drawbacks

Popluga et al. (2016) mentioned a reduction of direct N_2O emissions by 47-187 kg CO_2-eq/ha by fertilisation planning and the associated reduced N fertiliser input.

3.3.4. Minimum/threshold or optimum level of implementation

A fertilisation plan and registration of the fertiliser use demands some time of the farmer but is free. A reduction of the fertiliser dose and/or a higher nutrient efficiency by a better planning, can result in savings on fertilizer purchases, so it can be a cost neutral measure in practice. This may be the case even where a more elaborate nutrient management planning with soil sampling is considered.

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4. CLUSTER IV - LAND MANAGEMENT, VEGETATION COVER IN SENSITIVE PERIODS, IRRIGATION SYSTEMS

4.1. Land use management, crop rotation, permanent crops

4.1.1. Background

According to the Nitrates Directive, Annex II, Member States may include in their code(s) of good agricultural practices items regarding "land use management, including the use of crop rotation systems and the proportion of the land area devoted to permanent crops relative to annual tillage crops". Crops differ from each other when it comes to their sensitivity to nitrate leaching. This is partly due to differences in agricultural practice: fertilization, soil preparation, harvest date, crop rotation, etc. In addition, the properties of the crop also play an important role, for example the crop rooting depth, the efficiency of nitrogen uptake and the growing period.

Maize, especially continuous maize cropping (monoculture), is known to be very sensitive to nitrate leaching. In the Netherlands, the median nitrate concentration in soils with maize cropping was observed to be 4 to 8 times higher than in soils with grass (https://www.rivm.nl/landelijk-meetnet-effecten- mestbeleid/nieuwsbrieven/verkenning-gewasspecifieke-nitraatuitspoeling-in-lmm). Maize is more sensitive than grass to nitrate leaching due to a smaller root system, a shorter period of nitrogen uptake and the frequent soil tillage inducing increased mineralisation of soil organic N. In a field experiment in Bulgaria, the losses of nitrate nitrogen out of the root zone were significantly reduced in cereal crops compared to maize (Simeonova, 2024).

Vegetable crops may also lead to considerable N losses through leaching during winter and pose a threat to meeting water quality objectives, due to their often large amounts of crop residue biomass with high N contents (Agneessens et al., 2014). From literature review on nitrate leaching in 40 vegetable crop species, Zemek et al. (2020) concluded that more nitrogen is leached under field vegetable cultivation than under arable and grassland. Specifically cabbage species showed a high N leaching potential, whilst the risk of leaching was considered lower for most leafy vegetable species. Main influencing factors are the amount and N-content of crop residues and the rooting depth.

An illustration of median nitrate concentrations in groundwater and soil moisture for a number of crops in different soil types in the Netherlands is given in Table 1 (van Duijnen, 2020).

Table 17 Median nitrate concentration in groundwater / soil moisture per soil type for a number of crops in 2009-2017 in the Netherlands (based on van Duijnen, 2020)

Crop type	median nitrate concentration ($mg.l^{-1}$)			
Soil type:	Sand	Löss	Clay	Peat
Grass	7	18	7	6
Maize	62-68	77-99	37	8
Potato	42-45	75		
Leaf & stem vegetables	113			
Barley	48	54		
Temporary grassland	38	17		
Sugar beet	49	52		
Wheat	51	61		

In addition to the differences between crops, it is also essential to look at nitrate leaching at the level of the entire crop rotation. Accurately attributing N leaching to preceding-crops-current crop combinations is often challenging (Beillouin et al., 2021). Based on model simulations, Rashid et al. (2022) found that rotations with a higher proportion of spring crops are more prone to leaching than rotations having a higher proportion of winter cereals and semi-perennial grass-clover leys. Based on measurements in a small catchment area in northern France, Beaudoin et al. (2005) observed that a sugar beet-wheat rotation gave the lowest nitrate concentration at the end of the winter and a pea-wheat rotation the highest one. The higher was the proportion of spring peas in the crop succession, the greater was the nitrate concentration. De Notaris et al. (2018) tested crop rotations with different annual crops (barley, hemp, pea, wheat, potatoes) and catch crops in Denmark and found that spring wheat and potatoes were the two crops with highest N leaching. Beillouin et al. (2021) simulated 20-year series of N leaching for various cropping sequences and found an impact of the preceding crop on N leaching from the current crop.

Crop rotation and the introduction of alternative and/or (semi-)perennial crops in the rotation as well as the conversion to permanent crops can play an important role in minimizing the risk of nitrate leaching by improving soil N availability, reducing the amount of N fertilizer applied and optimizing nutrient absorption throughout the soil profile as well as throughout the year. According to Beillouin et al. (2021), diversified cropping sequences can reduce N leaching by up to 40%. The effects of land use and crop rotation is illustrated by the literature review given in Table 18, of annual N leaching figures for different crops and crop rotations.

Table 18 Nitrogen leaching in different crop rotations

Reference	**Area**	**Method**	**Crops/Rotations**	**Average N leaching per rotation ($kg\ N.ha^{-1}.year^{-1}$)**
Beaudoin et al. (2005)	northern France	field measurements; modelling	rapeseed - winter wheat	32 kg $N.ha^{-1}$
			winter wheat – rapeseed	17 kg $N.ha^{-1}$
			winter wheat – winter barley	31 kg $N.ha^{-1}$
			pea – winter wheat	42 kg $N.ha^{-1}$
			sugar beet – winter wheat	11 kg $N.ha^{-1}$
			winter barley – cover crop	27 kg $N.ha^{-1}$
De Notaris et al. (2018)	Denmark	field experiment; field measurements; modelling	barley-hemp-pea-barley-wheat-potatoes without catch crop	51-63 kg $N.ha^{-1}$
			barley-hemp-pea-barley-wheat-potatoes with catch crop	31-36-39 kg $N.ha^{-1}$
			barley-green manure crop-barley-wheat without catch crop	54 kg $N.ha^{-1}$
			barley-green manure crop-barley-wheat without catch crop	24-33 kg $N.ha^{-1}$
Ferchaud & Mary (2016)	northern France	field experiment; modelling	sorghum-triticale	1.8-5 kg $N.ha^{-1}$
			switchgrass	0.5 kg $N.ha^{-1}$
			fescue-alfalfa	0.5-2.3 kg $N.ha^{-1}$
			miscanthus	2 kg $N.ha^{-1}$
Hussain et al. (2019)	Michigan, US	field measurements; modelling	continuous maize (corn)	34 kg $N.ha^{-1}$
			switchgrass	9.2 kg $N.ha^{-1}$
			miscanthus	7.6 kg $N.ha^{-1}$
			grasses	4.4 kg $N.ha^{-1}$
			poplar	7.2 kg $N.ha^{-1}$
			restored prairie	2.4 kg $N.ha^{-1}$
Jiang et al. (2022)	Canada	field experiment; modelling	buckwheat-buckwheat-potatoe	135 kg $N.ha^{-1}$
			buckwheat-red clover-potato	193 kg $N.ha^{-1}$
			Buckwheat-Timothy-potatoe	86 kg $N.ha^{-1}$
Kanwar et al. (2005)	Iowa, US	field experiment	maize-soybean-oats strip intercropping	12.6 kg $N.ha^{-1}$
			alfalfa(3yr)-maize-soybean-oats	11.8 kg $N.ha^{-1}$
			soybean-maize	13.7 kg $N.ha^{-1}$
			maize-soybean	13.3 kg $N.ha^{-1}$

Reference	Area	Method	Crops/Rotations	Average N leaching per rotation (kg N.ha^{-1}.year^{-1})
Kunrath et al. (2015)	France	field experiment	arable rotation: maize-wheat-barley grassland arable + 3 yr grass arable + 6 yr grass	37 kg N.ha^{-1} 70 kg N.ha^{-1} 18 kg N.ha^{-1} 8 kg N.ha^{-1}
Masoni et al. (2015)	Italy	field experiment	barley field bean red clover sulla alfalfa italian ryegrass, preceded by: alfalfa (3yr) sulla(2yr) barley(1yr) red clover(2yr) field bean(1yr)	37 kg N.ha^{-1} 47 kg N.ha^{-1} 26 kg N.ha^{-1} 26 kg N.ha^{-1} 18 kg N.ha^{-1} 7 kg N.ha^{-1} 9 kg N.ha^{-1} 22 kg N.ha^{-1} 9 kg N.ha^{-1} 32 kg N.ha^{-1}
Olesen & Askegaard (2004)	Denmark	field experiment	barley+ley-grassclover-wheat-lupin barley+ley-grassclover-wheat-lupin+grassclover barley+ley-grassclover-wheat-peas/barley barley+ley-grassclover-wheat+grass-peas/barley+grassclover oat-wheat-cereal-peas/barley oat+clover- wheat+clover-cereal+clover-peas/barley+grassclover	102 kg N.ha^{-1} 68 kg N.ha^{-1} 32-49-95 kg N.ha^{-1} 23-35-63 kg N.ha^{-1} 25-34 kg N.ha^{-1} 24-36 kg N.ha^{-1}
Pugesgaard et al. (2015)	Denmark	Field experiment; modelling	winter wheat grass-clover young willow (1-3 yrs after planting) old willow	36-68 kg N.ha^{-1} 3-6 kg N.ha^{-1} 18, 3, 0.3 kg N.ha^{-1} 6-27 kg N.ha^{-1}
Randall & Mulla (2001)	US	literature review	continuous maize (corn) maize-soybean, soybean-maize alfalfa grass-alfalfa mix	54 kg N.ha^{-1} 51 kg N.ha^{-1} 50 kg N.ha^{-1} 2 kg N.ha^{-1} 1 kg N.ha^{-1}
Simeonova et al. (2024)	Bulgaria	field experiment	maize wheat	5.6-28.5 kg N.ha^{-1} 1.2-6.3 kg N.ha^{-1}
Smith et al. (2013)	Illinois (US)	field experiment	maize (corn) soybean miscanthus switchgrass prairie	7-21 kg N.ha^{-1} 10 kg N.ha^{-1} 0.8-22 kg N.ha^{-1} 0.5-8 kg N.ha^{-1} 0.4-9 kg N.ha^{-1}

4.1.2. Assessment on the effectiveness and efficiency in terms of preventing and reducing nutrient losses

In the following assessment, the effects of some possible land use and cropping system measures on nitrate leaching are reviewed more in detail, based on literature.

4.1.2.1. Breaking maize monoculture

Growing silage maize on the same field for years has serious disadvantages: yields will decrease, in some cases with up to 20%. Soil organic matter also decreases and nitrate leaching increases.

At the same time, results from the Netherlands show that breaking maize monoculture with grass can result in more nitrate leaching than the continuous and separate cropping of each of these crops, due to the large amount of mineral N released after the ploughing up of the grass (Kayser et al. 2008; Hooijboer et al., 2017). However, trials on experimental farms have shown that nitrate leaching in maize-grass rotations can be reduced with adapted cropping techniques, such as zero N fertilization in the first year of maize after grass and growing of catch crops after maize (Hooijboer et al., 2017).

Based on field experiments in the US (Pasley et al., 2021), rotating maize with soybean allows for a greater environmental buffer than continuous maize with regard to the impact of overfertilization on nitrate leaching. The cropping system plays a significant role in determining the degree to which the fertilizer N rate affects nitrate leaching as well as the rate above which leaching increases significantly.

Including legumes such as soybean or alfalfa, in maize-based rotation systems such crops decrease nitrate-N concentrations in subsurface drainage discharge (Koropeckyj-Cox et al., 2021) and reduces nitrate leaching to aquifers (Singh et al., 2023). From the analysis of field soil samples in Nebraska, Singh et al. (2023) observed that including alfalfa in maize crop rotations effectively reduces the nitrate leaching potential thanks to an improved N uptake (less nitrate-N stock in the soil) and less deep soil water percolation than continuous maize.

4.1.2.2. Introducing break or rest crops in crop rotations

In the context of reducing the risk of nitrate leaching, “break crops” are defined as crops grown to diversify the cropping sequence and reduce N leaching at the rotation due to their contribution to increasing nutrient use efficiency and crop N recovery of the following cereal crop (Beillouin et al., 2021). In the Netherlands, “rest crops” are defined as crops that are less susceptible to nitrate leaching and at the same time have a positive effect on soil quality. Examples are cereals, grass seed, rapeseed, alfalfa and flax (Rijksdienst voor Ondernemend Nederland). In the Netherlands, it is mandatory since 2023 to grow “rest crops” once every 4 years, on all arable fields on sandy and loess soils.

4.1.2.3. Introducing (semi-) perennial crops

Annual crops leak substantially greater amounts of nitrate compared with perennial crops, that have an extended growing period and greater root activity (water and nutrient uptake) and where cycling of N is optimized (Randall et al., 2001; Masoni et al., 2015; Pugesgaard et al., 2015; Ferchaud & Mary, 2016). Deep-rooting crops such as alfalfa are very effective at accessing deep-leached N (Kelner et al., 1997). Therefore, the introduction of perennial crops can contribute substantially to the improvement of the water quality. However, difficulties getting a satisfactory economic return with many perennials may be an obstacle (Randall et al., 2001). Moreover, perennial crops may decrease groundwater recharge in deep soils or dry climates (Ferchaud & Mary, 2016).

In a field experiment in Denmark, significantly lower nitrate leaching compared to winter wheat in grass- clover and willow was attributed to a combination of a longer growing season enabling both willow and grass-clover to take up N during autumn and early spring, higher levels of evapotranspiration and, hence, lower level of drainage from the root zone and minimal soil tillage, resulting in less mineralization (Pugesgaard et al., 2015).

Compared to annual crop production, perennial grasses reduce N losses by exploiting the entire growing season for several years without re-establishment. In field experiments in Denmark, they reduced nitrate leaching by 70–80% compared to the traditional annual cropping systems (Manevski et al., 2018).

The introduction of perennial mowed grassland sequences into arable crop rotations in a field experiment in France caused a marked reduction in the nitrate levels in groundwater and the greater the proportion of grassland within the rotation the more marked was the reduction in nitrate concentration (Kunrath et al., 2015).

In a field experiment in Minnesota, potential nitrate leaching was estimated to be 77–96% lower under intermediate wheatgrass (perennial grass) than under the annual maize-soybean rotation. The wheatgrass captured a greater proportion of the soil N, which was likely facilitated by its greater root biomass (5.4 times higher than that under the annual crop rotation) (Reilly et al., 2022).

On a yearly basis, nitrate leaching from legumes increases with the decrease in crop cycle length (Masoni et al., 2015). The use of perennial legume crops also reduces nitrate leaching in subsequent grass crops, due to a better synchronisation of their N demand with the N supplied from residue mineralisation. E.g. in a field experiment in Italy, nitrate leaching from Italian ryegrass was higher when cultivated after annual crops than after perennials, attributed to a longer intercrop period following the annual crops (Masoni et al., 2015).

However, perennial crops may leach more nitrate during their establishment phase, in the first few years after planting (Hussain et al., 2019; Ferchaud & Mary, 2016). In their field experiment in Denmark, Pugesgaard et al. (2015) found that nitrate leaching from young willow was higher in the first winter after planting than in the two subsequent winters. The reason is most likely that the willow had not established an extensive root system until after the second growth year after which nutrient uptake is very effective.

4.1.2.4. Permanent crops: orchards

Orchards, like any agricultural system, require nitrogen fertilizers to support plant growth and fruit production. In orchards high value crops are produced and they are often managed in an intensive manner on highly fertile soils. Like all horticultural crops orchards are susceptible to nitrate leaching. A study of the nitrogen dynamics in orchards in Europe has estimated nitrate leaching between 4 and 117 kg N/ha per year (Table 19).

Orchards are often irrigated so an increase of the irrigation efficiency will decrease nitrate leaching out of the root zone. Hydrological studies near in the Jucar river basin near Valencia (Pool et al., 2022, Pérez-Martín and Benedito-Castillo, 2023) estimated for citrus that drip irrigation would reduce the nitrogen leaching fraction from 86% to 18% compared to furrow or flood irrigation. The effect of an improved irrigation management to reduce nitrate leaching is discussed further in the section on *“Prevention of leaching and run-off in irrigation systems”* below.

Table 19 Nitrogen leaching calculated in fruit orchards in Europe.

Author and study period	Crop and location	Fertilization dose	Irrigation	Nitrate leaching per year
Fernández-Escobar et al., 2012 2001-2007	Olive, Cabra, Province of Córdoba, Spain	80-200 kg N/ha	Drip irrigation according to deficit irrigation scheme	14-117 kg N/ha
Lidón et al., 2013 1991-1993	Citrus, Valencia, Spain	150-300 kg N/ha	Yes but no specification of irrigation scheme	40-76 kg N/ha
Nesme et al., 2009 2002-2003	Apple, Nîmes-Montpellier, France	50-150 kg N/ha	Yes, optimal irrigation schedule filling to field capacity after period of water stress.	50-100 kg N/ha
Verellen et al., 2022 2020	Pear, Belgiu m	65 kg N/ha	Yes, optimal irrigation scheme according to soil water balance	58-75 kg N/ha
Demestihas et al., 2018 2014-2015	Apple, France	18-36 kg N/ha	Yes but no specification of irrigation scheme	4-38 kg N/ha

4.1.3. Environmental co-benefits

Perennial crops can lead to a reduction of N2O emissions (Smith et al., 2013).

Crop diversification contributes to increased biodiversity, improving soil quality, built-up of soil organic matter (SOM).

Perennial crops results in a build-up of soil organic matter and reduces erosion risk.

4.1.4. Minimum/threshold or optimum level of implementation

Not applicable to this measure.

4.2. Vegetation cover in rainy periods, winter (cover crops)

4.2.1. Background

Cover crops are planted/sown to cover the soil during the period between the harvest of a main crop and the sowing of the next main crop (mostly autumn and winter). During these periods, cover crops can "catch" the residual nitrate in the soil after the harvest of the main crop and retain it during winter, releasing it again in spring. In this way, cover crops reduce nitrate leaching and increase the transfer of N to the subsequent crop. Such cover crops are also sometimes referred to as "catch crops". Their effectiveness depends on soil, climate as well as land and crop management. Globally, the greatest cover crop efficacy is observed in coarse-textured soils, vegetable cropping and tilled soils (Nouri et al., 2022).

4.2.1.1. Cover crop functions

In the literature and in agricultural practice, different names are used for non- (or less) productive crops that are sown after the main crop (cash crop), depending on the intended crop function. For example, a “cover crop” can be defined as a crop whose function is to keep the soil covered during autumn or winter to reduce the risk of erosion and compaction. The function of a “catch crop” is to capture and retain residual nutrients (N) in the soil in the autumn with the aim of preventing nutrient losses during autumn and winter. A “green manure crop” is a crop that, after dying and incorporation in the soil, provides nutrients for the next main crop. Finally, an “intermediate” crop is a crop that is intended to achieve an additional yield, such as grass after maize.

In practice, there is overlap between these functions. Non-leguminous cover crops also function as catch and green manure crops. Leguminous cover crops mainly function as a green manure and only to a lesser extent as a catch crop. They do absorb some N from the soil, but they do this rather inefficiently. Intermediate crops are (usually) also in the field in autumn-winter, but are harvested (e.g. grass sown after maize, a cutting of which is harvested in the spring). In addition to their yield function, they also function as a catch and cover crop, but not as a green manure. Winter cash crops, such as winter grains, sown in time, can also function as a catch crop (but mainly only after winter) but do not function as a cover or green manure crop.

4.2.1.2. Mechanisms for prevention and reduction of nutrient losses

When a cover crop is sown, the nitrate N content in the soil initially increases slightly, mainly due to a temporary increase in the mineralization of soil organic matter after the seedbed preparation. Once the root system has developed, the cover crop can absorb nitrate from the soil, causing the nitrate content to decrease and remain low during the winter months. In the spring, after the cover crop has been terminated, this absorbed nitrogen will mineralize again and be released in the soil.

A catch crop can also reduce nitrate leaching through a reduction in drainage due to increased transpiration and thus water depletion in the soil (Thapa et al., 2018). On the other hand, Nouri et al. (2022) found that the impact of cover crops on water drainage was non-significant which would imply that nitrate leaching control by cover crops is less likely to be exerted through a reduction in water drainage.

A third mechanism of the reduction of nitrate leaching by cover crops could lie in the microbial immobilization of nitrate due to carbon inputs from cover crop roots into the soil (Thapa et al., 2018).

The effects of cover crops on P losses are less clear. Norberg & Aronsson (2018) conducted a field experiment with oilseed radish in southern Sweden and found no impact on P leaching. A field study review of Aronsson et al. (2016) in southern Scandinavia and Finland, indicated that cover crops do not substantially reduce total P losses by runoff and leaching.

4.2.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses

4.2.2.1. Reduction of nitrate leaching – figures

Based on a global meta-analysis, Thapa et al. (2018) concluded that non-leguminous cover crops reduce nitrate leaching by 56% on average. The potential of the cover crops to reduce nitrate leaching was partially dependent on their sowing date, aboveground biomass and weather conditions (relative precipitation). Early sowing in autumn and higher aboveground biomass in spring are both linked to a longer growth period of the cover crop and therefore a higher nitrogen uptake. The reducing effect on nitrate leaching was higher in relatively drier years (compared to the long-term average).

Nouri et al. (2022) conducted a similar meta-analysis and found that non-leguminous cover crops reduce nitrate leaching by an average of 69%. Overall, the reduction was greatest with cruciferous (75%) and grassy (52%) cover crops.

In Scandinavian regions, the possibilities of cover crops sown after harvest of the preceding cash crop to reduce nitrate leaching is limited by the poor light and temperature conditions during autumn and winter. For this reason, cover crops are often undersown in the preceding main crop (see below) and the use of cover crops sown at harvest is restricted to southern Scandinavia. Aronsson et al. (2016) summarized the results of field leaching studies at 11 different sites and found an average relative reduction in N leaching of 43%, but it ranged between an increase of 62% (after red clover) to a reduction of 85% to 89% with perennial ryegrass on sandy soils in Denmark. They concluded that, under the climatic conditions of southern Scandinavia and Finland, undersown ryegrass cover crops are robust with respect to growth and reduction of N leaching, while crucifers sown after the main crop are less reliable due to the short time available for growth under cool climate conditions. Valkama et al. (2015) conducted a meta-analysis of studies conducted in Denmark, Sweden, Finland and Norway and found that non-leguminous cover crops, mainly ryegrass species, reduced N leaching loss by 50% on average, whilst leguminous cover crops did not diminish the risk for N leaching.

4.2.2.2. N-uptake and influencing factors

The reduction of nitrate leaching by cover crops is mostly achieved by their N uptake. The N uptake capacity of a crop generally is promoted by the following properties:

- a high leaf growth rate (leaf expansion rate),
- a high radiation conversion efficiency (radiation use efficiency),
- a fast and sufficiently deep root development.

The extent to which these properties are expressed is determined by the weather conditions. In addition, the length of the growing season (sowing time), the soil properties (texture, structure and nutrient availability), the field history, the type of tillage before sowing and the fertilization also have an important impact on the root development and therefore on the efficiency of a cover crop. E.g. Selzer & Schubert (2021) determined the N uptake capacity of different types of cover crops in semi-controlled, non- restrictive growing conditions and found average N uptakes for yellow mustard, Phacelia and ryegrass of 183, 221 and 85 kg N/ha, respectively. Under practical conditions, most studies reported nitrogen uptake rates ranging between 10 and more than 100 kg N/ha.

In a review on the use of catch crops in temperate climates, Thorup-Kristensen et al. (2003) reported N uptakes for non-leguminous catch crops of 10 to 200 kg N/ha and in rare cases even up to 300 kg N/ha. They attributed the variation mainly to weather conditions and N availability.

Cover crop type

Cover crop type is an important factor determining the effect on the reduction of nitrate leaching. An important distinction has to be made between non-legume and legume cover crops.

Non-legume cover crops can be divided into 2 large groups based on their crop growth characteristics and properties:

- Broadleaf cover crops have a rapid and mainly above-ground growth with significant biomass production and N uptake in late summer and autumn. Typical species are mustard, radish, kale and Phacelia. After winter, these catch crops are incorporated in the soil and, because of their relatively low C:N ratio, they decompose quickly, resulting in an early release of N. Typically, they have little root biomass compared to grasses and leguminous plants. Broadleaf catch crops are generally sensitive to frost, which means that, depending on the climate, they usually freeze in winter and then stop absorbing N.

- Grassy cover crops show a high N uptake, which, however, only starts slowly in the autumn. However, they are not or only moderately sensitive to frost, which means they exhibit stronger growth and N uptake, especially during winter and spring. Typical species are Italian or perennial ryegrass, rye or oats. In contrast to leafy catch crops, grasses have a pronounced root development, which means they make an important contribution to the organic matter in the soil and prevent soil compaction. The higher C:N ratio of these crops ensures a slow decomposition after their termination. As a result, the absorbed N is only released later in the growing season.

When planted early (August-September) and/or no winterkill of broadleaf species takes place, both grasses and non-legume broadleaf species are equally effective in reducing nitrate leaching. When sowing late September-October (after maize, potatoes, sugar beets, etc.), grasses outperform broadleaf species (Thapa et al., 2018). Overall, cover crops from Brassicaceae and Poaceae families show the greatest effect with 75% and 52% reduction in nitrate leaching, respectively (Nouri et al., 2022).

In agricultural practice, the choice of a cover crop depends on various parameters such as the harvest date of the main crop and the weather and soil conditions. In a field trial in northern Germany, Böldt et al. (2021) recorded nitrogen uptakes of 33 to 136 kg N/ha by cover crops. They also measured nitrate leaching and found that it was greater in frost-sensitive winter-killed cover crops, due to the premature mineralization of the crop residues.

Legume cover crops provide a wide range of ecosystem services, including N supply as well as reduction of nitrate leaching (De Notaris et al., 2020). However, they retain less N than non-legume species and therefore are less efficient in reducing nitrate leaching because they meet some of their N demand through atmospheric N fixation (Thapa et al., 2018; Shackelford et al., 2019; White et al., 2017). Numerous studies have also shown that their effectiveness in depletion of soil N is not reliable (Aronsson et al., 2016). Also, high N_2O emissions following the incorporation of monoculture legume cover crops have been reported (reviewed by Jensen et al., 2011).

Non-legume-legume cover crop mixtures can reduce nitrate leaching as effectively as non-legumes, and significantly more than legumes alone (Thapa et al., 2018). Moreover, mixtures of legumes and non- legumes can maximize the provision of ecosystem services, by reducing nitrate leaching and increasing N availability to the following crop, replacing synthetic fertilizer inputs with biologically supplied nitrogen (Kaye et al., 2019) and providing more flexibility to the system and a better ability to adapt to the specific environmental conditions (De Notaris et al., 2020, based on several other sources).

Cover crop species selection and mixture design can substantially mitigate trade-offs between N retention and N supply to cash crops (see also below, 2.2.3.2, Trade-offs), providing a powerful tool for managing N in temperate cropping systems (Kaye et al., 2019). Based on multiple regression analyses with data from on-farm experiments, White et al. (2017) found that potential nitrate leaching below 30 cm declined as the presence of non-legume species increased in a cover crop.

The C/N ratio of cover crop biomass is positively related to the N retention by cover crops and is negatively related to the N supply to cash crops (Finney et al., 2016). White et al. (2017) used the C:N-ratio as an indicator representing the relative contribution of legume and non-legume (grassy) species to the cover crop biomass. In order to visualise and manage the trade-off between nitrogen supply and retention with cover crop mixtures, they developed a graphical analysis tool to predict relative maize yield and potential nitrate leaching across categories of cover crop type in different environmental conditions in the US (Figure 20 and Figure 21).

Figure 20 The framework of regression models driving the graphical analysis tool (White et al., 2017). Variables that directly predict relative maize yield or potential nitrate leaching are indicated by solid arrows. Variables that are included in the auxiliary regression models to predict fall C:N, spring C:N and fall non-legume biomass N content in legume cover crops are indicated by dashed arrows. Variables that are user inputs to the graphical analysis tool are bounded by the gray box on the left. The graphical output of the tool (gray box on the right) then predicts relative maize yield and potential nitrate leaching across categories of cover crop type and gradients of fractional non-legume seeding rate and spring biomass N content. Because fall C:N, spring C:N and fall non-legume biomass in legume cover crops are predicted by the auxiliary regression models, they are not needed as user inputs to the tool.

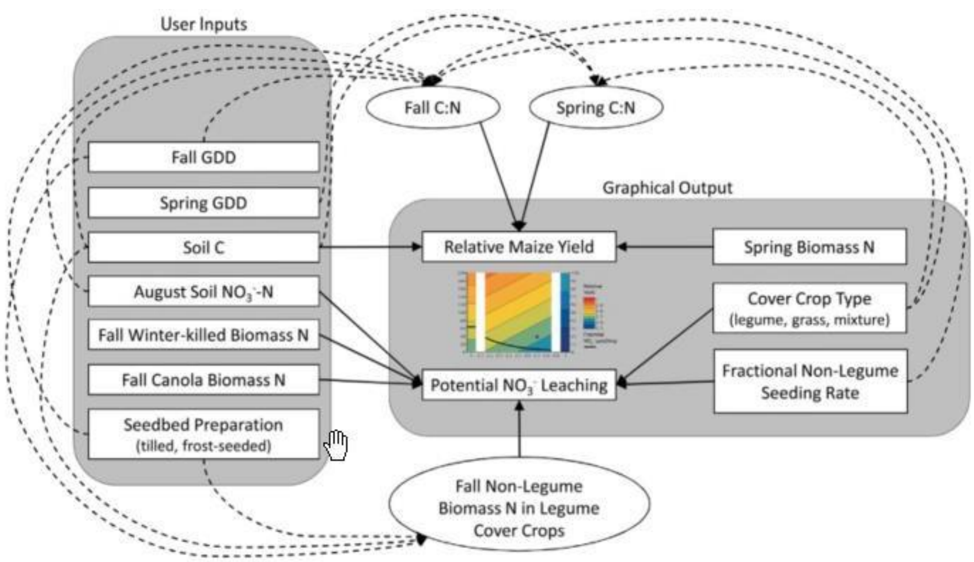
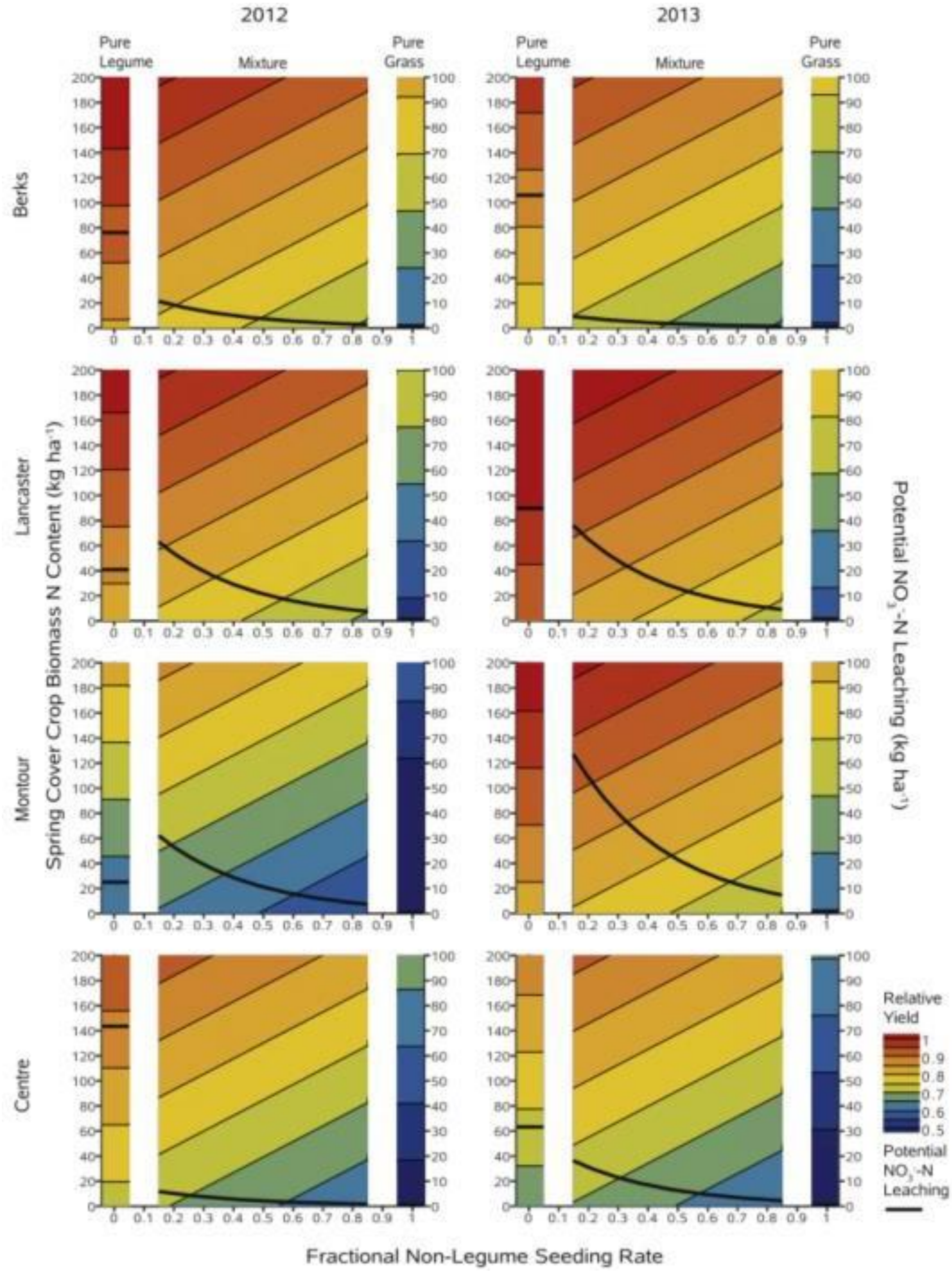


Figure 21 Outputs from the graphical analysis tool predicting potential NO_3^--leaching (black response curves; kg.ha^{-1}) and relative maize yield (colored contour plots) in eight site-years based on site-specific environmental conditions and cover crop composition for the 4-species mixture (White et al., 2017). Location on the contour plot for relative maize yield is determined by the fractional non-legume seeding rate (x-axis) and the spring cover crop biomass N content (left y-axis), where the value of each contour interval is defined by the color scale in the legend. Location on the potential NO_3^--leaching response curve is determined by the fractional non-legume seeding rate, with the predicted value read from the second (right-side) y-axis. Each output is divided into three columns on the x-axis, corresponding to the regression models for pure legumes, mixtures, and pure grasses.



Sowing date

The sowing date is an important parameter that determines the total N uptake of a cover crop. With later sowing dates, not only does the total growing season become shorter, but also day length, light intensity and temperature decrease, while the precipitation surplus generally increases. All this ensures that the (total) biomass production and N uptake of a cover crop decrease on average with later sowing dates. Constantin et al. (2015) performed large-scale simulations to determine the optimal emergence and maturity dates of catch crops to reduce nitrate leaching and concluded that cover crops can be very efficient in reducing nitrate leaching if their management, in particular the sowing and termination dates are adjusted to local weather conditions.

Komainda et al. (2018) conducted two field trials in northern Germany with 4 different cover crops, sown between two maize crops on 4 different sowing dates (from September 10 to October 15). Total N uptake by the cover crops varied between 7 and 56 kg N/ha and the latest sowing date yielded an average N uptake reduction of 67% compared to the earliest sowing date. Earlier sowing also resulted in a higher C/N ratio because of the more developed crop during incorporation. Thapa et al. (2018) concluded that a delay in planting till November meant that the cover crop did not induce a significant reduction of nitrate leaching anymore.

Vos (1992) collected data from several empirical studies on N uptake by cover crops. If at least 100 kg/ha of mineral N was initially present, the above-ground N yield at end of November - beginning of December could be approximated using the following equation:

$$\text{N-uptake} = 522 - 1{,}8 * \text{sowing date}$$

Vos & van der Putten (1997) also demonstrated a linear relationship between the maximum above-ground N uptake and the sowing date for the fertilized catch crops from their study (1991-1996):

$$\text{N-uptake} = 960 - 3.4 * \text{sowing date}$$

(the sowing date is expressed as a day of the calendar year)

However, there is a large variation in the effect of the sowing date, which can be attributed to annual meteorological differences, to the differences between the types of cover crops and to the presence or absence of additional fertilization. Cover crops usually are sown after the harvest of the main crop, but especially in northern regions (at higher latitudes) where light and temperatures during winter are limiting, their growing period can be extended by establishing them already in the growing main crop. This can be done by undersowing in spring or by preharvest broadcasting of seeds later in the growing season (Thomsen & Hansen, 2014).

Constantin et al. (2015) simulated the emergence of catch crops for different regions in France, depending on seedbed conditions with regard to temperature, moisture content, soil structure and crust formation. The results showed that in (Mediterranean) areas with dry summers, early sowing (mid-July) often resulted in poor germination and emergence of the catch crop, while later sowing (mid-September) was usually more successful. They concluded that sowing date is the management factor with the greatest impact and that it must be carefully adjusted to weather conditions to maximize nitrate uptake.

Weather conditions

The effects of sowing date cannot be completely separated from the effect of climate and weather conditions. In general, there is a direct link between the dry matter production of crops and the captured solar radiation (Vos & van der Putten, 1997). Komainda et al. (2016) found that the temperature (sum) was the determining factor for dry matter production and nitrogen uptake, while solar radiation had no meaningful influence on the underground plant parts. Although dry matter production is directly related to solar radiation as an energy source, leaf development in autumn is mainly controlled by temperature (Bartholomew & Williams, 2005).

Komainda et al. (2016) developed exponential functions based on temperature sums to predict above- and below-ground biomass and N uptake for rye and Italian ryegrass. As a base temperature for the calculation of temperature sums, the best model fit was obtained for a base temperature of 5°C. This way, they calculated for example that for rye a temperature sum of 278 °C is required to achieve a total N uptake of 20 kg N/ha. In northern German conditions, this corresponds to a sowing date at the latest in the middle part of September. In an earlier study by Schröder et al. (1996), a linear relationship was found between above-ground N uptake and temperature sums (with a base temperature of 5°C) for Italian ryegrass.

In addition to solar radiation and temperature, precipitation and drought can also play an important role in the growth and nitrogen uptake of catch crops. Thapa et al. (2018) concluded that the effect of non-leguminous cover crops in reducing nitrate leaching was greater during years of low precipitation (<90% of the long-term normal). Emergence rate is also crucial for cover crop establishment, especially in dry areas and is mostly influenced by water availability and therefore by climatic conditions in the period around the sowing date. Tribouillois et al. (2016) calculated optimal and baseline values for temperature and soil moisture content for the germination of a range of catch crops (basic temperatures for germination ranged from 0.0 to 8.1°C, optimal temperatures from 20.2 to 37.2° C, maximum temperatures from 27.7 to 43.4°C and basic moisture stress values ranged from 0.1 to 2.6 MPa). Based on field trial data and simulations with the STICS model, Tribouillois et al. (2018) proposed a simple statistical model to predict the emergence of catch crops, in which both soil moisture content and precipitation (and irrigation) after sowing were significant predictive parameters:

$$\text{Emergence duration} = 6.9 - 0.29 \times SM + 0.44 \times DWR + 0.03 \times DD_e$$

where SM is soil moisture (%) in the seedbed layer at sowing, DWR is the number of consecutive days without significant water input (daily rainfall or irrigation < 3 mm) after sowing and "DD_e" is the number of degree days required for each species to emerge without stress. The values of DDe for each species are given in Appendix. According to the GLM, DWR, SM and DD_e were significant ($P < 0.001$, $=0.001$ and $=0.05$, respectively) and explained 52%, 6% and 4% of the variability, respectively.

Soil type

Cover cropping on Ultisols, Histosols, and Inceptisols resulted in the greatest reduction in nitrate leaching (77%, 78%, and 77%, respectively). Greater efficacy of cover crops at reducing nitrate leaching was evident with increasing soil sand content (Nouri et al., 2022).

4.2.2.3. Effects on nutrient losses after termination of cover crops

Once terminated, cover crops can also increase the risk of nitrate leaching in spring, if the N released during decomposition is not recaptured by the subsequent cash crop. In other words, leaching during the subsequent cash crop period after termination of cover crops is associated with the time lag between the N released from cover crop residues and the N demand of the cash crop. Cover crops are typically terminated 2 to 6 weeks before the subsequent cash crop is planted, inducing a relatively long period without living plants absorbing soil N. Therefore, strategies to reduce nitrate leaching should not only focus on growing cover crops but also on the efficient use of the N captured by the cover crops and released after their termination.

The risk of N leaching after cover crop termination depends on the C:N ratio of the cover crop residues, the soil type, and the N uptake by the subsequent cash crop. Typically, legume and other broadleaf cover crops have a lower C:N ratio (see above), with a quick decomposition and an early release of N, whilst grassy cover crops have a C:N ratio that increases with plant maturity (Greenwood et al., 1990), ensuring a slow decomposition after their termination and an N release only later in the growing season. In this perspective, legume-non-legume mixtures have the advantage of having an intermediate C:N-ratio, ensuring a more balanced N mineralization and immobilization turnover, an improved coordination between the N release from cover crop residues and uptake by subsequent cash crops. As a result, they can reduce nitrate leaching in spring and at the same time partially meet the N fertilizer requirements of the subsequent cash crop.

4.2.3. Environmental co-benefits and drawbacks

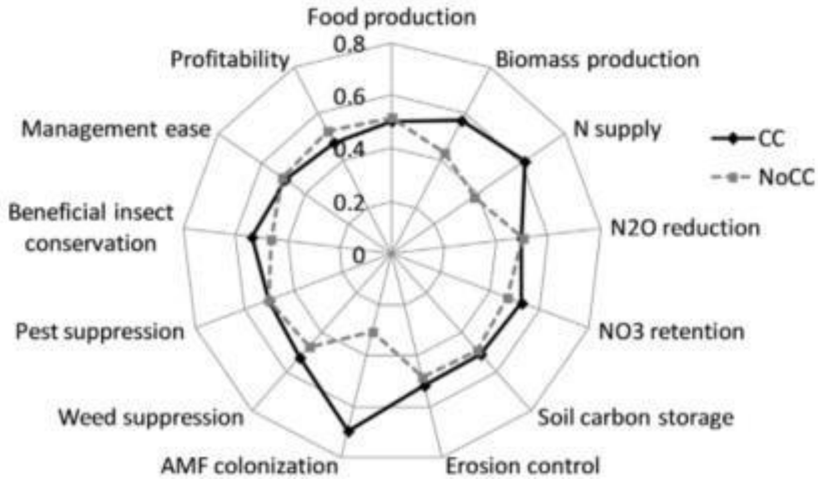
4.2.3.1. Co-benefits

Besides reduction in nitrate leaching, cover crops can enhance numerous other agroecosystem services, including

- Weed suppression (Shackelford et al., 2019; Thapa et al., 2018; Blanco-Canqui et al., 2021) ;
- Increase in soil organic matter and soil microbial biomass (De Notaris et al., 2020 ; Shackelford et al., 2019; Thapa et al., 2018; Blanco-Canqui et al., 2021; McClelland et al., 2021);
- Reduction of soil erosion and soil compaction (Thapa et al., 2018; Rivière et al., 2011; Blanco-Canqui et al., 2021; Lopez-Vincente et al., 2020). Cover crops are used to minimize soil erosion caused by intensive tillage, overuse of herbicides and loss of soil organic matter (Lopez-Vincente et al., 2020; Quintarelli et al., 2022). García-Díaz et al. (2022) observed that the use of cover crops could reduce soil losses by 3.8 to 0.7 Mg/ha in a vineyard. Novara et al. showed that using cover crops reduces by 27% annual water runoff and can be used as an agronomic strategy for improving water use efficiency (see also Task 4: use of cover crops and intercropping to reduce nutrient losses on sloping ground);
- Increase of soil porosity and improved soil-water balance (Quintarelli et al., 2022);
- Increase of biodiversity (Thapa et al., 2018; Rivière et al., 2011; Blanco-Canqui et al., 2021; Quintarelli et al., 2022) ;
- Stimulation of beneficial micro-organisms (Quintarelli et al., 2022);
- In case of legume cover crops, fixation of atmospheric N and reduction of N fertilizer inputs.

Schipanski et al. (2014) integrated a suite of ecosystem services into a unified analytical framework and estimated that cover crops increase 8 of 11 ecosystem services (Figure 3).

Figure 3 Normalized values for 11 ecosystem services and two economic metrics averaged across the 3-year rotation of cropping systems with (CC) and without (NoCC) cover crops (Schipanski et al., 2014)



4.2.3.2. Drawbacks and trade-offs

In general, cover crops do not significantly reduce the yields of the following cash crops (Thapa et al., 2018). Leguminous cover crops can even increase crop yields (depending on fertilisation practices) (Shackelford et al., 2019). However, resource competition (for nutrients and water) with the cash crop can induce lower yields of the cash crop in the short-term (Shackelford et al., 2019; Alonso- Ayuso et al., 2017). This aspect is especially important in Mediterranean regions/climates. Applying management strategies including timely cover crop termination, selection of cover crop species and other strategies can contribute to cover crop success in water-limited environments (Blanco-Canqui et al., 2021; Shackelford et al., 2019). Effects of cover crops on water availability can also be masked by management practices such as irrigation (Shackelford et al., 2018).

In organic systems, cover crops can supply N to subsequent crops and retain nitrate against leaching (Thorup-Kristensen et al., 2012). However, individual cover crop species can often provide optimal levels of only one or the other of these services. Therefore, the use of cover crop mixtures (legume-non-legume) is often more favourable. For example, Engedal et al. (2023) concluded, based on the results of a field trial, that cover crop mixtures showed the highest total carbon input and generally a higher or similar mineral N depletion potential than the average of the pure stands, suggesting that cover crop mixtures offer a realistic means of overcoming trade-offs among ecosystem functions. White et al. (2017) suggested that the trade-off between N retention and N supply could be minimized by managing environmental conditions and cover crop composition. This could be achieved by maintaining low soil nitrate concentrations prior to cover crop planting, not including winter-killed legumes in the mixture, and using non-legume species that are the most efficient at N retention. Finally, trade-offs between ecosystem benefits, production costs, and management risks should be considered (Schipanski et al., 2014).

4.2.4. Minimum/threshold or optimum level of implementation

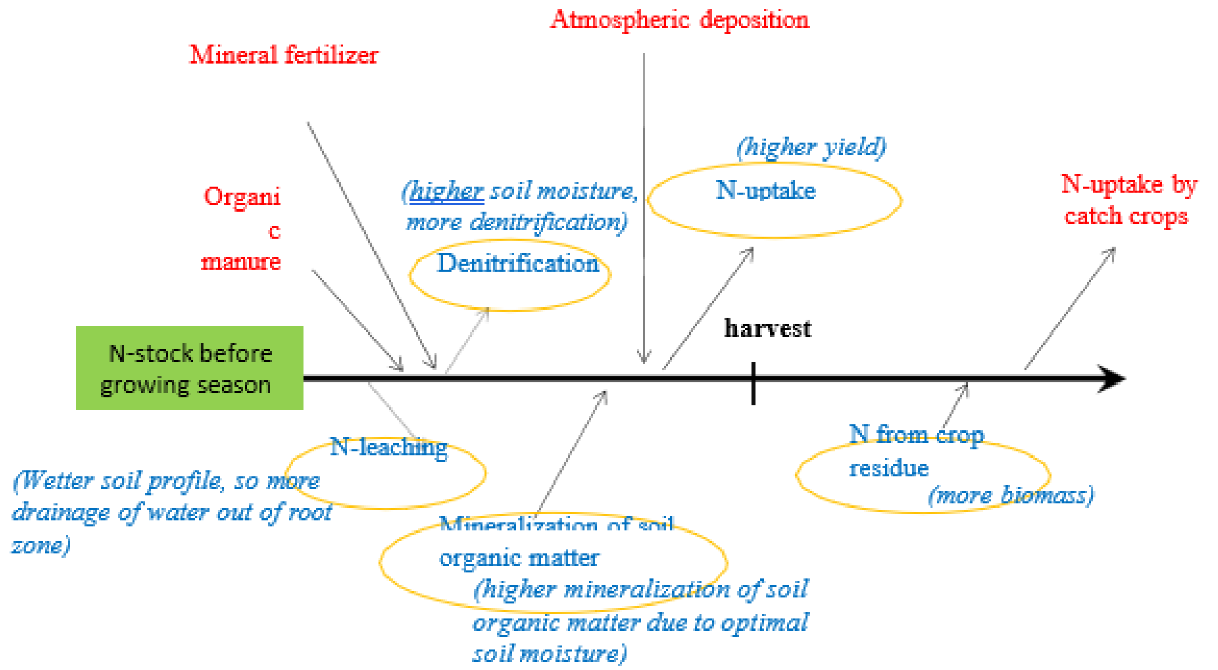
Not relevant for this measure.

4.3. Prevention of leaching and run-off in irrigation systems

4.3.1. Background

Irrigation affects the water status of the soil, which influences the nitrogen cycle (Figure 22). A higher water content in the soil will lead to more rapid drainage and thus possible nitrate leaching. A higher water content will stimulate microbiological activity and mineralisation of organic matter. If the increased water content leads to anaerobic conditions, also the denitrification rate will be higher. Generally, a higher water status in the soil during the growing season leads to more biomass production, higher yields and thus more nitrogen uptake by the crop but also more crop residues at harvest.

Figure 22 All factors that have an influence on the NO_3^--N concentration in agricultural soils; factors that are influenced by the soil water status are indicated in blue.



The effect of irrigation on water quality has been particularly examined for southern Europe (Table 20. Irrigation with water showing low NO3--concentration has been shown to dilute NO3-contaminated groundwater (Andrade and Stigter, 2009; Serra et al., 2021; Ventura et al., 2008). Next to more efficient fertilizer application a better management of irrigation has been suggested to lower NO3--N-losses to aquifers (González Vázquez et al., 2005; Lyra et al., 2022; Malik and Dechmi, 2020; Serra et al., 2021).

Table 20 Overview of studies on the relation between irrigation and NO_3^--N pollution in water bodies in Europe.

Author and study period	**Method used**	**Location and area**	**Relation between NO_3^--N pollution and irrigation**
Andrade and Stigter, 2009 (2000-2002)	Factorial correspondence analysis	Mondego River drainage basin - Central Portugal (51 km²)	Irrigation with water from the river dilutes the NO_3^- concentration in the groundwater.
González Vázquez et al., 2005 (1994-1996)	Multivariate statistical techniques (factor and cluster analysis)	Aquifer Nr 28 Unit Seville-Carmona, Province of Seville, Spain	NO_3^--N pollution is linked to cotton and potato production, due to the type of farming carried out, flood irrigation and excessive fertilizer application.
Isidoro et al., 2006 (1995-1998)	Calculation of the water budget	La Violada Irrigation District (VID) is located in the middle Ebro River basin (North-Eastern Spain)	Irrigated agriculture in was the main source of salt loading in La Violada Gully.
Lyra et al., 2022 (1991-2018)	Integrated Modeling System for agricultural coastal watersheds	Almyros basin located in the Thessaly region, central-eastern Greece (856 km^2)	The results of the study indicate that priority should be placed upon deficit irrigation under reduced fertilization.
Malik and Dechmi, 2020 (2014-2017)	Calculation with a Decision Support System for Agro-technology Transfer model	La Violada Irrigation District (VID) is located in the middle Ebro River basin (northeastern Spain)	Optimum N management could reduce NO_3^--N leaching below the root zone by 51%. These reductions could be improved further by 35%, under the combined N fertilization and irrigation optimum management.
Serra et al., 2021 (1999 and 2009)	Calculation of nitrogen balances	The methodology was applied on a national scale, considering mainland Portugal	Irrigation leads to a higher leaching risk but leads to aquifer recharge and thus also dilutes the NO_3^- concentration in the groundwater when the irrigation source has a low NO_3^- concentration.
Ventura et al., 2008 (2004-2006)	Calculation of nitrogen balance	Valle Volta, located in the eastern Po river valley (Italy)	Irrigation with open surface water that contains little NO_3^--N as a limited contribution to the influx of NO_3^--N in the area.

4.3.2. Effectiveness and efficiency in terms of preventing and reducing nutrient losses

Quemada et al. (2013) reviewed 44 irrigation experiments published between 1981 and 2012 and mainly executed in Europe and North America. The authors concluded that an improved water management has the highest potential to reduce nitrate leaching in irrigated areas. More specifically, a better irrigation scheduling according to crop needs was considered the most effective measure. Moreover, these measures did not result in yield decline. The implementation of deficit irrigation offers additional opportunities to reduce nitrate leaching, although this may be associated with a decrease in yield. Also important is that a reduction in irrigation must be accompanied by a reduction in fertilization dose. In this study, a better water management in irrigated fields was more successful than the improvement of fertilizer management or the use of cover crops to reduce nitrate leaching.

4.3.2.1. Irrigation scheduling

Irrigation scheduling is the practice of determining when and how much water to apply to crops or plants to meet their water needs efficiently. Irrigation scheduling has been closely linked to nitrate leaching and optimal N fertilization (Table 21).

Over-irrigation can cause water to percolate deep into the soil profile, carrying nitrates along with it. If more water is applied than the plants can use, the excess water can push nitrates past the root zone, increasing the risk of leaching. This has been described for irrigated maize in Portugal (Cameira et al., 2003) Spain (Daudén et al., 2004) and Romania (Marinov and Marinov, 2014). Also for open field processing tomato, nitrate leaching has been linked to excessive irrigation both in Italy (Rinaldi et al., 2007) and Spain (Vázquez et al., 2006).

On the other hand, insufficient irrigation may lead to water stress in plants, reducing their ability to take up nitrogen from the soil. This can result in unused nitrogen remaining in the soil, which is then more susceptible to leaching during subsequent rainfall or irrigation events. Especially in northern Europe studies emphasize the link between optimal agricultural production and nitrate leaching (ten Damme et al., 2022, Hack-Ten Broeke, 2001).

Table 21 List of studies in Europe linking irrigation scheduling to NO_3^--N leaching

Author and study period	**Crop(s) and growing conditions**	**Location and soil type**	**Relation between irrigation scheduling and NO_3^- - N leaching**
Cameira et al., 2003 (1996-1997)	maize for grain, silage maize grown in open fields	South of Portugal, Silty loam alluvial soil, with a shallow water table, sandy soil with a very low water retention capacity	Over-irrigation has been linked to NO_3^--N leaching. Irrigation depths and frequency should be limited to crop demand. In the crop water demand, capillary rise from a shallow groundwater table should be taken into account.
Daudén et al., 2004 (1997)	corn grow in in lysimeters	Zaragoza, Spain, Clay loam soil with an organic matter content of 18 g/kg	It is important to adapt N applications to crop extractions. Improvement of irrigation decreased the amount of irrigation leached.

Diez et al., 2000 (1991-1997)	rotation of corn, wheat and oat grown in open fields	30 km southeast of Madrid, Spain, sandy loam texture in the first 0.5 m with an increasingly sandy texture below. Water table is some 4 m below the soil surface.	Adapting irrigation schedules reduced nitrate discharge to groundwater without impairing grain yields. Discharge of nitrate was related to both irrigation management and fertilizer application.
Marinov and Marinov, 2014	Corn, conclusions only based on model calculations no data	Balta Brailei region of Romania, Chernozem and alluvial loamy sand	Over-irrigation increases the net nitrate leakage to the groundwater, while lowering the nitrate concentration in the root regions and restricting plant growth. An intermediate trade-off between the two is necessary.
Rinaldi et al., 2007 (simulation period 1952-2004, calibration and validation based on data between 2002-2004)	Processing tomato grown in open field	Foggia, Italy, silty-clay vertisol of alluvial origin, organic matter 2.1%	Irrigation applications of 600–800 mm, which are representative of the typical application rates, will results in low WUE and NUE values. Optimal irrigation rates to minimize crop water stress and N leaching should be approximately 400 mm.
ten Damme et al., 2022 (1988-1992)	spring barley, winter oilseed rape grown in open field	Jyndevad Experimental Station in Denmark, coarse sandy soil, classified as an Orthic Haplohumod (USDA Soil Taxonomy)	Irrigation does not need to increase the risk of nitrate leaching, despite it increasing the soil water content and percolation. Synchronised management of irrigation and N-application is key for managing the risk of nitrate leaching.
Hack-Ten Broeke, 2001 (1991-1995)	Grassland grown in open field	Hengelo, The Netherlands. Cambic Podzol, organic matter 3-5%. 89 to 92% sand	Increased irrigation dose did not lead to higher nitrate leaching. In one field more irrigation led to a higher agricultural production and reduced nitrate leaching.
Vázquez et al., 2006 (2001-2002)	processing tomato grown in open field	La Rioja, Ebro Valley, Spain. Eutric Fluvisol	Over-irrigation was linked to nitrate leaching. Treatments receiving water according to 80% of maximal evapotranspiration, showed almost no drainage during the growth period and no reduction in crop yield.
Granados et al., 2013 (2006-2007)	Muskelon, Pepper grown in plastic greenhouse.	El Ejido, Almeria, Spain, an artificial layered soil, 10 cm of coarse sand on top of 30 clay	Better irrigation management by avoiding over-irrigation reduced NO_3^--N leaching by 58% in pepper and by at least 49% in muskmelon.

4.3.2.2. Irrigation technique

With respect to the application of irrigation, farmers in Europe mainly choose drip irrigation or overhead sprinkler or gun irrigation. The acreage in furrow irrigation is declining but still relevant (Lopez-Gunn et al., 2012). Hydrological studies in the Jucar river basin near Valencia (Pool et al., 2022; Pérez-Martín and Benedito-Castillo, 2023) estimated for citrus that drip irrigation would reduce the nitrogen leaching fraction from 86% to 18% compared to furrow or flood irrigation. Quiñones et al. (2007) compared nitrogen use efficiency of low frequency nitrogen applied furrow irrigated citrus trees with the efficiency of high frequency nitrogen applied drip irrigated trees with aid of a lysimeter. Drip irrigation led to a higher nitrogen uptake and a lower residual soil NO_3^--nitrogen.

Poch-Massegú et al. (2014) evaluated furrow irrigation in Girona, Spain, to be less responsible for nitrate leaching since nitrate leached mostly out the soil profile during the fallow period and not during the irrigation season. In the same study the authors point out the risk of irrigation applied for frost protection. These irrigations events are usually applied with a big volume of water during a phase where the crop does not consume a lot of water and thus imply a high risk of nitrate leaching. Next to drip irrigation, overhead irrigation applied by irrigation guns is an important irrigation technique in Europe. Zhou et al. (2018) compared both irrigation techniques in a potato experiment in Denmark. They found that there is no clear advantage of changing gun irrigation to drip irrigation. Yield, nitrogen uptake and nitrogen use efficiency were not improved with drip irrigation.

4.3.2.3. Fertigation

Fertigation is a method of fertilizing plants by applying fertilizer directly to the irrigation water. In this technique, soluble fertilizers are dissolved in water and then applied to plants through the irrigation system, allowing for a precise control over the timing and dosage of fertilizers. Experimental results in fertigation trials in Europe do not unambiguously prove its efficiency (Table 22). Quiñones et al. (2007) concluded that nitrogen use efficiency is 13% higher in citrus trees when applied under high frequency drip irrigation. Rolbiecki et al. (2020) observed a higher yield in fertigated watermelon. The added value of fertigation compared to broad band fertilization was observed to be significant but also rather moderate for arable crops (Żarski and Kuśmierek-Tomaszewska, 2023; Zhou et al. ,2018). Frequent application of nitrogen under moist conditions may entail a risk of nitrate losses to leaching (Rubio-Asensio and Intrigliolo, 2024). The importance of proper irrigation management has already been discussed. It makes sense that a correct irrigation schedule is a crucial requirement for successful fertigation application.

Table 22 Efficiency of fertigation discussed in experiments in Europe

Author and study period	**Crop(s) and growing conditions**	**Location and soil type**	**Efficiency of fertigation**
Arbat et al., 2013 (2009-2010)	Corn grown in open field	Mas Badia, Spain, an alluvial soil, very deep, moderately well drained 28-35% sand, 9-17% clay	70 m³/ha pig slurry (120 kg N) and maximal 75 kg N applied by fertigation in 3 fractions allowed maximal yield. Higher doses did nog increase yield and led to increased nitrate leaching.
Quiñones et al., 2007 (1998)	Citrus grow in lysimeters	Spain Typic Xerofluvent soil (67.4% sand, 10.8% silt, 21. 8% clay; 0.6%)	Whole tree nitrogen use efficiency was significantly higher under high frequency drip irrigation (75.1%) than under low frequency flood irrigation (62.7%).
Rolbiecki et al., 2020 (2021-2015)	Watermelon grown in open field	Krajenski near Bydgoszcz, Poland, slightly loamy sand, 1.5% organic matter	Fertigation lead to a higher yield (20%) compared to broadband fertilization, under the same irrigation treatment.
Rubio-Asensio and Intrigliolo, 2024 (2019-2022)	Lettuce or endive in the fall–winter, and melon in the spring–summer under plastic soil mulching	Torre Pacheco, Murcia, SE Spain, clay- loam texture (36% clay, 36% sand)	The results demonstrated that high frequency fertigation increases shoot fresh weight, but did not improve shoot nutrient uptake or nitrogen nutritional status. High and medium fertigation frequencies decreased nitrate concentration in the root zone, where it may be prone to leaching.
Żarski and Kuśmierek-Tomaszewska, 2023 (2015–2017)	Maize grown in open field	Bydgoszcz, Poland, Sandy soil 82% sand	The application 180 kg N trough fertigation resulted in a significant increase in the yield of grain maize compared to the traditional method of nitrogen fertilization (3% increase).
Zhou et al., 2018 (2013-2015)	Potato grown in open field	Jyndevad Research Station, Denmark soil 93% sand	Simulations with a crop model and observations showed more nitrate leaching in the gun irrigation treatment although yield between the fertigated and gun irrigated treatment was similar.

4.3.3. Environmental co-benefits

A better water management by more careful irrigation scheduling or a better water application reduces the risk of nitrate leaching. Primarily a better water management will contribute to a lower water use. Water savings reported in Portugal, Spain and Italy vary between 10 and 80% (Table 23). In Denmark, a better irrigation management led to a higher water use. Irrigation leads to higher emissions since energy is needed to bring the water to the field. However, expressed per kg of biomass irrigation, the environmental impact of food production decreases, provided irrigation is carefully scheduled. This has been documented by Abrahão et al. (2017) for irrigated corn in North West Spain and by Rogy et al. (2022) in France. Haverkort and Hillier (2011) calculated that 40 mm of irrigation in Denmark is responsible for 1.4% of the CO_2 emission of potato production while this amount increases potato production by 8%.

Table 23 Water savings realized by a better water management when irrigating

Author	Water saved
Water savings due to improved irrigation scheduling	
Cameira et al., 2003	45% (157 mm/year) of the water applied drained to the root zone; this could be prevented by better irrigation scheduling.
Diez et al., 2000	12% less water (43 mm/year) was applied in an improved irrigation schedule.
Rinaldi et al., 2007	33% (200 mm/year) or 50% (400 mm/year) less irrigation should be possible form an economic point of view when carefully scheduled.
Vázquez et al., 2006	Irrigation scheduled according to 80% of the maximal crop evapotranspiration did not result in any yield loss.
Granados et al., 2013	Irrigation volume was reduced by 17% in pepper and by 20% in muskmelon.
ten Damme et al., 2022	20 to 150 mm more irrigation applied led to higher yield and lower nitrate leaching.
Water savings due to a better irrigation application	
Pool et al., 2022	Water leakage out of flood-irrigated orchards (230 mm/year) was 1.3 times higher than the annual recharge in drip-irrigated orchards (174 mm/year).
Pérez-Martín and Benedito-Castillo, 2023	A reduction of 60% in irrigation volume was assumed (223 mm/year).

4.3.3. Minimum/threshold or optimum level of implementation

Due to the differences between regions, thresholds or optimum levels of implementation need to be judged in function of the specific field and crop conditions.

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5. CLUSTER V – BUFFER STRIPS, NATURE-BASED SOLUTIONS

5.1. Targeted establishment of buffer areas

5.1.1. Introduction

Buffer areas can be defined as areas between agricultural fields and watercourses or other protected zones, i.e., nature protection areas or surface water wells. Buffer areas are designed to reduce losses of nutrients (nitrogen and phosphorus, but also pesticides) from the agricultural fields into watercourses and drinking water abstraction locations. Buffer areas for nature protection can be defined as a certain area of land that is necessary to protect the natural protected area from anthropogenic pressures. These anthropogenic pressures can differ, but in the scope of this assessment, we will focus on nutrient losses from agricultural land use surrounding the protected area. The buffer areas are not protected sites themselves but help to reach the goals in the protected sites.

Nutrients are transported from agricultural fields towards watercourses and other protected zones either sorbed onto soil particles or dissolved in water by different routes or via the atmosphere. These transport routes are i) erosion of soil particles, ii) surface runoff of precipitation, iii) superficial subsurface runoff, iv) leaching to groundwater, v) artificial drainage, vi) atmospheric volatilization and subsequent deposition and vii) direct losses during application of fertilizers. Buffer areas can be effective in mitigating all these types of transport processes, often tackling multiple of them, depending on the local conditions. The effect of buffer areas to the latter two transport processes, i.e., atmospheric deposition and direct losses, lies in physically separating sensitive zones from the source of nutrients (i.e., the application of fertilisers and manure on agricultural fields).

The effect of buffer areas on direct losses is not discussed here, as it was already covered above in Cluster II (sloping ground, water courses). After arrival to the buffer area, immobilization or removal of nutrients within the buffer areas further reduce the risk of re-release of nutrients from the buffer areas into watercourses and other protected zones. Mechanisms involved here are the settling of soil particles and infiltration of excess water into the soil of the buffer area allowing for i) uptake of transported nutrients by vegetation, ii) sorption of phosphorus on soil particles and iii) denitrification of nitrate (Carstensen et al., 2020; Hill, 2019; Mancuso et al., 2021; Owenius & Van Der Nat, 2011; Pistocchi, 2022; Rizzo et al., 2023; Robotham et al., 2023; Walton et al., 2020; Zhu et al., 2020).

In the following, buffer areas to protect water quality are discussed separately from buffer areas to protect natural areas.

5.1.2. Buffer areas to protect water quality

Usually, buffer areas are vegetated, as the vegetation slows down incoming runoff and has a function in immobilizing incoming nutrients into the buffer area. In addition, buffer areas are often not fertilized and not treated with pesticides. To be effective, buffer areas occupy critical positions in the landscape where they can prevent the transport of nutrients and pesticides towards the sensitive zones (Hill, 2019; Owenius & Van Der Nat, 2011).

The vegetation of the buffer areas can be herbaceous, with grass or mixtures of grass and other plants, or with the addition of woody plants, i.e., shrubs and trees. A third type of buffer areas are wetland buffer areas, where the vegetation can also be herbaceous or woody, but where the soil of the buffer area is water saturated. A wetland buffer area can also be a shallow water body between the terrestrial and the aquatic system (Owenius & Van Der Nat, 2011; Walton et al., 2020).

In the following sections, herbaceous and woody buffer areas are assessed separately from wetland buffer areas, with the implicit assumption that the former type of buffer zones are not under wetland conditions.

5.1.2.1. Herbaceous and/or woody buffer area

In the design of herbaceous and/or woody buffer areas to protect water quality, the local hydrogeological system needs to be well understood. Furthermore, buffer areas need maintenance to ensure a dense vegetation cover. Harvesting could be necessary to remove immobilized nutrient loads taken up in plants. Nutrient losses from adjacent agricultural fields may still bypass buffer areas due to deeper water flow pathways, or in the case of strong surface runoff. The presence of tile drainage can also diminish the effectiveness a buffer area in the riparian zone of a watercourse, as water flowing through the drainage system can bypass the buffer area (Hill, 2019; Owenius & Van Der Nat, 2011). Finally, the risk of flooding may need to be considered due to the increased roughness of the river section (Pistocchi, 2022).

Efficiency in terms of preventing and reducing nutrient losses to watercourses

A recent meta-analysis (Rizzo et al., 2023) showed that buffer strips can reduce nitrogen losses to water from 7 to 98% and total phosphorus from -40 to 98%, when designed to intercept run-off, while buffer strips designed to remove groundwater nitrate have removal efficiencies ranging from -500 to 100% for nitrate. A negative removal efficiency indicates that the buffer area is a source rather than a sink of nutrients.

The median values presented in this meta-analysis for these two types of buffer strips are presented in Table 24 and show that overall, herbaceous and woody buffer areas along watercourses can be effective in decreasing nutrient loads towards the aquatic environment. However, the variation in their effectiveness can be large, and in a worst-case-scenario, buffer areas can be a net source of nutrient losses to the watercourse. As discussed above, proper design considering the local hydrogeology and proper management can be essential in this respect.

Table 24 Median values of nitrogen and phosphorus removal in herbaceous and woody buffer areas along watercourses in the global meta-analysis of Rizzo et al. (2023).

Type	Study area	Efficiency	Remarks	Reference
Buffer strips designed to intercept **run-off** towards watercourses	Global, studies with climate relevant for EU	72% for TN 64% for nitrate 74% for TP	Meta-analysis, 34 studies, n = 52 for TN, n = 51 for nitrate, n = 47 for TP.	Rizzo et al. (2023)
Buffer strips designed to intercept **groundwater flows** of nitrate to watercourses	Global, studies with climate relevant for EU	58% for nitrate	Meta-analysis, 43 studies, n=111 for nitrate	Rizzo et al. (2023)

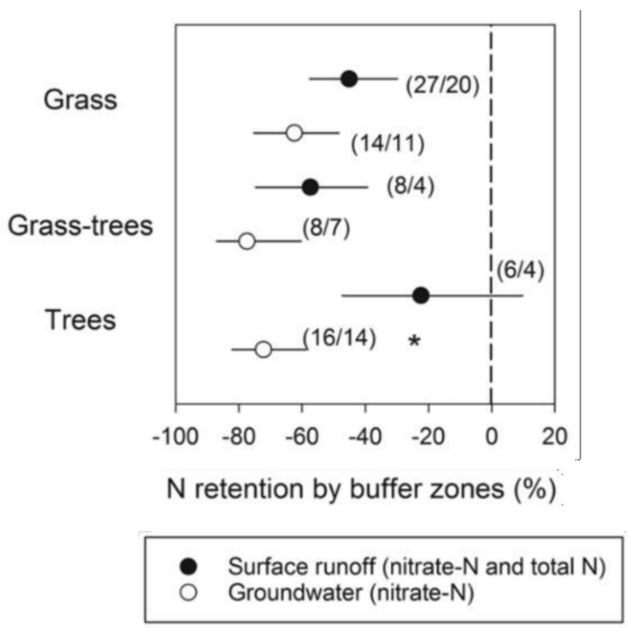
N=nitrogen, P=phosphorus, T=total, n=number of data points

Similar results were found in older meta-analyses, for example, the mean removal efficiency of nitrate was 54% in herbaceous buffer strips (n=32), 72 % in forested buffer strips (n=31) and 80% in mixed buffer strips (n=11) in a meta-analysis by Mayer et al. (2007). The results of this meta-analysis suggest that buffer strips with woody vegetation perform better than without, and that combining both herbaceous and woody vegetation therefore provides the highest nutrient removal efficiency.

This pattern could be explained by the fact that mixed vegetation buffers combine the different characteristics of both type of vegetation, i.e., herbaceous vegetation can be more effective in intercepting runoff as it provides a dense soil cover, while woody vegetation can be more effective in intercepting subsurface transport of nutrients due to deeper root systems.

Such an interpretation is corroborated by another meta-analysis (Valkama et al. (2019)), shown in Figure 23, which indicates that mixed vegetation buffer strips are most efficient in reducing both surface runoff and groundwater N, while grass buffer strips are more efficient than tree buffer strips in reducing surface runoff, and tree buffer strips are more efficient than grass buffer strips in reducing groundwater N.

Figure 23 N retention by buffer strips with different vegetation types. N retention by buffer strips targeting surface runoff and by buffer strips targeting groundwater N is presented separately. Figure from Valkama et al. (2019).



For herbaceous and woody buffer strips designed to reduce runoff to watercourses, climate variables and soil types are not found to significantly affect N-reduction efficiency. In contrast, the P-reduction efficiency in this type of buffer strips is significantly higher in regions with higher temperature, lower precipitation or soil types with more clay content. This can be partly explained by the fact that these buffer strips aim to reduce soil particle transport, which is important for phosphorus due to strong association of phosphorus with soil particles. Hence also the higher efficiency on soil types with more clay, as these soil types are more prone to erosion (Rizzo et al., 2023).

Because buffer strips intercept runoff soil particles more than runoff water, they are more efficient at removing total N and P (including N and P associated to sediments and N and P in organic matter), compared to dissolved N and P alone. Indeed, the ability to remove nitrate (dissolved) is lower in buffer strips designed to capture runoff compared to the efficiency to remove total N (Table 1). This picture is also corroborated in another meta-analysis (Valkama et al. (2019)), where it is shown that buffer strips can achieve a 17-48% reduction in nitrate N from runoff, while the reduction is 43-68% for total N from runoff. For phosphorus, the difference between the retention of dissolved and total P is less clear, due to the large ranges in the literature consulted. Based on two review studies, the efficiency of buffer strips for P can be estimated at -57 to 93% for dissolved P and -64 to 98% for total P (Dorioz et al., 2006; Roberts et al., 2012).

This shows that buffer strips can be a source of P (negative removal = release of P), although this depends very much on the local situation: the meta-analysis of Prosser et al. (2020), which included 38 studies for P, reports no negative efficiencies, whilst in the studies of Dorioz et al. (2006) and Roberts et al. (2012), 4 out of 19 studies report negative efficiencies. Net release of P can be explained by re-mobilization of previously fixed particulate P, by desorption of dissolved P whenever the top layer in the buffer strip becomes P saturated, by mineralisation of P fixed in organic material and by the reductive dissolution of iron hydroxides (Roberts et al. , 2012). Iron hydroxides are the most important P-binding minerals in the soil, but under anaerobic conditions they are unstable, they dissolve and release the bound P. Anaerobic conditions occur in wet, water-saturated soils where sufficient organic matter is present, which can therefore certainly be relevant in buffer strips along waterways.

As a conclusion, buffer strips can be effective in reducing nutrient loads to surface water, as many studies show removal efficiencies higher than 50% for N and for P. Overall, buffer strips with woody vegetation perform better than without, and combining both herbaceous and woody vegetation provides the highest nutrient removal efficiency. High variability of the efficiency is reported in literature, stressing that proper design that considers the local hydrogeology and proper management of the buffer strip is essential for a good functioning of the buffer zone. In addition, care needs to be taken to ensure no P re-release occurs under anaerobic conditions in the buffer zone.

Socio-economic consequences

The implementation of herbaceous and/or woody buffer strips is not very expensive. However, sufficient design work needs to be performed beforehand, the maintenance of buffer strips can be time consuming and an inevitable consequence of establishing buffer strips is the loss of productive farmland and related income losses for farmers (Owenius & Van Der Nat, 2011).

Environmental co-benefits and trade offs

Natural development of the buffer strips can contribute to biodiversity in agricultural landscapes. To keep a diverse vegetation, cutting and removing the herbaceous vegetation at least yearly will be necessary. This leads to a secondary co-benefit when the harvested vegetation can be reused as green manure or after composting as organic soil amendment, hereby contributing to closing the nutrient cycle within the agricultural system (Owenius & Van Der Nat, 2011). Buffer strips can also increase faunal biodiversity and can increase aquatic biodiversity as well, for example in the case where shrubs and trees provide shade over an adjacent watercourse. Depending on rules regarding public access, buffer strips may also be used for recreation (Owenius & Van Der Nat, 2011), and they will contribute to some extent to carbon sequestration in agricultural landscapes.

A potential environmental tradeoff is the risk of shifting from a water quality problem to a greenhouse gas problem. This can be an issue in cases where the buffer strip is designed to remove nitrogen through denitrification, an unavoidable by-product of which is the creation of N_2O, an important greenhouse gas.

Threshold level of implementation

Thresholds for buffer strips width in respect of subsurface nitrate removal (groundwater nitrate) depend on sediment texture and depth to a confining layer. Buffer strips need to be wider when sediments are coarse (gravel to sand) and when the permeable sediment is deeper. Buffer strips on fine textured (loam/silt/clay) and shallow permeable sediment can be more narrow. Buffers with less than 4 m depths of sand and gravel sediments in sloping landscapes often achieve 90% nitrate removal efficiency within widths of 30–60 m, whereas many buffers with less than 4 m depths of fine-grained sediments reach a 90% efficiency in 10–20 m. Conversely, studies report considerable increases in required buffer widths (up to 100-200 m) for 90% nitrate removal when permeable sediment depths increase to more than 4 m of sand or gravel layers and in some cases deep flow paths bypass the buffer (Hill, 2019).

5.1.2.2. Wetland buffer areas

Different types of wetlands can be considered to reduce nutrient input to watercourses. In the concept of a buffer area, a wetland buffer area is in this assessment considered to be water saturated soil or shallow aquatic areas located between agricultural fields and a watercourse, designed to capture diffuse nutrient pollution by draining water from the agricultural fields towards the watercourse. Wetlands constructed for manure or wastewater treatment (i.e., point sources) are not considered in this assessment.

Efficiency in terms of preventing and reducing nutrient losses

A recent meta-analysis (Rizzo et al., 2023) showed that removal efficiencies of free water surface wetland buffer areas (i.e., shallow aquatic areas) ranged between -43 and >99% for nitrate, -45 and 90% for total N and -101% and 80% for total P. A negative removal efficiency indicates that the buffer area is a source rather than a sink of nutrients. The median values reported in this meta-analysis for wetland buffer areas are presented in Table 2 and show that wetland buffer areas along watercourses can have an effect on decreasing nutrient loads towards the aquatic environment. As with the unsaturated buffer strips, the variation in their effectiveness can be large and, in a worst-case-scenario, wetland buffer areas can be a source of nutrients.

Another meta-analysis (Land et al., 2016) on wetland buffers tackling agricultural runoff reported 95% confidence intervals of 25 – 46 % total N removal and 31 – 54% total P removal. Median values reported in that study are also presented in Table 2, and show very similar results as the previous one for free water surface wetlands. Average retention values for different types of wetlands from a third meta-analysis are also included in Table 25, reporting similar summarized values with a large variation and no significant differences between the different types of wetlands.

Table 25 Median values of nitrogen and phosphorus removal in wetland buffer areas.

Type	Study area	Efficiency	Remarks	Reference
Open surface water wetland buffer areas	Global, studies with climate relevant for EU	Median values of 31% for TN, 44% for nitrate, 34% for TP	Meta-analysis, 35 studies, n = 57 for TN, n = 40 for nitrate, n = 33 for TP.	Rizzo et al. (2023, note, data were retrieved from Supporting Material)
Unspecified wetland buffer areas	Global	Median values of 36% for TN and 44 % of TP	Meta-analysis, n = 18 for TN, n= 35 for TP	Land et al. (2016)
Riperian mineral soil wetlands, peat-accumulating wetlands, floodplains	Global, 67% of studies were in the EU	Average with standard deviation between brackets of 51(31)% for nitrate, 43(30)% for TN and 21(72)% for TP. Efficiencies did not differ significantly between the different types of wetlands	Meta-analysis, n=56 for nitrate, n=58 for TN, n=55 for TP.	Walton et al., (2020)

N=nitrogen, P=phosphorus, T=total, n=number of datapoints

The variation in removal efficiency can be explained by several physical, biological and chemical variables, highlighting the importance of proper design and proper management of wetland buffer areas (Owenius & Van Der Nat, 2011). With low nitrate load into the wetland, no effect of soil type (organic vs mineral soil) was found on nitrate removal efficiency, while with high nitrate loads, organic soils had a higher nitrate removal efficiency (Walton et al., 2020), which could be explained by carbon-limitation in a mineral soil hampering denitrification of a higher nitrate load.

In the meta-analysis of Rizzo et al. (2023), statistical analysis of the effects of climate and occurrence of clay soil was done, combining results of wetland buffer areas and vegetated drainage ditches, where similar mechanisms are driving nutrient removal. The results indicate that increased annual precipitation increases total N removal, likely caused by a greater input of N, while a higher number of months with low temperature decreased nitrate removal, due to the negative effect of low temperature on denitrification. Increased potential for evapotranspiration resulted in a decrease in both total nitrogen and total phosphorus removal, which could however be due to evaporation of water from the wetland or drainage ditch, resulting in higher a concentration of nutrients in the outflow water and reducing the measured decrease compared to inflow water. No effect was found of clayey soil on nitrogen or phosphorus removal. As climate change will impact precipitation and evapotranspiration, the efficiency of wetlands to remove nutrients will likely change depending on the climate.

Socio-economic consequences

The costs of designing and establishing a wetland depends on its size, construction method and technical requirements and can be significant. In addition, maintenance can be time consuming. Finally, loss of productive farmland will have a negative impact on the local farmers' income.

Environmental co-benefits and tradeoffs

Wetland buffer areas can provide multiple ecosystem services, such as biodiversity enhancement, carbon sequestration, reservoirs for water and recreation (Land et al., 2016). An environmental tradeoff related to wetland buffer areas is that to some extent it shifts a water quality problem to a greenhouse gas problem, as the main nitrogen removal process relied on in wetlands is denitrification, a microbial process converting nitrate NO_3 to nitrogen gas N_2, an unavoidable by-product of which is the production of nitrous oxide N_2O, which is an important greenhouse gas.

Threshold level of implementation

To have a continuous water supply for a free water surface wetland, a sufficiently large catchment is a necessary precondition for this measure. Up to 200 ha of utilized agricultural land is needed per ha of established wetland, so that the wetland corresponds to about 0.5 – 4% of the total drainage area (Owenius & Van Der Nat, 2011). The exact location of the wetland and the local hydrogeological conditions will have an important impact on the efficiency of the wetland in capturing agricultural nitrates and acting as an effective buffer. A recent study based on modelling concluded that restoring 27% of wetlands historically drained for agriculture in the EU (3% of the EU land area), targeted in high nitrogen input areas, could reduce current nitrogen loads to the sea by 36%, but would have potential costs to agricultural productivity. A more efficient strategy targeting wetland restoration on farmlands projected to be abandoned by 2040 would yield a 22% load reduction and could enable major rivers such as the Rhine, Elbe and Vistula to meet water quality targets with minimal agricultural impact (Bertassello et al, 2025).

5.1.2.3. Specific case for surface drinking water abstraction points

For buffer areas to protect surface drinking water abstraction points, in theory, the above description of buffer areas along watercourses is still relevant, i.e., the design to have a good efficiency of reduction of nutrients will depend on the local hydrogeological conditions in the catchment that is draining into the surface drinking water abstraction point. Thus, buffer areas can be effective in protecting surface water abstraction points, however, they need to be properly designed to reach the desired level of protection (EPA (IE), 2011). In the Fontaine du Theil catchment, in Brittany, France in the early 2000's (Basilico & Domange, 2011), the use of buffer strips was combined with increasing erosion prevention and incremental use of mechanical pest control instead of chemical pest control to protect surface water quality. No results of nutrient reduction were documented, but the combined approach successfully reduced the presence of pesticides in water samples. Obviously, in large catchments with large scale farming that drain into surface water abstraction points, this will be increasingly more challenging, as many farmers will need to be engaged and adapt to new practices. Finally, monitoring of achieved results needs to be executed in action plans to improve the water quality of drinking water abstractions, in order to assess the true effectiveness of the measures taken (Basilico & Domange, 2011).

The discussion of socio-economic consequences, potential synergies/offsets, environmental co-benefits and threshold level of implementation of the previous sections of buffer areas, applies also for the use of buffer areas to protect surface drinking water abstraction points.

5.1.3. Buffer areas to protect natural areas

Transport routes of nutrients into protected natural areas via watercourses that flow through an area with low water quality due to agricultural pressures or by direct runoff or subsurface water flow, could be tackled by buffer areas as described above. In theory, such buffer areas can be effective in reducing nutrient pressures into the natural area. Similarly to the protection of a surface drinking water abstraction point, the size of the greater area surrounding the protected area will need to be determined, and buffers will need to be designed according to the local hydrogeological conditions.

There is little quantitative data available to assess the efficiency of buffer zones for the protection of a protected natural area against nutrient inflows. The atmospheric transport of volatile nitrogen compounds will likely not be abated sufficiently with these types of buffer areas. As some natural protected sites are very sensitive to atmospheric deposition of nitrogen, additional measures will need to be taken to ensure nutrient pressure does not jeopardize the nature protection goals of the protected area (Van Dasselaar, 2000). These additional measures could be to decrease the ammonia emissions from livestock farms in a buffer zone around the protected natural site. A scenario-analysis for Denmark showed that decreasing emissions on farms that fall within in a 250 m wide buffer zone around different types of protected natural sites certainly decreases total ammonia emissions to nearby protected natural sites (Schou et al., 2006). However, the effectiveness of a buffer zone in which ammonia emissions are reduced will depend on the type of sources that contribute to the atmospheric nitrogen deposition on a specific natural area. In case the sources are mainly local, decreasing the emissions locally can have high efficiency on the total atmospheric deposition in that area. However, when non-local sources affecting a natural area are significant, the effect of establishing a buffer zone will be lower. In that case, emission reduction measures need to be taken on a larger regional basis, to decrease "background" nitrogen deposition rates (Erisman et al., 2023; van Pul et al., 2004). A possible alternative to a buffer zone directly adjacent to natural areas could be a "targeted emission reduction plan". A study from the Netherlands (Erisman & Brouwer, 2021) estimated the effect of applying more emission reduction measures in certain areas throughout the country which had the highest contributions to total ammonia deposition in all Natura 2000 areas, and less emission reduction measures in the rest of the country. The results were compared to a generic reduction emission plan, in which the same emission reduction measures were applied to all areas. Only the ammonia emissions from stables was included in the calculation. This modelling exercise indicated that with the same percentage reduction of the total agricultural ammonia emissions in the Netherlands, the "targeted emission reduction approach" could reduce the total ammonia deposition in all Natura 2000 areas by almost 50%, while this was only 26% with the generic approach. This shows that a targeted approach based on emission and deposition modelling could be more efficient than a buffer zone approach, defining an area specifically surrounding the natural sites.

Socio-economic consequences

Buffer areas are used to protect natural areas can incur costs in terms of reduced agricultural production due to restrictions on agricultural activities within the buffer. Requirements on the reduction of ammonia emissions can require significant investments.

Environmental co-benefits

Depending on the specific measures introduced, buffer areas to protect natural areas can contribute more generally to reducing nitrogen losses.

5.2. Other nature-based solutions

5.2.1. Introduction

Nature-based solutions are actions or measures to protect, manage or restore natural ecosystems, inspired and supported by natural processes (Pistocchi, 2022). In the framework of this assessment, we focused on nature-based solution to decrease nutrient losses from agricultural fields into the natural ecosystems surrounding agricultural areas, which are not discussed yet in any of the other tasks in this assessment. In the next sections several nature-based solutions to tackle nutrient losses will be discussed, namely agroforestry, drainage mitigation measures, enhanced in-stream retention by two-stage channels and vegetated drainage ditches.

5.2.2. Agroforestry

Agroforestry is a type of farming practice wherein trees are combined with arable crops (silvoarable) or pastures (silvopasture), in a wide variety of possible combinations. Nutrient losses from agroforestry systems are lower than in a conventional farming system mainly due to decreased soil erosion due to the formation of a litter layer from leaves and stabilization of soils by tree roots, as well as the absorption of nutrients below the crop rootzone by relative deep tree roots (Zhu et al., 2020).

Efficiency in terms of preventing and reducing nutrient losses

A recent meta-analysis (Zhu et al., 2020) shows that silvoarable agroforestry systems can reduce nitrogen losses by -193 – 100% and phosphorus losses by 2 – 100%. Silvopastural agroforestry systems can reduce nitrogen losses by -48 – 91% and phosphorus losses by 0 – 38%. Linear tree plantings at field edges can reduce nitrogen losses by -114 – 98% and phosphorus losses by 0 – 50%. A negative removal efficiency indicates that the agroforestry system has higher nutrient losses compared to the conventional system. The median values reported in this meta-analysis are presented in Table 26 and show that agroforestry systems have generally lower nutrient losses compared to conventional farming systems. However, as shown above, the variation in effectiveness can be large, and in some cases, they can also lead to higher nutrient losses.

Table 26 Median values of reduction in nitrogen and phosphorus losses of agroforestry systems compared to conventional farming systems in the global meta-analysis of Zhu et al. (2020).

Type of agroforestry system	Study area	Efficiency	Remarks	Reference
Silvoarable	Global	Median values of 58% for TN, 68% for TP	Meta-analysis, 38 studies	Zhu et al. (2020)
Silvopastural	Global	Median values of 55% for TN, 57% for TP	Meta-analysis, 6 studies	Zhu et al. (2020)
Linear tree plantings	Global	Median values of 55% for TN, 18% for TP	Meta-analysis, 6 studies	Zhu et al. (2020)

N=nitrogen, P=phosphorus, T=total

Socio-economic consequences

Changing from conventional agriculture to agroforestry in arable and pasture systems can be challenging for farmers due to high costs of implementation, lack of financial incentives, limited agroforestry product marketing, lack of education and awareness and lack of field demonstrations (Sollen-Norrlin et al., 2020).

In temperate regions, yield losses of summer arable crops (i.e., potato, maize) grown in combination with older (large) trees will potentially impact the farmers' income negatively. A smart selection of crops and trees, taking the development of the trees into account by changing the crop type, could address this. In addition, proper selection and management of the tree species could have a positive effect on the farmer's income (Pardon et al., 2018).

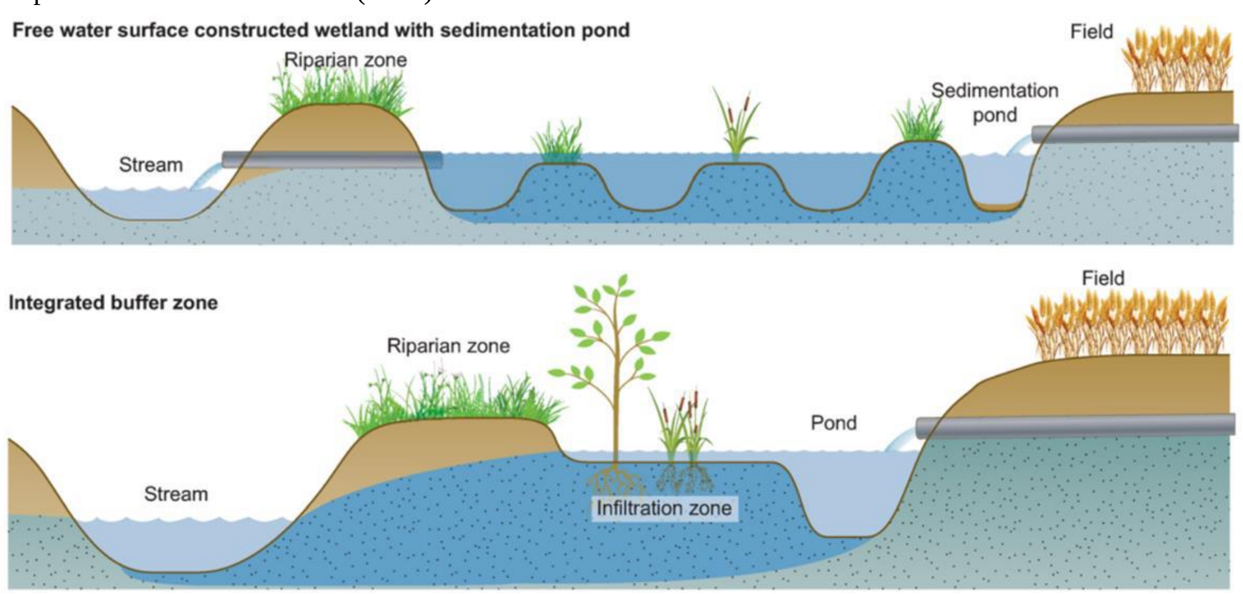
Environmental co-benefits

Agroforestry systems provide multiple ecosystem services, such as enhanced carbon sequestration in agricultural landscapes, greater biodiversity and wind erosion control (Zhu et al., 2020).

5.2.3. Drainage mitigation measures

When agricultural fields are drained with artificial drainage systems, direct transport of nutrients to watercourses is further enabled. This effect could be mitigated by a combination of rerouting the drainage outflow and natural-based solutions similar to wetland systems as in Figure 24 below. Drainage water of the agricultural field flows first into an area that lies between the field and the stream, similar to wetland buffer areas described above. This area could be designed as a sedimentation pond, followed by a free water surface wetland area or as a simple pond with an infiltration zone. The latter option is named an "integrated buffer zone", and has recently developed and tested in northwestern Europe to improve the nutrient reduction capacity of traditional riparian buffer zones. The nutrient removal process in such systems is similar to buffer areas, mainly sedimentation, uptake by vegetation and denitrification of nitrate (Carstensen et al., 2020).

Figure 24 Conceptual scheme of natural-based solutions for mitigation of nutrient losses via drainage, figure adapted from Carstensen et al. (2020).



Efficiency in terms of preventing and reducing nutrient losses

Results from a recent meta-analysis (Carstensen et al., 2020) shows that rerouting drainage outflow in combination with free water surface wetlands reduces nitrate loading with -8% – 63% and total phosphorus with -103 – 68%. A negative removal efficiency indicates the system is rather a source than a sink of nutrients. Fewer data are available regarding nutrient removal efficiencies for integrated buffer zones to mitigate nutrient losses via drainage systems, the mean removal efficiency for nitrate was 26% (standard deviation of 4%, based on only 2 studies), while it was 48% for total phosphorus (standard deviation of 6%, based on only 2 studies). The weighted averages reported in this meta-analysis are presented in Table 27 and show that overall, nature-based solutions to mitigate drainage nutrient losses can be efficient. However, as shown above, the variation in their effectiveness can be large and in a worst-case scenario even be a source of nutrients rather than a sink.

Table 27 Meta-analysis weighted average nutrient removal efficiencies of nature-based solutions to mitigate drainage nutrient losses reported in the global meta-analysis of Carstensen et al. (2020).

Type	Study area	Efficiency	Remarks	Reference
Drainage + free water surface wetland	Global	41% (standard deviation 21%) for nitrate, 33% (standard deviation 28%) for TP	Meta-analysis, 14 studies for nitrate, 15 studies for TP	Carstensen et al. (2020)

P=phosphorus, T=total

For the drainage + free water surface wetland systems, the variability in removal efficiencies can be explained by the different local conditions of the included studies, such as hydraulic retention time or carbon availability, similar as discussed for wetlands. Optimal designing, implementation and management is important and will depend on the local hydrogeological and geochemical conditions.

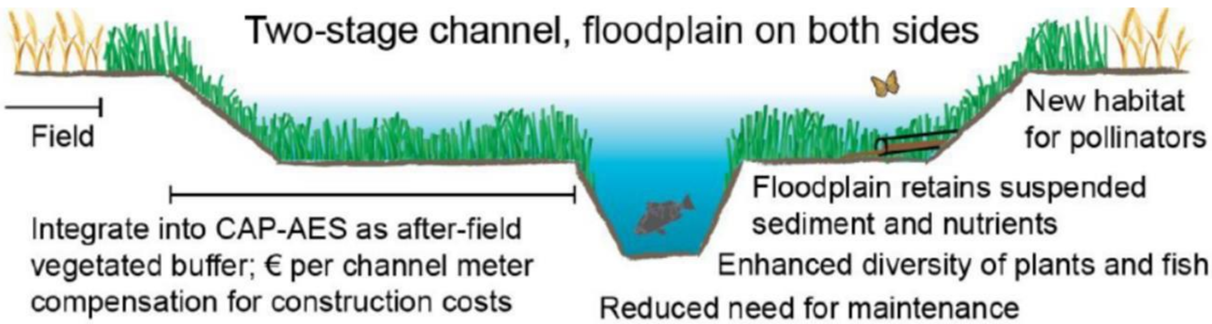
For the drainage + integrated buffer zones, few data were available and no meta-analysis weighted- average could be calculated. The nitrate removal efficiency was relatively low in the 2 studies included, likely due to suboptimal design of the system. In addition, the riparian zone after the integrated buffer zone could further remove nitrate before it reaches the stream. More studies of this type of mitigation measure for drainage nutrient losses are needed (Carstensen et al., 2020).

The discussion of socio-economic consequences, potential synergies/offsets, environmental co-benefits and threshold level of implementation of wetlands applies also here, as the drainage mitigation systems of nature-based solutions are an extension of wetland buffer areas.

5.2.4. Enhanced in-stream retention

With adaptions to the geometry or vegetation in streams, increased retention of nutrients within the stream itself (i.e., after losses from agricultural fields) could be achieved. Two examples are the two-stage channel design and vegetated drainage ditches. The two-stage channel design consist of a constructed floodplain on one or both sides of the existing main channel (Figure 25). This channel design optimizes transport of water and sediment, requires less maintenance than traditional channels (dredging) and is expected to enhance ecological functioning and biodiversity (Pistocchi (2022)).

Figure 25 Conceptual figure of a two-stage channel. Figure from Pistocchi (2022).



Vegetated drainage ditches are agricultural drainage ditches that are properly vegetated and shaped to provide the same physical and biological processes occurring in wetlands (Rizzo et al., 2023).

Efficiency in terms of preventing and reducing nutrient losses

Few studies of the effect on water quality in the EU of two-stage channels are available. A study in Finland showed a net retention (both in the floodplain and low-flow channel) of 2.1% for suspended sediment and 3.5% for total phosphorus per km of channel length (Pistocchi, 2022). However, the authors note that this study was done on a newly installed two-stage channel and they expect lower resuspension rates from the sediment in the low flow channel with time, as the loose deposits originating from the last conventional dredging have been flushed away. A study conducted in the US (Indiana, Hodaj et al. (2017)) showed that a two-stage channel decreased total phosphorus, soluble reactive phosphorus, total suspended sediment and nitrate concentrations in the water. By taking the water discharge rate into account, the study concluded that the two-stage channel also decreased the total phosphorus load (by 40%), the soluble reactive phosphorus load (by 11%) and the total suspended sediment load (by 22-40%), while no significant reduction in nitrate load was achieved. A recent meta-analysis (Rizzo et al., 2023) showed that vegetated drainage ditches have removal efficiencies of nitrate ranging from 1 – 93%, of total nitrogen ranging 9 – 92% and of total phosphorus ranging 14 – 100%. The median values reported in this meta-analysis are presented in Table 28 and show that overall vegetated drainage ditches can be efficient in removing nutrients, however, as shown above, the variation in their effectiveness can be large.

The effects of climatic and soil variables discussed for wetlands are also applicable for vegetated drainage ditches, due to similar mechanisms of nutrient removal.

Table 28 Median values of reduction in nitrogen and phosphorus in vegetated drainage ditches in the global meta-analysis of Rizzo et al. (2023).

Type	Study area	Efficiency	Remarks	Reference
Vegetated drainage ditches	Global, but only including studies with climate relevant for EU	54% for nitrate, 53% for TN, 57% for TP	Meta-analysis, n=13 for nitrate, n=16 for TN, n=10 for TP	Rizzo et al. (2023, note, data were retrieved from Supporting Material)

N=nitrogen, P=phosphorus, T=total, n=number of data points

Socio-economic consequences

Two-stage channels can either be implemented by changing the current geometry of the stream or by lower the adjacent riparian area next to the stream. The installment of two-stage channels will have a significant cost, but maintenance costs will be lower compared to the traditional channel, due to lower dredging requirements. The installment of two-stage channels can be disruptive to local farmers and could possibly result in area losses (when adjacent riparian area next to the stream needed would be productive area). Fear for increased risk of flooding outside of the stream could hinder adoption rates by locals. Proper design of the two-stage channels and communication towards the local stakeholders of the area could improve this. Vegetated drainage ditches do not need changes in geometry, but do require periodic removing of excessive vegetation. Ditches that are essential for water discharge of a flooding prone area are not a good candidate for the installment of vegetated drainage ditches (Hodaj et al., 2017; Pistocchi, 2022).

Environmental co-benefits

Two-stage channels have positive effects on plant biodiversity and improve riparian and instream habitat quality. In addition, these types of channel improve water conveyance with lower management requirements. This is beneficial in light of climate change, which may increase the need for efficient drainage and water flow conveyance (Hodaj et al., 2017; Pistocchi, 2022).

An important environmental offset of enhanced in-stream retention of nutrients is the possibility to shift a water quality problem to a greenhouse gas problem, when the main nitrogen removal process is denitrification of nitrate to N_2-gas, where N_2O is an unavoidable byproduct.

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