Chapter 4. The unintended consequences on the nitrogen cycle of conservation practises in agriculture

This chapter warns against the possible unintended effects of nitrogen pollution management measures by providing a case study on agriculture. The chapter reviews the various practices implemented in the United States to manage agricultural nitrogen pollution. The possible unintended effects of each measure are detailed and general lessons are learned.

Regardless of the policy approach (risk, precautionary), it is crucial to consider the reality of the nitrogen cascade to guide nitrogen policy making (see Chapter 1. for a description of the cascade). The design of best nitrogen management practices, and ultimately of instruments to incentivise them, must evaluate their unintended effects on other forms of nitrogen due to the nitrogen cascade. This principle applies to all nitrogen sources such as agriculture, energy, transportation, industry and wastewater treatment.

This Chapter focuses on agricultural practices and argues that they can have unintended consequences if the nitrogen cascade is not taken into account. Many transformations of nitrogen in various forms contribute to its movement between terrestrial, atmospheric, and aquatic ecosystems. Because of this lability, the intended beneficial effects often become unintended detrimental effects for adjacent ecosystems, or even within the ecosystem to which nitrogen is applied (Follett et al., 2010).

4.1 Managing nitrogen for agriculture and the environment

Increased use of commercial nitrogen fertiliser in the United States fuelled an increase in yields, but also posed increased risks to environmental quality. The Natural Resource Conservation Service (NRCS) of the U.S. Department of Agriculture (USDA) evaluated the use of conservation practices on U.S. cropland in major river basins using survey data collected over 2003-06 (USDA-NRCS, 2011a, 2012b,c, 2013a,b). The state of nitrogen management was based on a set of criteria defining appropriate application rate, method of application, and timing of application. Meeting all three of these three criteria would provide adequate nutrients to crops but reduce excess applications that are most at risk for leaving fields and entering air and water.

NRCS found that, on the whole, a great deal of improvement was needed (Table 4.1). The low percentages of good nutrient management practices in the Upper Mississippi, Lower Mississippi and Ohio-Tennessee are noteworthy because this is where a majority of corn is grown. Corn is the largest user of nitrogen, both on a per-acre[1] and total use basis. This region is also a major source of nitrogen entering the Gulf of Mexico via the Mississippi River, which is the primary cause of the large zone of hypoxic water found there (Alexander et al., 2008).

Table 4.1. Percentage of cropland meeting nitrogen rate, method, and timing criteria consistent with "good" management, Mississippi River Basin, 2003-06

River Basin	% cropland
Ohio-Tennessee	17
Upper Mississippi	14
Lower Mississippi	14
Missouri	35
Arkansas-White-Red	33

Source: USDA-NRCS Conservation Effects Assessment Project.

The Clean Air Act and Clean Water Act regulate the environmental impacts of nitrogen in the air and water. USDA conservation programmes promote general improvements in nutrient management through voluntary programmes. Decisions on which conservation measures to adopt are made by individual farmers. In protecting a particular environmental medium from agricultural pollution, specific sets of management practices are promoted, either through regulation (rare in the United States) or financial assistance and education (common). The choice of practices to support is often made based on their expected effectiveness for preventing pollutant losses to a particular media. The focus on one pathway can result in unintended consequences that are detrimental to other media.

In the following section the movement of nitrogen is traced for crop production and for confined animal operations. Both are important sources of nitrogen to the environment, with different management approaches.

4.2 Nitrogen pathways in crop production

Emissions of nitrogen to water and to the atmosphere are not independent events, but are linked by the biological and chemical processes that produce the various nitrogen forms. Crop production is characterised by stochastic weather and soil conditions that affect crop yields and nitrogen loss. From a nutrient standpoint, crop production is a "leaky" system. It is impossible to ensure that every bit of nutrient input to cropland via direct application of commercial fertiliser or animal waste, or fixed by legumes, is taken up by the planted crop. Nitrogen applied to cropland can be "lost" in a variety of ways:

- *Soil erosion* - Nitrogen can be lost from the soil surface when attached to soil particles that are carried off the field by wind or water. Although wind and water erosion can be observed across all regions, wind erosion is more prevalent in dry regions and water erosion in humid regions. Overall, little nitrogen is lost through erosion when basic conservation practices are in place (Iowa Soybean Association, 2008).
- *Runoff* – Surface runoff of dissolved nitrogen, generally in the form of nitrate (NO_3^-), is only a concern when fertiliser and or manure are applied on the surface and rain moves the nitrogen before it enters the soil (Legg and Meisinger, 1982; Iowa Soybean Association, 2008).
- *Leaching* - Leaching occurs when there is sufficient rain and/or irrigation to move easily dissolvable NO_3^- through the soil profile (Randall et al., 2008). NO_3^- eventually ends up in underground aquifers or in surface water via tile drains and groundwater flow. Tile drains may be a chief passageway by which nitrogen moves from crop soils to surface water in regions with high water tables (Turner and Rabalais, 2003; Randall et al., 2008; Randall et al., 2010; Petrolia and Gowda, 2006).
- *Ammonia (NH_3) volatilisation* - Significant amounts of nitrogen can be lost to the atmosphere as NH_3 if animal manure or urea is surface applied and not immediately incorporated into the soil (Hutchinson et al., 1982; Fox et al., 1996; Freney et al., 1981; Sharpe and Harper, 1995; Peoples et al., 1995). Additionally, warm weather conditions can accelerate the conversion of manure and other susceptible inorganic nitrogen fertilisers to NH_3.

- *Denitrification* - When oxygen levels in the soil are low, some microorganisms, known as denitrifiers, will convert NO_3^- to dinitrogen and nitrous oxide (N_2O) (Mosier and Klemedtsson, 1994). Denitrification usually occurs when NO_3^- is present in the soil, soil moisture is high or there is standing water, and soils are warm. The ratio of dinitrogen to N_2O is governed by the amount of oxygen available to the denitrifying organisms. The higher the level of oxygen, the greater the amount of N_2O produced.

To maintain economically viable farming operations, farmers manage temporal variability in weather and soil nitrogen by applying more nitrogen than plants need to protect against downside risk (i.e. use an "insurance" nitrogen application rate) (Sheriff, 2005; Babcock, 1992; Babcock and Blackmer, 1992; Rajsic and Weersink, 2008). This ensures that nitrogen needed by the crop is available. Additionally, farmers may take a "safety net" approach to maximise economic returns by setting an optimistic yield goal for a given field based on an optimum weather year to ensure that the amount of nitrogen needed for maximum yields is available (Schepers et al., 1986; Bock and Hergert, 1991). Thus, during years in which weather is not optimal for maximising yields, nitrogen will be over-applied from an agronomic standpoint. Almost by definition, optimal conditions are infrequent, so farmers following this approach over-fertilise crops in most years. The decision to apply "extra" nitrogen is economically justified if the cost of over-applications is low compared to the cost of under-application (Rajsic and Weersink, 2008).

4.3 Nitrogen pathways in animal production

Nitrogen cycling in animal production adds the dimension of manure production, storage and management. Nitrogen enters the system in animal feed. Some of the nitrogen is retained in the animal products (meat, milk, eggs), but as much as 95% is excreted in urine and manure, much of which is applied to cropland as fertiliser (Follett and Hatfield, 2001).

Manure can collect in or under the production house for a few hours or several months, depending on the collection system. Production houses are ventilated to expel gases that are emitted, including NH_3. The manure is eventually removed from the house to a storage structure (lagoon, tank, pit, or slab) and stored anywhere from a few days to many months. Losses of nitrogen to air and water can occur during this time, depending on the system and the extent of contact with rain and wind. The stored manure is eventually transported to fields where it is applied. Losses to air and water from the field vary, depending on application method and rate. Nitrogen in the field helps produce crops, which may in turn be fed to animals, thus completing the cycle. Nitrogen lost to the air eventually returns to earth, where it can be a source of plant nutrients, or be lost, as decribed above.

The form nitrogen takes in its journey from animal to field depends on a host of factors, including storage technology, manure moisture content, temperature, air flow, pH, and the presence of micro-organisms. Reducing nitrogen movement along one path by changing its form will increase nitrogen movement along a different path (NRC, 2003). For example, reducing NH_3 losses from a field to the atmosphere by injecting waste directly into the soil increases the amount of nitrogen at risk of moving to water resources as NO_3^- (Oenema, 2001; Abt Associates, 2000). Ignoring the interactions of the nitrogen cycle in developing manure management policies could lead to unintended and adverse effects on environmental quality.

4.4 Conservation practices and the nitrogen cycle

Standard conservation practices have been evaluated for their impacts on nitrogen losses along different pathways. Changes to crop rotations as a means of reducing nitrogen losses are not considered, other than the addition of a cover crop.[2] The following practices influence nitrogen losses to the environment. Their effectiveness varies tremendously across the setting (crop, soil, climate, management skill) in which they are applied. The discussion is therefore limited to general impacts. It is recalled that practices designed to achieve specific environmental policy objectives are often used in combination (Box 4.1).

Box 4.1. Conservation practices are often used in combination

Nitrogen pollution problems cannot usually be solved with one management practice because single practices do not typically provide the full range and extent of control needed at a site. Multiple practices are combined to build management practice systems, which are generally more effective in controlling the pollutant since they can be used at two or more points in the pollutant delivery process. On the other hand, a set of practices does not constitute an effective management practice system unless the practices are selected and designed to function together to achieve specific environmental policy goals. For example, if water quality is the environmental policy goal, nutrient management, conservation tillage, field borders, and riparian buffers can be associated to control nitrogen losses. Conservation tillage can help reduce overland transport of nitrogen by reducing erosion and runoff, and nutrient management will minimize subsurface losses due to the resulting increased infiltration. Filter strips can be used to decrease nitrogen transport by increasing infiltration, and through uptake of available nitrogen by the field border crop. Nitrogen not controlled by nutrient management, conservation tillage, and filter strips can be intercepted and remediated through denitrification in riparian buffers.

Each practice in a management practice system must be selected, designed, implemented, and maintained in accordance with site-specific considerations to ensure that the practices function together to achieve the overall management goals. If, for example, nutrient management, conservation tillage, filter strips, and riparian buffers are used to address a water quality problem, then planting and nutrient applications need to be conducted in a manner consistent with conservation tillage goals and practices (e.g. injecting rather than broadcasting and incorporating fertiliser). In addition, runoff from the fields must be conveyed evenly to the filter strips which, in turn, must be capable of delivering the runoff to the riparian buffers in accordance with design standards and specifications.

Source: USEPA (2003).

4.4.1 Nutrient management

Nutrient management is defined by USDA's NRCS as managing the amount, placement, form, and timing of the application of plant nutrients to the soil (USDA-NRCS, 2006). The focus is on overall nutrient use efficiency (NUE),[3] and not on any particular vector or pathway of nutrient loss to the environment. But decisions related to the rate, timing, and method of application have implications for the form and pathway of nitrogen movement in and out of the soil.

Nitrogen can be applied to cropland in a variety of forms. Some of the more widely used nitrogen fertiliser forms include anhydrous NH_3 (gas), urea (solid), Urea Ammonium Nitrate (liquid),[4] and manure (solid). These forms vary in how quickly they can be transformed into NO_3^-, which is what crops actually use. The closer in time the fertiliser is applied to when the crop needs it, the faster it needs to be transformed into NO_3^-. A mismatch of fertiliser form with appropriate timing can lead to large environmental losses and poor yields.

NUE can be increased by placing fertilisers in the soil rather than broadcasting them on the surface. Such placement reduces the risks of losses to the atmosphere or to surface runoff. Subsurface placement of nitrogen also reduces losses stemming from NH_3 volatilisation. However, some studies have found that incorporation into the soil increases N_2O emissions (Flessa and Beese, 2000; Wulf et al., 2002; Drury, 2006). Injection or incorporation could also increase NO_3^- leaching, especially where soils are coarse textured (Abt Associates, 2000).

NUE can be increased by improving the synchronisation between crop nitrogen demand and the supply of nitrogen throughout the growing season (Doerge et al., 1991; Cassman et al., 2002; Meisinger and Delgado, 2002). This implies maintaining low levels of inorganic nitrogen in the soil when there is little plant growth and sufficient inorganic nitrogen fertiliser during periods of rapid plant growth. For example, the corn plant's need for nitrogen is not very high until about 4 weeks after it emerges from the ground, which typically falls in June through July in the major corn producing regions of the United States. However, farmers often apply nitrogen in the fall for a spring-planted crop, due to lower fertiliser prices and time constraints in the spring.

Cold soils and the use of nitrogen inhibitors can slow the conversion of nitrogen fertiliser to NO_3^-, but losses to leaching can be quite substantial. Switching to spring applications can reduce overall application rates and NO_3^- loss to water, but may increase N_2O emissions as applications are shifted to generally warmer, wetter conditions (Delgado et al., 1996; Rochette et al., 2004; and Hernandez-Ramirez et al., 2009). Applying nitrogen during the growing season also introduces some risk, as weather may prevent an application when plants need it, negatively affecting yields. Whilst nitrification inhibitors are effective at decreasing direct N_2O emission and NO_3^- leaching, recent studies suggest that they may increase NH_3 volatilisation and, subsequently, indirect N_2O emission (Lam et al., 2017).

Precision agriculture technologies are playing an increasing role in farm production, creating more opportunities for increasing NUE. Precision technologies – such as tractor guidance systems using a global positioning system (GPS), GPS soil and yield mapping, and variable-rate input applications (VRT) – help farms gather information on changing field conditions to adjust fertilisation practices (Schimmelpfennig, 2016). The first wave of precision agriculture was GPS guidance for tractors; introduced in the early 1990s it is now widespread (Schmaltz, 2017). GPS reduces driving errors and any overlap of fertiliser spreading. Uptake of the second technological wave, VRT, is currently estimated at 15% in North America and is expected to continue to grow rapidly (ibid). The third wave may well be that of "Big Data" and the fourth, in the longer term, that of robotics. Big Data are massive volumes of data with a wide variety that can be captured (e.g. by drones), analysed and used for decision-making (Wolfert et al., 2017).

4.4.2 Tillage

Conservation tillage is a popular practice that has several attractive features, including reduced fuel use, more available soil water, and reduced soil erosion. Reducing soil erosion keeps soil particles and attached nutrients out of surface water. In some studies, no-till systems have been shown to reduce NO_3^- leaching over conventional tillage, as well as proper crop rotation, especially those including a nitrogen-fixing crop (Meek et al., 1995). However, another study has shown that conservation tillage increases the infiltration rate of soils (Baker, 1993). The potential for NO_3^- leaching may increase due to greater availability of water and increased soil porosity (i.e. large pore spaces) (USEPA, 2003). Soil macroporosity and the proportion of rainfall moving through preferential flow paths often increase with the adoption of conservation tillage, potentially increasing the transmission of NO_3^- and other chemicals available in the upper soil to subsoils and shallow groundwater (Shipitalo et al., 2000). Also, reduced tillage creates an environment more favourable to ammonification and denitrification, which could lead to increased emission of nitrogen forms to the atmosphere. However, this depends on soils and soil moisture (MacKenize, Fan, and Cadrin 1997).

4.4.3 Cover crops

Cover crops are planted after the principal crop has been harvested to provide soil surface cover and reduce soil erosion, and to prevent nitrogen from leaching or running off to surface water (Dabney et al., 2010). Cover crops do this by taking up water and NO_3^-, thus reducing the volume of NO_3^- percolating through the soil. For example a 6-year (1999-2005) study on corn and soybeans in Canada showed that planting a winter wheat cover crop could reduce NO_3^- losses relative to no cover crop while increasing yields (Drury et al., 2014). Cover crops have the added benefit of building up soil carbon. However, cover crops are not as effective at reducing atmospheric losses of nitrogen, as these occur primarily during or shortly after fertiliser applications.

4.4.4 Filter strips

Vegetative filter strips remove sediment, organic matter, and other pollutants from surface runoff, thus keeping them out of streams. The impact on dissolved nitrogen is less clear. Denitrification may be enhanced by the filter. Also, NO_3^- leaching may increase because surface water movement is slowed. Fares et al., 2010 review the literature on vegetative filters and the nitrogen cycle.

4.4.5 Restored wetlands

Restored wetlands have been shown to be effective at reducing the movement of nitrogen from cropland to surface water (Jansson et al., 1994; Hey et al., 2005; Mitsch and Day, 2006). Wetland vegetation takes up nitrogen, and wet soils enhance denitrification. Most of nitrogen removal occurs through denitrification (Crumpton et al., 2008). There is some support for the belief that denitrification produces more of the inert dinitrogen and little N_2O (USEPA, 2010; Hernandez and Mitsch, 2006; Crumpton et al., 2008).

4.4.6 Field drainage

Tile drainage is used to reduce runoff and increase soil drainage. It lowers the water table of fields that would otherwise be too wet to crop intensively. Drained soils tend to be highly productive. About 26 % of cropland receiving nitrogen is tiled, most of this in corn production (Ribaudo et al., 2011). However, about 71 % of tiled cropland acres do not meet the three nitrogen management criteria.

Tiles provide a rapid conduit for soluble NO_3^-, effectively bypassing any attenuation that may occur in the soil or in vegetative buffers. As such, tile drains can have the undesirable effect of concentrating and delivering nitrogen directly to streams (Hirschi et al., 1997). NO_3^- losses from fields with tile drainage are a major source of water quality degradation in areas where tiled fields are present (Dinnes et al., 2002; David et al., 2010, Randall et al., 2010).[5]

In order to reduce the nitrogen pollution caused by tile drains, other management practices, such as nutrient management for source reduction and biofilters (or bioreactors) that are attached to the outflow of the tile drains for interception, may be needed (USEPA, 2003). However, bioreactors also enhance denitrification.[6]

On the other hand, practices which control runoff may contribute to reduced in-stream flows (USEPA, 2003). One practice that is being used to control NO_3^- losses is drainage water management (DWM). DWM is the process of managing the timing and the amount of water discharged from agricultural drainage systems. During the growing season water levels are lowered (drainage increased) to allow optimal crop growth. During fallow periods the water level is allowed to rise, which reduces the volume of drainage water leaving a field and enhances denitrification within the soil profile (Randall et al., 2010).

4.4.7 Chemical additions to manure

Chemicals such as alum can be added to manure during its collection in order to bind odorous compounds and to reduce NH_3 emissions (Moore et al., 2000). By doing so the nitrogen content of manure that is spread on fields is increased. This could pose an increased risk for NH_3 and NO_3^- losses from the field, unless appropriate field management practices are employed.

4.4.8 Tank covers

Covering manure storage tanks can greatly reduce the discharge of NH_3 to the atmosphere, primarily by altering pH and preventing the formation of NH_3 (Jacobson et al., 1999). While reducing NH_3 emissions, covers also increase the nitrogen content of effluent that is eventually spread on fields, increasing the potential for both NH_3 emissions and loss of NO_3^- through leaching unless appropriate field management practices are employed.

4.4.9 Slurry lagoon covers

Plastic covers that float on the lagoon surface or that are tented over lagoons can greatly decrease the emission of gaseous nitrogen forms (Jacobson et al., 1999; Arogo et al., 2002). Some systems (anaerobic digesters) also capture methane and use it as a biofuel to generate electricity. Covering a lagoon increases the nitrogen content of the effluent that is eventually sprayed on fields. While NH_3 emissions from fields sprayed with lagoon effluent might increase, the net effect is a reduction in NH_3 emissions from both lagoon storage and field applications. However, the risk of NO_3^- loss to water increases.

4.4.10 Manure incorporation and injection

About 10% of crop acres treated with nitrogen received animal manure in 2006 (Ribaudo et al., 2011). Whereas 62 % of the cropland acres receiving only commercial fertiliser were not meeting the criteria for good nitrogen management, 86 % of cropland acres receiving only manure were not meeting the criteria, and 96 % of the cropland receiving both commercial fertiliser and manure were not meeting the criteria. Research has indicated that farms with confined animals tend to over-apply nutrients to crops, primarily because of the large amount of manure produced relative to available cropland on the farm (Ribaudo et al., 2003; Gollehon et al., 2001).

Rapidly incorporating manure into the soil, either by plowing or disking solids into the soil after spreading, or by injecting liquids and slurries directly into the soil, reduces NH_3 emissions (Abt, 2000; Arogo et al., 2002). But this also increases the nitrogen available for crops in the soil, and thus the risk of NO_3^- leaching and runoff to water bodies. Therefore, manure incorporation and injection needs to be accompanied by appropriate soil testing and other field management practices.

4.5 Changing nutrient management on cropland may result in environmental trade-offs

A study by USDA's Economic Research Service (ERS) (Ribaudo et al., 2011) assessed how the adoption of different management practices affected the movement of nitrogen along different pathways by using the Nitrogen Loss and Environmental Assessment Package (NLEAP) model with Geographic Information System (GIS) capabilities (Shaffer et al., 2010; Delgado et al., 2010). Of particular interest in this research was the extent to which trade-offs in environmental outcomes might occur as overall NUE is improved.

Because NLEAP is a field-level model, eight different soils in four States (Arkansas, Ohio, Pennsylvania, and Virginia) were selected to assess changes in nitrogen emissions to the environment from management changes in non-irrigated corn production.[7] Four of the soils were type A or B soils (well drained), and four were type D soils (relatively poorly drained). For each soil, two rotations (corn-corn and corn-soybeans), two tillage practices (conventional and no-till), and two sources of nitrogen (inorganic fertiliser and inorganic fertiliser + animal manure) were examined. The slopes for these soils were 0 to 6%, with low erosion potential.

For each soil/rotation/tillage/nitrogen-source combination, eight different scenarios were modelled with NLEAP. Starting from a baseline where at least one of the criteria for good nitrogen management is not being met (right rate, right timing, and right application method), changes in management are made so that all criteria are met. Changes in only timing and/or method of application assumed that application rate did not change.

All the scenarios show the expected changes in total nitrogen losses as the criteria were met, with reductions indicating improvements in NUE. The NLEAP results were consistent with the expectation that nitrogen emissions are minimised when all three criteria are met. Since nitrogen cascades through different forms and ecosystems, the long-term environmental benefits of reducing total nitrogen are clear. However, some of the trade-offs between different forms of nitrogen could pose environmental problems. In our example, adopting injection/incorporation always increased NO_3^- leaching, sometimes substantially (more than doubling leaching in some cases). Similarly, shifting applications from fall to spring (without changing application rate) reduces NO_3^- losses and total nitrogen losses but increases N_2O emissions as applications are shifted to generally warmer, wetter conditions, which is consistent with the findings of Delgado et al., 1996, Rochette et al., 2004, and Hernandez-Ramirez et al., 2009. Due to the potential for increasing greenhouse gas (GHG) emissions, this outcome would have to be carefully considered when making recommendations to improve NUE.

Injection/incorporation and eliminating fall applications again produced mixed results. NH_3 emissions are always reduced. Leaching is generally reduced, but in some cases where manure is used, it may increase. N_2O emissions almost always increase, from 5 to 50%, depending on the situation. Only reducing the application rate guarantees that losses of all three forms of nitrogen are reduced (Mosier et al., 2002; Meisinger and Delgado, 2002). This suggests that in areas where leaching to drinking water sources is possible, improvements in NUE could focus on application rate reductions or improvements in timing.

4.5.1 NRCS Conservation Effects Assessment Project

These trade-offs can also be observed at a larger scale. NRCS evaluated the effects of conservation systems (tillage, nutrient management, erosion controls) on nitrogen loss from cropland in 12 large watersheds in the United States as part of the Conservation Effects Assessment Project (CEAP) (USDA-NRCS, 2011a,b; 2012 a,b,c; 2013 a,b,c; 2014 a,b,c,d; 2015). The emphasis on nutrient management by USDA has been to reduce losses to water. Through a combination of field surveys and modelling, the fraction of nitrogen applied to cropland that is taken up by crops (a measure of NUE), as well as losses through different pathways were estimated for cropping practices observed during the years 2003-06.[8] In general, the percentage of nitrogen application that is taken up by crops ranged from 60% in the South Atlantic Gulf to 76% in the Souris-Red-Rainy (Table 4.2).

Table 4.2. Changes in nitrogen loss to the atmosphere with the adoption of nutrient management practices, by major watershed

Watershed	Crop uptake of nitrogen applied (%)	Change in per-acre nitrogen loss through volatilisation, with vs. without observed conservation measures (%)	Change in per-acre nitrogen loss through denitrification, with vs. without observed conservation measures (%)
Delaware River	72	1	16
Chesapeake Bay, 2006	62	-4.2	1.2
Chesapeake Bay, 2006-2011	66	3.2	1.9
South Atlantic Gulf	60	23	32
Great Lakes	72	7	-6
Ohio-Tennessee	73	21	0
Upper Mississippi	72	3	5
Lower Mississippi	64	10	45
Texas Gulf	57	-11	-9
Missouri	75	-11	-4
Arkansas-White-Red	72	-31	-34
Souris-Red-Rainy	76	-28	-28
Pacific Northwest	72	-37	-26

Source: USDA-NRCS (2011 a, b; 2012 a, b, c; 2013 a, b, c; 2014 a, b, c, d; 2015).

NRCS estimated through modelling how observed management practices affect nitrogen loss assuming an alternative state of no conservation practices. The results are summarised in Table 4.2. In some major agricultural watersheds, such as Upper and Lower Mississippi and Ohio-Tennessee, the observed mix of conservation practices were estimated to have increased the amounts of nitrogen lost to the atmosphere due to volatilisation and denitrification (3 % and 5 %, respectively) even though total nitrogen losses were reduced (NUE increased). This is generally because nitrogen fertiliser remained on the field longer, where it is exposed to wind and weather conditions that promote volatilisation and denitrification. On the other hand, nitrogen losses to all pathways were estimated to have been reduced by the observed management systems.

The results for the Chesapeake are particularly instructive. A field survey was conducted both in 2006 and 2011, spanning a time period when there was great emphasis on reducing nutrient loads to the Chesapeake Bay. Between 2006 and 2011, NUE was estimated to have increased from 62% to 66%, yet nitrogen losses to volatilisation (NH_3) and denitrification (primarily dinitrogen but also N_2O) were estimated to have increased 3.2% and 1.9%, respectively. The management practices chosen to reduce nitrogen losses to water (nutrient management, cover crops, and erosion controls) appeared to have resulted in increased losses to the atmosphere.

An analysis of the pathways of nitrogen in air and water would inform trade-off management between air pollution and water pollution. Following the Chesapeake Bay example, about half of the NH_3 emission from the Chesapeake watershed is not returning to the watershed because of atmospheric transport (Linker et al., 2013). It is estimated that 90% of the load deposited on the watershed is attenuated in the forests, fields, and other land uses, and about half of the remaining 10% is lost in transport in rivers to the tidal waters of the Bay (ibid). In other words, of 100 kg of NH_3 lost from volatilisation from the Bay cropland only an estimated 2 kg make it to the eutrophic waters of the tidal Chesapeake. Further, NH_3 emissions are generally not controlled by any Clean Air Act regulations.

4.6 Water-air trade-offs in manure management

Clean Water Act regulations require concentrated animal feeding operations (CAFOs) to meet a nutrient standard for land application in order to keep NO_3^- out of ground and surface water. This generally means more land is required for spreading manure than has been used in the past, an expensive proposition for many large farms (Ribaudo et al., 2003). One rational management response under a nitrogen standard is to encourage volatilisation of manure nitrogen (e.g. use of uncovered lagoons, surface application to fields) to reduce the nitrogen content in waste, thus allowing higher application rates on cropland and reducing the amount of land needed for spreading (Sweeten et al., 2000). But, such a strategy would increase atmospheric emissions of NH_3 and worsen air quality. Zilberman et al., 2001 cite the multi-path nature of animal waste as one reason why policies are inadequate. A policy focused on nitrogen applications to land also allows the build-up of other potential pollutants in the soil, such as phosphorus,[9] and ignores problems such as odour and dust.

An ERS study evaluated farm-level responses by hog producers of environmental restrictions on nitrogen emissions to water, nitrogen emissions to air, and nitrogen emissions to both water and air. This analysis demonstrated the potential unintended consequences of a policy that ignores the nitrogen cycle (Aillery et al., 2005).

The water protection policy considered by the ERS study was a nitrogen application standard that limited applications to a maximum of 50% more than crop uptake. The largest cost faced by a confined animal operation characterised by excess nutrient applications when confronted with a nitrogen application limit is the cost of hauling manure to a larger land base (Ribaudo et al., 2003). Hog operations met the restriction by spreading on more land, and for operations storing manure in tanks, decreasing slurry injection by 11.8% (thus enhancing NH_3 emissions and reducing the nitrogen content of the slurry). Overall, NH_3 emissions increased by 3.4%, mainly because more land is receiving manure and because of the switch by some producers from injection to surface application.

The ERS-modeled air policy was a limit on NH_3 emissions from pit operations at 10% above the minimum obtained if all manure is injected. For lagoon operations, NH_3 emissions were constrained to 20% above what is obtainable if lagoons were covered. The constraint on NH_3 emissions is in the form of a percentage reduction in net nitrogen emissions per pig. The NH_3 limits induced pit operations to switch from surface application to injection on some land, and induced some lagoon operations to cover their lagoons. The NH_3 limits resulted in a 38% decline in NH_3 emissions from manure storage facilities (the largest source of NH_3 emissions), and a 57% increase in NH_3 emissions from fields, for a net decline in emissions of 29%. The increase in emissions from fields results because more lagoons are covered, which raises the nutrient content of the lagoon liquid applied to fields, resulting in greater nitrogen volatilisation. The NH_3 standard resulted in a 79% increase in excess nitrogen applied to soil[10] (greatly increasing threats to water) – revealing an important trade-off between water and air quality.

4.7 Nitrous oxide (N_2O) management practices

Recent studies have examined how N_2O management practices could be a cost-effective lever for mitigating agricultural GHG emissions. For example, Pellerin et al., 2013 identified practices that could reduce N_2O emissions without entailing major changes in production systems or a significant reduction in yield (less than 10%) and estimated their mitigation potential and their cost over 2010-30. The study shows that eight practices are cost-effective in reducing N_2O emissions (Table 4.3). Since the final effect of these practices is a reduction of the amount of nitrogen applied on fields the risk of nitrogen pollution swapping is minimised.

Table 4.3. Practices to reduce nitrous oxide (N_2O) emissions

Practices	Sub-practices
Reduce the application of mineral nitrogen fertilisers to reduce the associated N_2O emissions	
Reduce the use of synthetic mineral fertilisers through their more effective use and making greater use of organic resources	Adjust nitrogen fertiliser application rates to more realistic yield targets* Make better use of organic fertiliser* Adjust application dates to crop requirements* Add a nitrification inhibitor Incorporate fertiliser to reduce losses*
Increase the use of legumes to reduce the use of synthetic nitrogen fertilisers	Introduce more grain legumes in arable crop rotations Increase legumes in temporary grassland*
Store carbon in soil and biomass	
Introduce more cover crops, intercropping and green cover strips in cropping systems	Introduce more cover crops Introduce more vineyard/orchard cover cropping
Optimise grassland management to promote carbon storage	Make the most intensive permanent and temporary grassland less intensive by more effectively adjusting nitrogen fertiliser application*
Modify the animal diet to reduce enteric methane emissions and N_2O emissions related to manure	
Reduce the amount of protein in the livestock diet to limit the quantity of nitrogen excreted in manure and the associated N_2O emissions	Adjust nitrogen content in the diet of dairy cows* Adjust nitrogen content in the diet of pigs*

Note: * Cost-effective measure (i.e. with negative annual cost per tonne of CO_2e avoided).
Source: Pellerin et al. (2013).

4.8 Summary, conclusions and areas for further analysis

Controlling nitrogen emissions from field applications without considering the nitrogen cycle can lead to unintended consequences. For example, injecting/incorporating nitrogen inorganic and organic fertiliser into the soil reduces atmospheric losses, but can increase NO_3^- leaching. Fertilising only during the growing season can reduce NO_3^- loss to water but may increase N_2O emissions. Reducing applied nitrogen reduces losses via all pathways.

Tillage and vegetative filters can also lead to unintended consequences. Conservation tillage reduces runoff of nutrients and sediment but may increase NO_3^- leaching and atmospheric losses. Vegetative filter strips remove sediment, organic matter, and other pollutants from surface runoff but may increase NO_3^- leaching and atmospheric losses.

Field drainage is a major source of NO_3^- losses to water. Drainage water management reduces the volume of drainage water and pollutants leaving a field but the practice can enhance denitrification and atmospheric losses of N_2O. Bioreactors reduce NO_3^- concentrations in field drainage by enhancing denitrification but losses of N_2O to the atmosphere could increase.

Manure management decisions affect nitrogen losses. Tank/lagoon covers reduce odour and atmospheric losses of NH_3 but increase the nitrogen content of the material that is eventually applied to fields, increasing the risk of loss during application as well as the cost. Chemical additions to manure such as alum reduce odour and NH_3 emissions but also increase the nitrogen content of material applied to fields.

NRCS CEAP evaluated the effects of conservation systems (tillage, nutrient management, erosion controls) on nitrogen losses in 12 large US river basins. Conservation systems reduced nitrogen losses to water, but in certain circumstances increased nitrogen losses to the atmosphere at the watershed scale.

ERS modelled an NH_3 emission limit on hog operations without restrictions on land applications. The ERS study revealed that reducing NH_3 emissions from manure can increase NO_3^- losses.

A recent study shows that the ultimate effect of cost-effective practices for reducing N_2O emissions is a reduction in the amount of nitrogen applied to the fields. All the risks of pollution by agricultural nitrogen are thus minimised.

In summary, trade-offs between air and water are prevalent in agricultural nitrogen management. Unintended consequences of uncoordinated farming practices can lessen overall environmental gains. Uncoordinated farming practices could impose extra costs on farmers. Reducing nitrogen at the source (fertiliser and manure over-application) could address multiple problems. The amount of manure nutrients that needs to be spread on land could be reduced through feed management and alternative uses for manure.

In conclusion, the multi-pollutant nature of nitrogen flows in the environment suggests there are benefits to coordination of practices, rather than a piecemeal control (Bull and Sutton, 1998; Baker et al., 2001). Both field and regional studies have shown focusing on one particular environmental media, such as water quality, can create incentives for management that result in degradation in a different media (air). Trade-offs might not be an issue if one media can absorb an increase in nitrogen loads. But if increases are undesirable, then farming practices that consider all pathways of nitrogen loss are more efficient than farming practices that do not.

In general, not creating pollution in the first place avoids the problems posed by trying to control different nitrogen pathways. In the case of cropland, reducing nitrogen application rates to achieve a better NUE reduces nitrogen loss along all pathways, avoiding the trade-offs characteristic of most other conservation practices. In the case of nitrogen and confined animal operations, increasing the efficiency of nutrient conversion to animal products can reduce nitrogen in the waste. This would reduce the threats to air and water quality, and make addressing either by managing manure less costly.

The analysis in this Chapter focuses on selected nitrogen management practices. Other management practices could be analysed, such as crop rotation, livestock feed management or irrigation. Assessing the specificity of nitrogen-fixing crops and organic farming could also be of interest. On the latter, for example, there appears to be no scientific consensus that N_2O emissions are lower in soils receiving manure compared to soils receiving inorganic nitrogen fertilisers (Graham et al., 2017).

Notes

[1] One acre equals 0.4 hectare.

[2] Also called « catch crop ».

[3] Defined as the fraction of nitrogen applied to cropland that is taken up by crops.

[4] Urea Ammonium Nitrate (UAN) is a liquid fertiliser consisting of a blend of ammonium, nitrate and urea.

[5] Much of the tile-drained cropland is located in the Mississippi River Basin, which has implications for hypoxia in the Gulf of Mexico.

[6] A bioreactor is a structure at the end of a tile line that is filled with wood chips or other material that promotes denitrification (Christianson et al., 2012).

[7] These four States were selected because they present a wide variation in growing conditions and because the data necessary for running NLEAP were already developed.

[8] For example, in the Upper Mississippi Basin, given the observed mix of management systems, the average amount of total nitrogen lost from a field to the environment was about 39 pounds per acre (USDA-NRCS, 2012c). This is parsed among the nitrogen pathways as follows: 6.9 lbs/acre due to volatilisation; 2.3 lbs/acre due to denitrification; 2.1 lbs/acre due to windborne sediment; 8.8 lbs/acre due to surface runoff; 18.7 lbs/acre due to subsurface flow.

[9] Because the ratio of phosphorus to nitrogen in manure is greater than the ratio needed by plants, meeting a nitrogen application standard allows the continued over application of phosphorus.

[10] Increasingly stringent NH_3 reductions in the US hog sector increase the amount of excess nitrogen applied to fields (Aillery et al., 2005).

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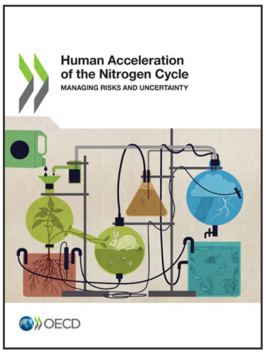

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