The European Nitrogen Case

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The N budget for Europe (excluding the former Soviet Union) indicates that the 3 principal driving forces of the acceleration of the European N cycle are fertilizer production (14 Mt (mill. tonnes) N yr^{-1}), fossil fuel combustion and other industry (3.3 Mt N yr^{-1}) and import of N in various products (7.6 Mt N yr^{-1}). The various leaks of reactive N species from European food, energy and industrial production systems are estimated and their effects on human health and terrestrial and aquatic ecosystems are assessed. Future European environmental policy measures to close the N cycle and to reduce leaks of reactive N can best focus on the three major driving forces, taking into consideration the possible consequences in the N cascade. Critical loads may be useful tools in determining N-emission ceilings and developing integrated policies for regulating N flows such as fertilizer use and imports and N levels.

INTRODUCTION

The Earth's nitrogen (N) cycle is strongly influenced by anthropogenic activities. Since pre-industrial times the total rate of N fixation has doubled (1). N fixation is the transformation of the highly abundant but biologically unavailable form, dinitrogen (N_2), in the atmosphere to "reactive" forms, including ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), nitrogen oxides (NO and NO_2), N_2O_5 and nitrous oxide (N_2O). Human activities have also increased the mobility of reactive N within and between terrestrial and aquatic ecosystems and the atmosphere (2). While N pollution occurs at the local level, its influence spreads regionally and globally as it moves through air and water across political and geographic boundaries. Losses of reactive-N compounds to the atmosphere and aquatic systems result not only in impacts on human health, visibility, and global warming, but also on natural and agricultural terrestrial and aquatic ecosystems.

The global biogeochemical cycle of N is discussed by Galloway and Cowling (this Ambio issue). This paper focuses on the major N flows on the European scale using a budget appproach. In this paper, Europe excludes the former Soviet Union. The implications of excess N for human health and ecosystems will be discussed on the basis of the European N budget, as well as options for reducing the various N flows and losses of reactive-N and existing and future policies in this respect. Regional budgets have specific flows such as N in imports and exports of products that do not occur in the global budget; these indicate environmental problems and issues related to the N cycle typical for the region considered. The type and direction of policies may therefore vary from one region to another.

THE EUROPEAN N BUDGET

Various sources of information were used to construct the European N budget presented in Figure 1 and Table 1. These include deposition estimates (3), emission estimates for NH_3 from agriculture and other sources (4) and NO_y from soils (5) and total NO_y emissions estimates (3), biological N fixation (6), fertilizer production and use (7, 8), N emission from point sources (6) and the riverine N flow to seas (9).

For some of the N fluxes assumptions had to be made due to lack of data. The trade of N in agricultural products in Europe

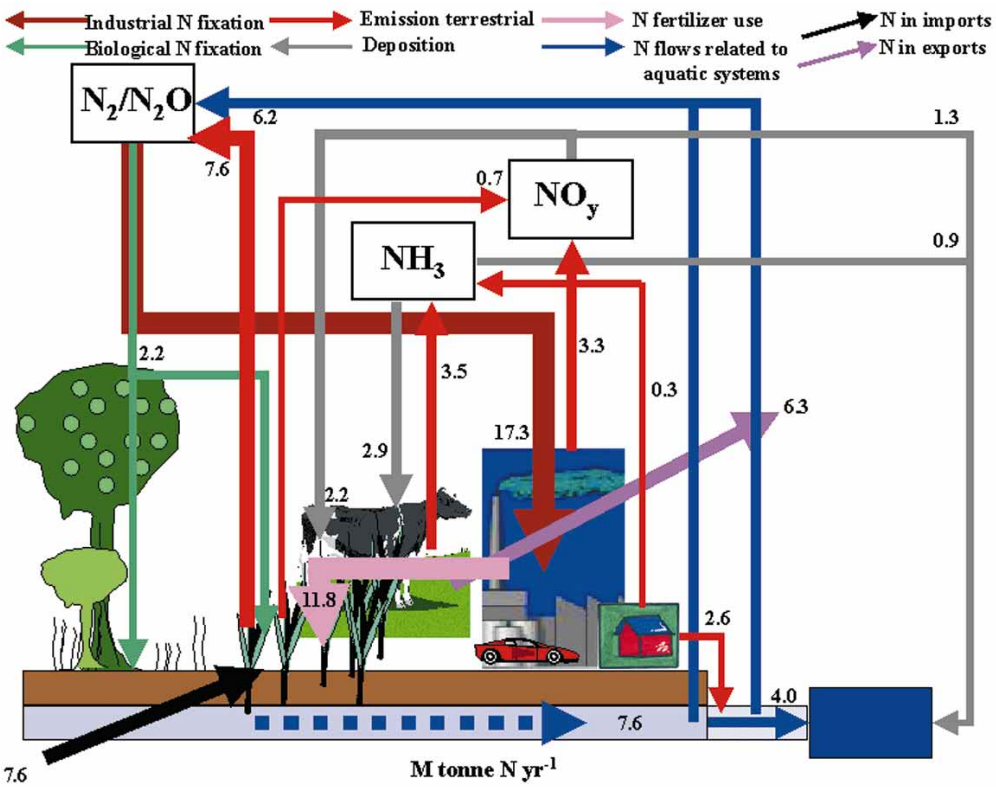


Figure 1. N budget for Europe (excluding the area belonging to the former Soviet Union) estimated from literature data.

was estimated from trade statistics (7) and the N levels in the sum of imports and exports of food, animals and feedstuffs from a recent inventory (10) for the EU. For forestry products, data from statistics (7) were used in combination with N levels in EU forestry flows (10) to calculate the European N flows. For trade in N fertilizers, we took the average of annual net trade for Europe (i.e., 1.3 Mt N net export of N fertilizers, and total import and export data including intranational trade based on fertilizer statistics (tonnes = metric tonnes) (7, 8). Denitrification and leaching losses from soils were assumed to be 50% of the surface N-balance surplus of agriculture and natural terrestrial ecosystems on the basis of Van Drecht et al. (6); soil-N accumulation was ignored. Furthermore, it was assumed that all NO_y emissions from combustion stemmed from fixation of atmospheric N_2. Finally, the denitrification loss from groundwater and riverine systems was calculated as total N leaching plus N from point sources minus riverine N transport to coastal seas. Here, the lifetime of N in groundwater, and retention of N in groundwater and surface waters was ignored.

Although there are major uncertainties in the various estimates due to scantiness of data and the use of different sources of information, the N budget indicates the order of magnitude of the various N flows and leaks in the system (Table 1). In Europe, the major process of N fixation is found in the energy and industrial production system (production of N fertilizers, 14 Mt N yr^{-1}; fossil-fuel combustion and other industry, 3.3 Mt N yr^{-1}). An important part of the N fertilizers produced is exported (2.2 Mt N yr^{-1}). Biological N fixation adds another 2.2 Mt N yr^{-1} of reactive N to the system. Another source of reactive N is the import of products (animals, animal feed, food, fertilizers, forestry products, etc.). Imports exceed exports, whereby the relative importance of the various components differs. For example, imports of feedstuffs exceed exports, while exports of N fertilizers exceed imports. Further inputs of reactive-N come from atmospheric deposition of NH_3 (3.8 Mt N yr^{-1}) and NO_y (3.5 Mt N yr^{-1}).

Total denitrification losses from soils, groundwater and surface waters are considerable. Denitrification products include a small amount of N_2O, for Europe approximately 0.5 Mt N_2O-N yr^{-1}. Additional N_2O (0.3 Mt N_2O-N yr^{-1}) stems from industrial sources and fossil-fuel combustion. N_2O emissions form only a minor N flow in the N budget. However, since N_2O is a powerful greenhouse gas, the increasing N_2O production plays an important role in the climate change issue.

Further losses from the system include emissions from agricultural production systems (0.7 Mt NO_y-N and 3.5 Mt NH_3-N yr^{-1}) and combustion, and industry and households (0.3 Mt NH_3-N and 3.3 Mt NO_y-N yr^{-1}).

There are also internal cycles involved, for example, in livestock production systems. Fertilizer N and animal manure are used in the production of food and animal feed. Since N efficiencies vary from less than 10% for cattle to about 20% for pigs and 30% for chickens (11), losses of N from these production systems are considerable. These losses end up in different compartments of the environmental system (N emissions including unreactive-N_2 and reactive-N_2O from denitrification, NH_3 volatilization to air, and NO_3^- leaching to groundwater). The emissions of reactive-N to air may be re-deposited to the surface, where further processing may occur.

The primary driver behind the acceleration of the European N cycle is N-fertilizer use. This increased from 4.6 to 11.8 Mt N yr^{-1} between 1960 and 1995. In the same period, the annual imports of agricultural products increased by a factor of 3 to 5. Another important cause of increasing NO_y emissions is energy use.

It is interesting to compare the flows of reactive N on the European scale with those on the global and Asian scale using data presented by Galloway and Cowling (this issue) and Zheng et al. (this Ambio issue). Total N mean annual deposition fluxes in Europe are close to 10 kg N ha^{-1}, while the Asian mean is about 7 kg N ha^{-1} and the global mean comes to about 5 kg N ha^{-1}, indicating the spatial concentration of the European N cycle. If we consider the annual fertilizer application (mineral fertilizer and animal manure) per ha of agricultural land (arable land + grassland), the mean N-application rate in Europe of about 90 kg N exceeds the Asian mean by 80% and the global mean by more than a factor of 3. European grasslands in particular are much more intensively used and fertilized than in other parts of the world. However, if we consider the areas that are actually fertilized from a recent inventory and exclude unfertilized areas, the Asian annual application rate of 152 kg N ha^{-1} exceeds the European (123 kg N ha^{-1}) and global (105 kg N ha^{-1}) rates. The high N-application rates in Asia reflect the high cropping intensity as a result of double and triple cropping. Finally, annual NH_3 emissions from agricultural systems, calculated as the NH_3-N flux ha^{-1} of total agricultural land in Europe, come to 16 kg, while the Asian mean flux is 10 kg and the global mean flux is about 8.

Although the total and per capita emissions of NO_y in North America are substantially higher than in Europe, the emission per unit area in Europe is about a factor of 2 higher than in North America and a factor of 3 higher than the world average (12). This indicates that just as NH_3, the European NO_y flows are strongly concentrated in a confined space.

Table 1. N budget for Europe excluding the area of the former Soviet Union (summarized from Fig. 1).

N input		N output	
N-fertilizer production	14.0	Denitrification	13.8
Combustion + other industry[a]	3.3	Emissions of NH_3 and NO_y	7.8
Biological N fixation	2.2	Sewage and industrial sources	2.6
Deposition	7.3	Riverine transport to the sea	4.0
Imported products	7.6	Exported products	6.3
Total	34.5	Total	34.5

Unit: Mtonne yr^{-1}.
[a] N fixation during combustion of fossil fuels.

EFFECTS OF INCREASED N INPUTS AND LEAKS

Reactive-N can cause a sequence of effects on its road from being fixed from N_2 (either natural or anthropogenic) to being denitrified to N_2 again. This sequential process is referred to as the N cascade (13). This section outlines the main effects in Europe without claims to completeness and without a thorough description of the causal relations.

Human Health

A thorough review of the direct and indirect impacts of excess nitrogen on human health is presented by Wolfe and Patz (this Ambio issue). In this paper, we briefly discuss health effects in European cities caused by air emissions of NO_y. Health effects related to N occur mainly near sources of emissions, primarily in urban areas. NO_x leads to a wide variety of health impacts because of the various compounds and derivatives in the family of nitrogen oxides, including nitrogen dioxide, nitric acid, nitrous oxide, nitrates and nitric oxide. First, NO_x contributes to ground-level ozone (smog). Second, NO_x reacts with NH_3, moisture and other compounds to form nitric acid and related particles. Human health concerns include effects on breathing and the respiratory system, damage to lung tissue and premature death. Small particles penetrate deeply into sensitive parts of the lungs and can cause or worsen respiratory disease such as emphysema and bronchitis, and aggravate existing heart disease. Finally, in air NO_x reacts readily with common organic chemi-

cals, and even ozone, to form a wide variety of toxic products, some of which may cause biological mutations. Examples of these chemicals include the nitrate radical, nitroarenes and nitrosamines (for more information see http://www.epa.gov). The direct effects of NO_x and derivatives will be dealt with here, while effects of ground-level ozone will not be discussed.

The EEA-Topic Center on Air gives detailed information on the situation of air quality in Europe, with special emphasis on small particles and ozone (14). An overview of the air quality in a selection of European cities is presented in Table 2, which shows the number of exceedances with respect to the standards of any of the average annual, maximum 24-hr and maximum hourly concentrations. Although these standards are based on what is known about the health impact of NO_x, their relationship to real health is not directly known and is generally calculated using statistical models.

Aquatic systems

Effects of nutrients can be studied both at system and species level. For this purpose aquatic systems are often subdivided into fresh- and saline water.

Freshwater systems are eutrophic in large parts of Europe. Phosphorus (P), not N, is the limiting nutrient in reference conditions in most of the lakes and reservoirs. Increasing N inputs in fresh waters will therefore have no direct effect on the growth of algae, but effects on macrophytes will occur. In freshwater systems the main effects of increased N inputs are associated with the shift from oligotrophic to eutrophic water. Such a shift generally results in relative increases of species that are less N-efficient, while species with optimal growth conditions at lower nutrient/nitrogen levels and N-fixing species tend to disappear with increasing N concentrations. N-enriched lakes and reservoirs are dominated by phytoplankton, while duckweed is the dominant species in N-polluted ditches and other small watercourses with low stream velocity. For fast-flowing rivers such a shift is generally not observed, but problems may occur where rivers feed into other more stagnant systems such as estuaries.

Recent reviews of the effects of increased nutrient inputs in European coastal waters and seas by OSPAR and EEA (15, 16) concluded that shifts in species composition and disappearance of species have occurred during the past 2 decades. For the Channel and Atlantic coastal seas a large number of algal and seaweed blooms have been reported, some of which were toxic to fish and shell fish and, as a consequence, had human health effects. The majority of such blooms are, however, caused by natural processes (17). In the North Sea area the growth of macroalgae in UK estuaries frequently occurs. In the Baltic (in the deep basin) and the German Bight anoxia has been reported, although its occurrence seems to be less frequent in more recent years.

The Black Sea has had numerous episodes involving algal blooms, mortality of seabed species and even mass development of jellyfish. Here hypoxia and anoxia occur almost every year. In the Mediterranean, eutrophication events (mainly blooms) have occurred since the early 1970s, with reported anoxia and death of fish in the Adriatic Sea yearly, increasing in frequency since 1975. Several species have disappeared.

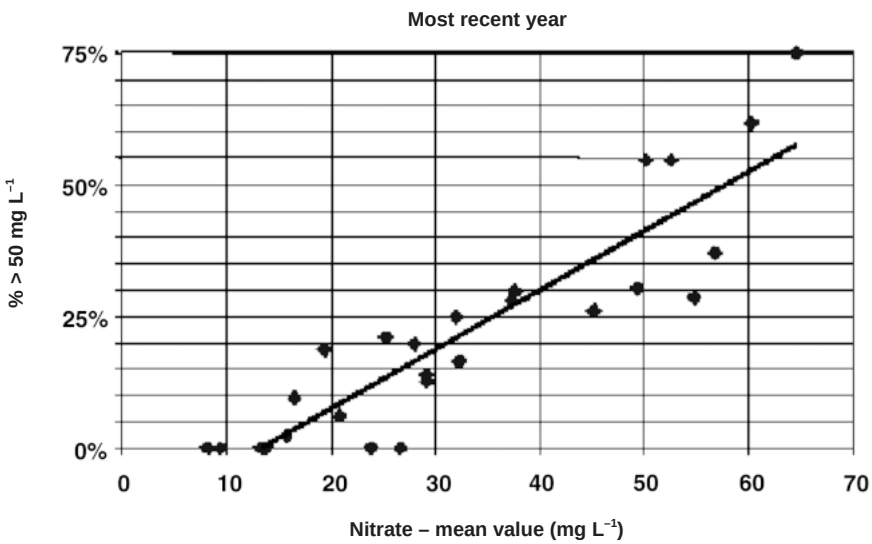
The toxicity of algae is correlated to the ratio of concentrations of P and N in the water. An attempt to evaluate reports on effects (15, 16) using data on nutrient inputs and concentrations has, so far, been unsuccessful. There are still no agreed-on in-

Table 2. NO_2 concentration in selected European cities.

City	Year	Number of Stations	City background (average)			
			Annual average	Max. 24 hrs	Max. 1 hr	Number of exceedances[a]
			(μ m^{-3})			
Athens	1990	3	57	137	386	0
	1995	4	46	104	244	0
Barcelona	1990	3	44	108	271	0
	1995	6	50	94	218	0
Berlin	1990/1	18	39	98	189	0
	1995	19	33	64	132	0
Glasgow	1990	1	50	103	218	2
Katowice	1985	2	97	215		33
	1990	2	80	194		31
	1994	3	33	75	129	0
	1995	25	28	82	207	0
London	1990	3	69	172	288	7
	1995	4	50	156	313	9
Lyon	1995	3	50	139	262	10
Manchester	1990	1	53	222	365	8
	1995	1	44	147	346	4
Milan	1990/91	1	104		621	
	1996	4	87		321	
Prague	1995	11	37	90	160	0
Sarajevo	1990	1	36	562		2
	1995	4	29	149	324	0
Turin	1990	1	56	219	357	0
	1996	2	52	108	265	0

Source: EEA (15).
[a] Exceedance of maximum values for annual average, 24 hour and hourly mean concentration. Up to 2001 the maximum admissible mean hourly NO_2 concentration was 200 µg m^{-3} (P98) for a reference period of one year. As of 1 January 2001 the standard for the mean hourly concentration is maintained at 200 µg m^{-3}, while the maximum admissible mean annual concentration is 40 µg m^{-3}; the admissible exceedance of these 2 values is 50% in 2001, decreasing to 0% in 2010; the mean hourly concentration may be exceeded during 18 days each year.

Figure 2. Correlation between nitrate mean values and the frequency class > 50 mg L^{-1}. (Data for the latest year available; 1992–1998).



dicators for effects of nutrients (N specifically) in fresh and marine aquatic systems in Europe (18). For trend analyses and policy advice this is a serious shortcoming.

Groundwater

Depending on soil type and land use a substantial portion of European groundwater bodies is affected by excess input of N. A preliminary overview was recently presented by the European Environment Agency, Topic Centre on Inland Waters (19). This technical report provides N-data on 33 groundwater bodies. In 24 groundwater bodies the mean value of at least one sampling site exceeded the standard of 50 mg L^{-1} (as NO_3^-). In 6 cases, the mean value of the whole groundwater body exceeded the standard. An overview of the frequency distribution of NO_3^- concentrations for the most recent year in the period 1992-1998 for which measurement data are available is presented in Figure 2.

Natural Terrestrial Ecosystems

The most important effects of N deposition include accumulation of N, soil-mediated effects of acidification and increased susceptibility to secondary stress factors, such as ozone or drought. Increasing N inputs can lead to unbalanced nutrition, N saturation of ecosystems and leaching of NO_3^-. The availability of nutrients is an important factor determining the species composition of vegetation, and N is the limiting nutrient for plant growth in many sensitive ecosystems. Most plant species from such habitats are adapted to nutrient-poor conditions and can only compete successfully on soils with low N levels. The impacts of increased N deposition on species diversity are diverse, and their severity depends on abiotic conditions; e.g. soil buffering capacity, soil-nutrient status and soil factors that influence the nitrification potential and N-immobilization rate.

The European N budget shows higher levels of airborne N pollution for European terrestrial ecosystems than for most other world regions due to higher intensity of the N cycle.

In European terrestrial ecosystems long-term N enrichment has gradually increased the availability of N in several vegetation types, resulting in increasing occurrence of more nitrophylic plants at the cost of characteristic species adapted to N-poor conditions. This is clearest under oligo- to mesotrophic soil conditions (20). Soil acidification is especially important in weakly buffered soil systems (with limited availability of exchangeable base cations), with the dominance of acid-tolerant species increasing at the expense of plants typically growing on intermediate soil pH.

Data suggest that in most European wetlands, seminatural grasslands, heathlands and forests, N deposition most seriously affects the changes in competitive relationships between plant species (20). Soil acidification caused by enhanced N deposition is most important in acidic coniferous and deciduous forests, with moderate effects on most heathlands and neutral to acidic grasslands. The direct toxic effect from N deposition is considered to be low (20). For example, the species composition of forest ground vegetation is already affected at low N loads, with an increase in N indicating fast-growing species occurring in both coniferous and deciduous forests (20). In several European countries, heathlands have become increasingly dominated by grasses, with up to 35% of the Dutch heathlands affected.

The concept of critical load is used to assess effects of N deposition on ecosystems. A critical load is defined as "a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge" (21). The exceedances of critical loads are calculated on a regular basis (22) thus providing a measure for potential effects in the ecosystems in the areas in which these critical loads have been derived (both terrestrial and aquatic). Examples of exceedance maps for both a historical and a future year (for an agreed-on scenario) are presented in Figure 3, and form the basis for negotiations among parties to the Convention on Emission Reductions.

TECHNICAL OPTIONS FOR REDUCING LEAKS OF REACTIVE N

As was noted above, the input of industrially fixed N in the form of fertilizers increased from 4.6 to 11.8 in the period, 1960–1995, i.e. by a factor of 2.5. In addition, the N budget clearly indicates that N in imported products also forms a major input to the sys-

Figure 3. Exceedance of the critical N load in 1995 (upper panel) and 2010 (lower panel). Source: CCE (22).

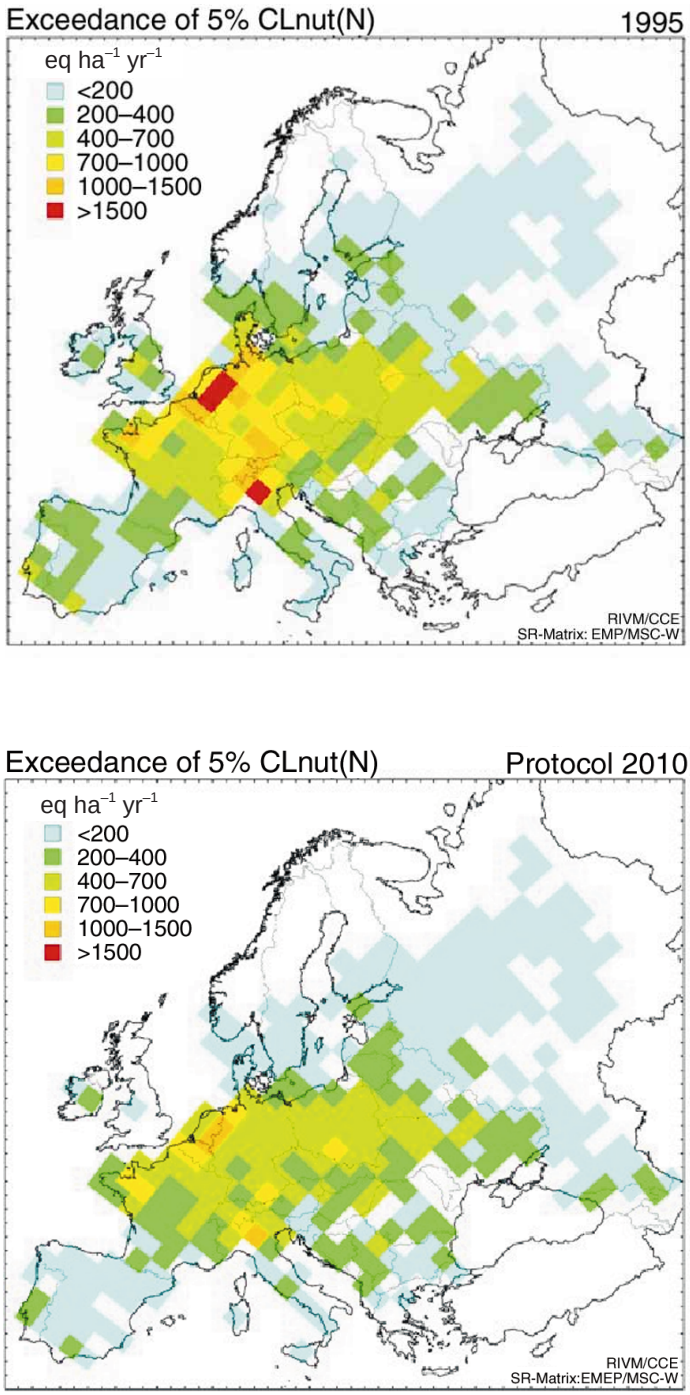


Table 3. Options for decreasing N flows, reducing N leaks and improving efficiency.

Option	Effect[a] Total N flow	Effect[a] Emission	Effect[a] Reactive-N in system	Remarks
Decreasing N flow				
– Reduction of livestock production	x	x	x	
– Lower share of livestock products in human diet	x	x	x	
– Reduction of over-production in agriculture	x	x	x	
Reducing leaks and improving efficiency				
– Optimizing feed conversion	x	x	x	
– Low-emission housing systems	–	x	–	More reactive N is retained in the manure; emission may occur with application
– Coverage of manure storage systems	–	x	–	
– Slurry injection	–	x	–	Reactive N remains available; air emission of N_2O and leaching to groundwater may be higher compared to broadcasting
– Mineral accounting system	x	x	x	
– Fertilizer recommendations that account for all inputs of N	x	x	x	
– De-NO_x catalytic reduction in industry	x	x	x	Formation of N_2O
– Three-way catalysts in vehicles	x	x	x	Formation of N_2O
– Energy-saving	x	x	x	
Other options				
– Transport of animal manure from N-surplus to N-shortage areas	–	–/x[b]	–	Prevents excessive application of animal manure
–Sustainable energy	x	x	x	

[a] x denotes a reduction, – denotes no effect.
[b] Depends on whether mineral fertilizer is substituted by animal manure.

tem, showing a sharp increase in the past decades. A major amount (25%) of the imported N is in the form of animal feedstuffs.

Because a number of internal N cycles are related to agricultural production causing reactive N emissions and re-deposition, it is clear that fertilizer inputs and imported animal feed form the major causes of the acceleration in the European N cycle. It is therefore logical to seek reduction of losses of reactive-N in the agricultural system. Additional possible reductions can be achieved in other sectors, such as the energy sector.

In agricultural production systems there are several options to reduce emissions of reactive N, such as reduction in the N input, increase in the efficiency of N use and reduction of leaks in the system (Table 3).

Livestock production in Europe is already efficient compared to many other parts of the world (12, 24). Improvements could be achieved by optimizing animal feed conversion, and preventing or reducing losses of NH_3 from animal manure. Simple techniques are available to reduce emission from housing systems (see also Oenema and Pietrzak, this issue; and Smil, this issue). Low-emission housing systems and coverage of manure storage systems reduce NH_3 losses and increase the N content of the manure, thus improving its quality as a fertilizer. However, if this manure-N applied in the field is not taken up completely by the crop or grass, the reactive N will be emitted to the atmosphere or groundwater.

Implementation of a mineral accounting system to monitor inputs, outputs and losses on the farm scale is one way to achieve further improvements (23). Leaks in the farm system become visible and solutions can be found to improve farm management and prevent excessive N application rates. In crop and grass, production efficiencies can be increased further through fertilizer recommendations that account for inputs from atmospheric-N deposition, better timing and management of fertilizers, choice of fertilizer compound, matching fertilizer application with crop needs, use of N-responsive varieties, integrated use of synthetic and organic fertilizers, etc. (24), (see also Oenema and Pietrzak; Roy et al.; and Fixen and West, all in this Ambio issue). The development of codes of best agricultural practice (25) may form a basis for reducing losses and increasing efficiency.

Although biological production is generally less efficient in terms of N, the N pollution may be less severe as a result of less spatially concentrated production and more possibilities to close the N cycle at the farm level than in landless production systems. This also involves the use of crop residues, and better use of food-processing by-products (for example, as animal feed).

With respect to the energy, transportation, and industrial sectors there are several options for reducing leaks of reactive N (Table 3): *i*) directly, through de-NO_x catalytic reduction (industry) and 3-way catalysts (transportation); and *ii*) indirectly, through energy-saving or the use of sustainable energy (see also Bradley and Jones, this issue). The disadvantage of catalytic reduction of NO_x emission is that N_2O may be formed as a by-product.

POLICY OPTIONS TO REDUCE LEAKS OF REACTIVE N

There are a number of problems in formulating policies on agricultural production. N is necessary and not substitutable; further, maintaining adequate food supplies requires the supply of adequate N to plants. In addition, N is a highly mobile element, moving through air and water and crossing political and geographic boundaries. Hence, policies aimed at solving environmental problems associated with N must often be multinational in scale. One further problem is that policies aimed at such aspects as reducing fertilizer use have to be created by a number of institutions, such as Ministries of Agriculture, Finance and Environment. Often these institutions have opposing objectives. Another problem is that policy-makers at the national level are generally different from those engaged in international activities, such as the World Trade Organization (WTO), World Health Organization (WHO), United Nations (UN), etc.

Policies aimed at preventing overproduction of some products, which still occurs in the EU, will lead directly to a reduction in the major N flows. Furthermore, reduction of the N input into the agricultural system can be achieved by a shift from an export-directed towards a more self-sufficient production. Massive trade in feedstuffs would seem to be a highly inefficient system. This will sometimes involve transport from other continents to Europe, conversion to meat and animal wastes in highly concentrated areas, where there is easy access by ship, train or road,

with high emission rates to air and groundwater, and export of meat and milk to other countries.

In Europe, the use of cereals, the major feedstuffs worldwide, was 3.7 kg kg^{-1} meat in 1994, while in developing countries this was 1.9 kg kg^{-1}, and globally 2.9 kg kg^{-1}. The use of starchy roots was about 1 kg kg^{-1} in 1994, and worldwide 0.6 kg kg^{-1}. In Europe, approximately 60% of total cereals available are used as animal feed, while worldwide this is 30%. Since the N-use efficiency in animal production systems is much lower than in crop production (12), a higher efficiency of the food production system as a whole can be achieved by reducing export-oriented livestock production. Such a change could be accompanied by a shift from the use of imported animal feed to on-farm production or animal feed purchased from local producers so as to close the N cycle on the farm or regional scale. Such changes may be forthcoming from WTO negotiations (see under "Current Policies").

Additional reductions may be achieved by shifting to a human diet with less animal protein, and by reduction of waste of food at retail and household levels, particularly in industrialized countries. However, use of food and human diets are generally not the subject of policy formulation, but can be influenced through strategies such as marketing and public awareness.

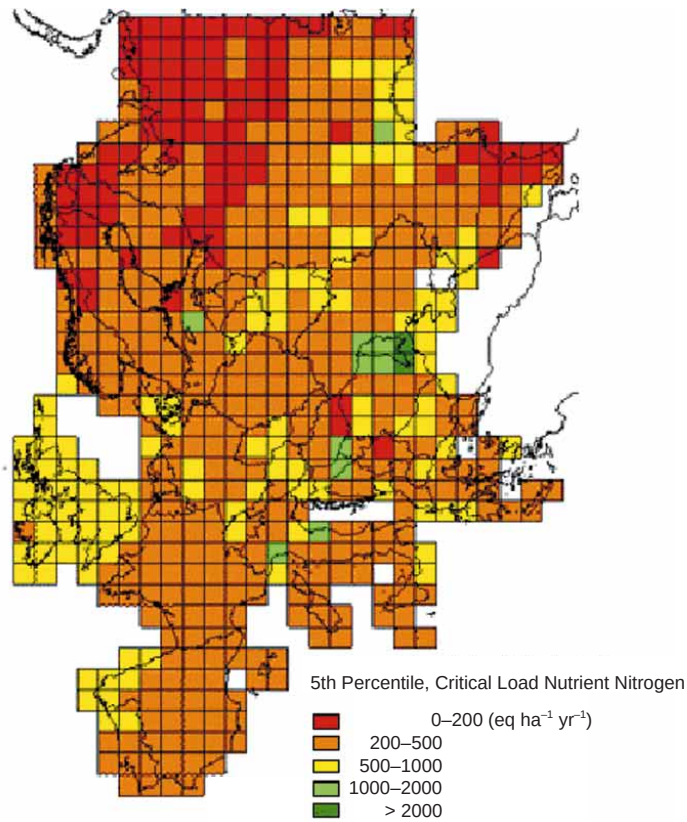
CURRENT POLICIES

Most of the policies in Europe are intended to decrease human and plant exposure to S and N pollutants and ecosystem loads. Countries agreed on reductions of air pollutants by signing different protocols under the Convention on Long-Range Transboundary Air Pollution (CLRTAP) (26). Table 4 gives an overview of different targets for the reactive N compounds. The last protocol, the Gothenburg Protocol, was unique in the sense that it came up with reduction of 4 pollutants (SO_2, NO_x, VOCs and NH_3) to abate 3 effects: *i*) acidification; *ii*) eutrophication; and *iii*) the effects of tropospheric ozone on human health and vegetation.

This Protocol, which at present has been signed by 29 European countries together with the United States and Canada, was based on a gap-closure method aiming to decrease the spatial exceedance of critical loads and levels in the most cost-efficient way. The critical loads are estimated by the countries themselves and may form a useful tool for determining emission ceilings in an integrated way and for regulating N inputs from fertilizers and imports (23).

An example of a map with critical loads is given in Figure 4, showing critical loads of N deposition (in acidifying equivalents) on natural ecosystems. A handbook for calculating critical loads was published by the German Federal Environmental Agency (27). Further references can be found on the website of the Coordinating Center for Effects of the UN-ECE, which is located RIVM in The Netherlands (www.rivm.nl/cce). The methodology described is also used to determine critical levels of air pollutants for humans and specific plant species.

Figure 4. 5%-critical load for N. Source: CCE (22).



Apart from the protocols, the EU member states have to adhere to the different directives. In May 1999, the European Commission presented a proposal for a Directive on National Emission Ceilings (NECD) for the same pollutants as CLRTAP and, for the first time, for ammonia (28). The proposed directive uses an approach similar to the Second Sulfur Protocol but extends it to include reduction in exceedance of critical limit values for ozone for human health and ecosystems. The approach is the same as used for the Gothenburg Protocol. The targets in the NECD proposal (Table 4) are much stricter than those in the existing EU agreements. For many countries, NECD targets are expected to be lower than those agreed on in the UNECE-CLRTAP Protocol.

The N input to the North Sea is subject to the North Sea Action Plan and the OSPAR convention. Reductions of up to 50% have been agreed on by Rhine and North Sea countries (2000 compared to 1985). Furthermore, the EU Nitrate Directive regulates the protection of waters by means of a ceiling for the N application rate of 170 kg N ha^{-1} (50% of current rates in intensive systems).

The recent European Framework Directive on Water includes all ecologically based former directives in an attempt to protect

Table 4. Nitrogen air emission reduction targets for the EU (29).

Policy/Pollutant	Base year	Target year	Reduction (%)
UNECE-CLRTAP			
NO_x[a]	1987	1994	Stabilization
NO_x[b]	1990	2010	50
NH_3[b]	1990	2010	12
5th Environmental Action Plan			
NO_2	1990	2000	30
Directive on National Emission Ceilings (NECD), proposed targets[c]			
NO_2	1990	2010	55
NH_3	1990	2010	21

[a] Targets from the first NO_x Protocol. Targets are the same for individual member states and for the EU as a whole.
[b] Targets from the Gothenburg Protocol (1 December 1999). The reduction target for the EU corresponds to the various emission ceilings of the individual member states since the EU was not formally a signatory to this protocol.
[c] Targets from the European Commission 1999 proposal for National Emission Ceilings Directive (NECD) (28). The emission reduction target for the EU corresponds to the various emission ceilings of the individual member states.

aquatic ecosystems (including coastal waters) and groundwater resources, and to regulate diffuse sources (e.g. agriculture) within river basins. Nutrients receive special attention in this directive. A distinction is made between types of water (rivers, lakes, transitional and coastal waters, and groundwater). For each type reference conditions are defined and a classification system is used with 5 categories (varying from poor to reference). High ecological quality and chemical quality will be the targets to be achieved in 15 years. A "good" ecological situation means at least no occurrence of significant anthropogenic eutrophication. Good chemical conditions correspond to a N concentration close to the reference value.

In the short term, the European policies which affect N inputs in agricultural production systems will probably be much more important for reduction of reactive-N than the overall agricultural policies. For example, current WTO agreements have not led to major changes in international trade. However, WTO negotiations on further liberalization may affect the European N cycle. Liberalization (gradual reduction of price-protective measures) leads to income losses, and will be accompanied by so-called cross-compliance, i.e. direct compensation of farmers to stimulate actions in the area of environment, nature and landscape rehabilitation. How such policies will affect the N flows in the medium term is, however, not certain.

CONCLUSIONS

Increasing N-fertilizer production and use, import of N in various products and energy consumption are the principal causes of the acceleration of the N cycle in Europe. Leaks of reactive N species (air emissions of NH_3 and NO_y as well as NO_3^- losses from soils to groundwater and surface waters) occurring in various points in the N cycle have adverse effects on human health in large cities, and on terrestrial and aquatic (fresh and saline) ecosystems. European policy measures can best focus on the 3 major driving forces and the main leaks from the economic system to the environment. Current policies generally focus on only one specific N leak. When considering specific measures to reduce leaks, possible consequences further on in the N cascade have to be taken into account so as to develop effective integrated policies. Integrated plans of action that focus on the main driving forces and leaks are needed. Critical loads may be useful tools in determining N-emission ceilings and developing integrated policies for regulating N flows such as fertilizer use and imports.

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