

# Technical summary

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## Part I Nitrogen in Europe: the present position

### Nitrogen inheritance

1. Gaseous di-nitrogen ( $N_2$ ) constitutes 78% of the earth's atmosphere. It is a rather inert chemical, being nearly unavailable for the biological cycle. The other nitrogen forms are much more reactive; these include nitrate ( $NO_3^-$ ), ammonium ( $NH_4^+$ ) and ammonia ( $NH_3$ ), gaseous nitrogen oxides ( $NO_x$ ), nitrous oxide ( $N_2O$ ) and many other inorganic and organic nitrogen forms. Collectively, they are termed 'reactive nitrogen' ( $N_r$ ). They are normally scarce in natural environments, with their low availability limiting the productivity of natural ecosystems. This was also the case for agricultural production before 1900, which long remained dependent on the recycling of  $N_r$  in human waste and manure, and the capacity of legumes to fix atmospheric  $N_2$  biologically.

2. With a growing human population through the nineteenth century and the need for more  $N_r$ , Europe increasingly operated a 'fossil nitrogen economy', dependent on the addition of nitrogen fertilizers from mined sources, including from guano, coal and saltpetre. The 'nitrogen problem' of the time was that these sources were fast becoming insufficient to meet Europe's escalating need for fertilizer  $N_r$ , and its military need for  $N_r$  in explosives [1.1].<sup>1</sup>

3. The situation changed dramatically shortly after 1900, with the invention of the Haber–Bosch process. This allowed the cheap industrial production of ammonia from di-nitrogen and hydrogen, permitting mass production of synthetic  $N_r$  fertilizers. By the 1930s, the European shortage of  $N_r$  had become a problem of the past, with  $N_r$  use in agriculture strongly increasing from the 1950s [1.1, 2.2].

4. The deliberate production and release of  $N_r$  in the Haber–Bosch process can be considered as perhaps the greatest single experiment in global geo-engineering that humans have ever made [1.1]. In Europe, human production of  $N_r$  fully met its

objectives to underpinning food and military security, while supplying a vital feedstock for many industrial processes [3.2, 3.5]. What was not anticipated was that this experiment would lead to a 'nitrogen inheritance' of unintended consequences [1.1], with  $N_r$  leaking into the environment in multiple forms, causing an even larger number of environmental effects [1.1, 2.6]. Simultaneously, the increasing extent of fossil fuel combustion for transportation and electricity production has led to a massive unintentional additional release of  $N_r$  into the atmosphere, mainly as nitrogen oxides ( $NO_x$ ) [2.4].

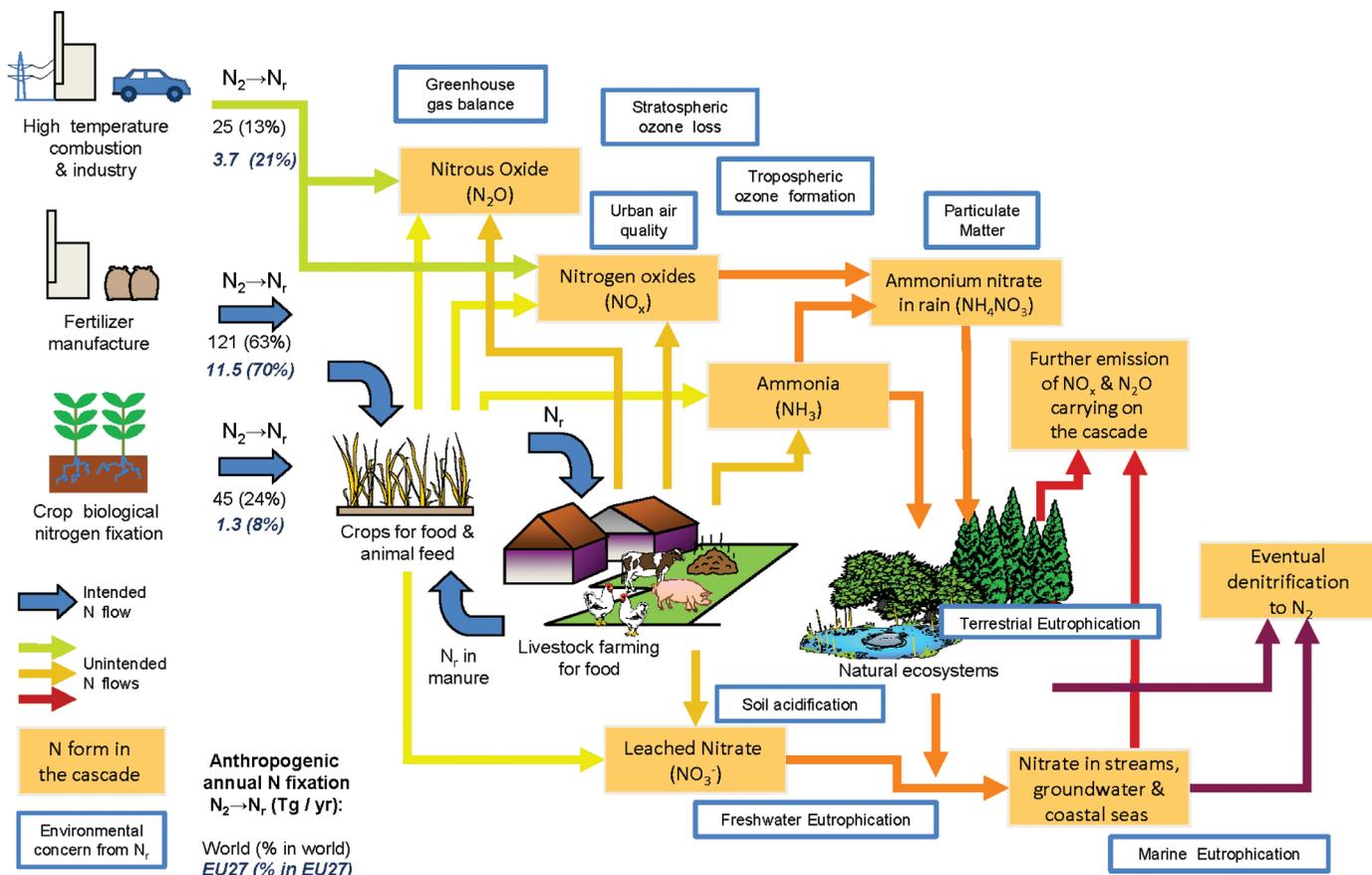
5. At the global scale, together with crop biological nitrogen fixation, these processes have altered the nitrogen cycle to an unprecedented extent, and much more than that of carbon or phosphorus. Humans introduce more  $N_r$  into the biosphere than all natural processes together [2.5, 13.2, 16.4]. Europe (EU-27) is a hot spot in this sense, producing 10% of global anthropogenic  $N_r$ , even though its surface covers less than 3% of the total world continental area [2.5, 13.2].

### Benefits, threats and current policies

6. The value of the benefits brought to the European economy by the production of fertilizers and the combustion of fuel is substantial. For example, the economic benefit of applying  $N_r$  fertilizers to wheat in the EU is estimated at around €8 billion per year. Accounting for benefits to other crops, livestock production, downstream food processing and many industrial benefits (including mining and chemical synthesis), the total benefits of  $N_r$  production will be very much larger [3.6]. By contrast, the formation of  $NO_x$  through high temperature combustion processes has no economic benefit, as control efforts focus on denitrifying  $N_r$  rather than its use [5.1].

7. Against these benefits must be listed the many effects on the environment and human health, as a result of  $N_r$  pollution. Several of these threats have been addressed by governmental policy measures related to abating atmospheric pollution or limiting nitrogen contamination of groundwater and surface water resources. These include several European directives, such as the Nitrates Directive, Water Framework Directive, Groundwater Directive, Ambient Air Quality Directive, National Emissions Ceilings Directive, Urban Waste Water Treatment Directive,

<sup>1</sup> References in this summary refer (e.g., [1.1, 2.2]) to chapter and section numbers of the *European Nitrogen Assessment*.



**Figure TS.1** Simplified view of the nitrogen cascade, highlighting the major anthropogenic sources of reactive nitrogen ( $N_r$ ) from atmospheric di-nitrogen ( $N_2$ ), the main pollutant forms of  $N_r$  (orange boxes) and nine main environmental concerns (blue boxes). Estimates of anthropogenic N fixation for the world (Tg /yr for 2005, in black) are compared with estimates for Europe (Tg /yr for 2000, in blue italic). Blue arrows represent intended anthropogenic N flows; all the other arrows are unintended flows [5.2].

Marine Strategy Framework Directive, Integrated Pollution Prevention and Control (IPPC) and the Habitats Directive [4.4].

8. The policy responses also include European commitments to multi-lateral environmental agreements, including the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution, the UN Framework Convention on Climate Change, the UN Convention on Biological Diversity, and the Oslo and Paris, Helsinki and Barcelona Conventions for the protection of the North East Atlantic, the Baltic Sea and the Mediterranean Sea, respectively [4.3].

9. Review of the current policies highlights a tendency to address individual  $N_r$  species from specific source sectors (agriculture, traffic, industry), media (air, freshwater, marine), and for specific issues (climate, urban air pollution, biodiversity, water quality, etc.). Until now, there has been little focus on developing policies that recognize the full extent and complexity of the nitrogen cycle [4.3, 4.4].

10. Trends in environmental pollution show that significant progress has been made in reducing emissions of  $NO_x$  to air. These policies have benefited from the availability of measures targeted at few stakeholders (e.g. electricity generation companies, industry, vehicle manufacturers), but have nevertheless been offset by increases in overall transport use and energy consumption. In the same way, significant progress has been

made in reducing water pollution due to water treatment policies that have engaged water companies [4.6].

11. By contrast, from a European perspective,  $N_r$  pollution from agriculture has shown only modest reductions in response to policies over the past 20 years [4.6]. Particular challenges faced in controlling  $N_r$  losses from agriculture are the many (often small) stakeholders, a highly diverse open system, and the perception of difficulty in passing anticipated costs to consumers [4.6]. In addition, the low price of  $N_r$  fertilizer, combined with its clear benefits to agricultural production, does not provide a strong incentive for farmers to use less than the (private) economic optimum [3.3].

### Nitrogen cascade and the need for integration

12. The lack of a holistic approach to developing nitrogen policies can partly be explained by the sectoral approach taken in government departments, and can also be linked to the tendency to scientific specialization. Nitrogen research communities have developed to become highly fragmented, providing a major challenge to integrate across all nitrogen forms, processes and scales [5.2].

13. The ‘nitrogen cascade’ (Figure TS.1) provides a useful concept to demonstrate the benefits of a holistic understanding

of the nitrogen cycle [2.6, 5.2]. Anthropogenic fixation of  $N_2$  to  $N_r$  raises the energy state of the nitrogen, with the energy being gradually dissipated as the  $N_r$  is converted through many different forms, until it is eventually denitrified back to  $N_2$ , the thermodynamically stable form of nitrogen in most environments. Even though  $N_r$  may be emitted from different processes and sectors, once released into the environment, the origin gradually becomes less relevant, causing multiple environmental effects. Each molecule of  $N_r$  emitted may cause several effects before eventual denitrification [2.6, 5.2].

14. The cascade concept highlights the potential for trade-offs and synergies in managing  $N_r$ . For example, if losses of  $N_r$  to one pathway are reduced, this can easily increase losses of another  $N_r$  form (e.g., some measures to reduce nitrate leaching may increase ammonia emissions and vice versa). Such trade-offs are sometimes termed ‘pollution swapping’. By contrast, because of the cascade, some measures may have benefits in reducing multiple forms and impacts of  $N_r$  [1.2].

15. It is concluded that a more holistic approach to managing the nitrogen cycle would have benefits to ensure more effective control of the different forms of  $N_r$  pollution and impacts. Such a development must be based on more-effective integration across environmental disciplines, providing the foundation to link traditionally separate policy domains [5.3].

## Approach of the European Nitrogen Assessment

16. Developing a more joined up approach to managing the nitrogen cycle necessarily proceeds gradually, and this has been encouraged through the European Nitrogen Assessment process. The overall goal of the Assessment was established as: *to review current scientific understanding of nitrogen sources, impacts and interactions across Europe, taking account of current policies and the economic costs and benefits, as a basis to inform the development of future policies at local to global scales* [1.4].

17. In developing this vision of gradual integration, based on analysis of the present position (Part I), the Assessment first examines the nitrogen turn-over processes in the biosphere (Part II), and then addresses nitrogen flows at different spatial scales (Part III).

18. One of the key conclusions of the first part of the Assessment is that the complexity of the nitrogen cycle needs to be distilled to highlight the priority concerns. This is important to limit the number of interactions when developing integrated approaches [5.4].

19. Recognizing these issues, the Assessment process established a comprehensive list of around 20 problems related to nitrogen. The list was first distilled down to nine ‘main environmental concerns’, setting the agenda for the Nitrogen in Europe (NinE) programme, as shown in Figure TS.1 [5.4].

20. In a second stage, the list was reduced to five ‘key societal threats’ of excess nitrogen, identified as: Water quality, Air quality, Greenhouse balance, Ecosystems and biodiversity, and Soil quality. These five threats provide a framework that automatically includes many of the other issues, balancing the complexity of the nitrogen cycle with the need for simplification [5.4].

21. The short-listing of the five key threats also provides a useful tool to communicate the nitrogen challenge to society. Together the five threats make an acronym as the ‘WAGES’ of excess nitrogen, while they can be also envisaged in direct analogy to the ‘elements’ of classical Greek cosmology (Figure TS.2) [5.4].

22. The Assessment applies the framework of five key societal threats to summarize the scale of the nitrogen challenge facing Europe (Part IV). Finally, the threats are brought together to examine the future perspective for European nitrogen policies (Part V). The Assessment used a network approach, where expert teams were formed for each chapter based on open invitations, including discussions of outlines during workshops held for each of the five parts (I–IV). The chapters take a variety of approaches across the Assessment, reflective of variation in data availability (e.g., limited data for some parts of Europe) and the nature of the issues being assessed. Draft chapters were subjected to internal and external peer review before being finalized.

## Part II Nitrogen processing in the biosphere

23. In recent decades substantial advances have been achieved in our understanding of the processes that govern nitrogen cycling in terrestrial environments (including natural and agricultural ecosystems), in aquatic environments (including freshwater, estuarine and marine ecosystems) and in the atmosphere. Each of these environments has been considered, integrating the processes of all relevant nitrogen forms.

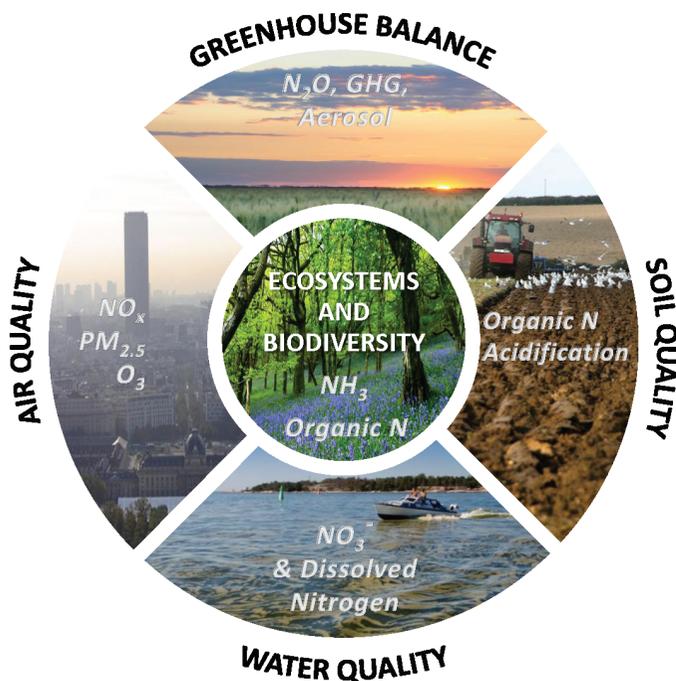
### Reactive nitrogen turnover in terrestrial ecosystems

24. The understanding of N cycling in terrestrial ecosystems has undergone a paradigm shift since 1990. Until then, the perception was that: (1)  $N_r$  mineralization is the limiting step in N cycling; (2) plants only take up inorganic  $N_r$ ; and (3) plants compete poorly for  $N_r$  against microbes and use only the  $N_r$  which is ‘left over’ by microbes. Since then studies have shown that plants compete effectively for  $N_r$  with microorganisms and take up organic N in a broad range of ecosystems [6.4].

25. On the ecosystem scale, soils are the main reservoir for  $N_r$ . This is more pronounced for agricultural systems than for forest systems, with more than 90%–95% of  $N_r$  being stored in the soil. Nitrogen stocks of managed systems are typically depleted and with retention processes negatively affected [6.2, 6.4].

26. In cereal farming, the use of only mineral  $N_r$  fertilizers, instead of animal manures or composts, as well as the simplification of the crop rotation scheme that this has made possible, has in some cases resulted in a decline of soil organic matter. In the long-term this practice of using only mineral fertilizers has decreased the buffer capacity of the soil towards inorganic N inputs, thus increasing its propensity to  $N_r$  leaching [6.4].

27. Nitrogen fixation in non-agricultural legumes or in other N-fixing organisms remains difficult to quantify, hampering a



**Figure TS.2** Summary of the five key societal threats of excess reactive nitrogen, drawn in analogy to the 'elements' of classical Greek cosmology. The main chemical forms associated with each threat are shown [5.4].  
Photo sources: Shutterstock.com and garysmithphotography.co.uk.

better understanding of the importance of biological  $N_2$  fixation for most terrestrial ecosystems [6.3].

28. Nitrogen-enriched terrestrial ecosystems lose significant amounts of N via nitrate leaching and gaseous emissions ( $N_2$ ,  $N_2O$ , NO,  $NH_3$ ) to the environment. Estimates of denitrification to  $N_2$  remain highly uncertain, due to difficulties in measurement and a high degree of temporal and spatial variability. There remain substantial uncertainties in the average fraction of  $N_r$  applied to fields that is emitted as  $N_2O$ , ranging from 1% to 3.5%–4.5% of fertilizer N applied, using bottom-up and top-down estimates, respectively. Further research is needed to better understand the relative contribution of direct and indirect  $N_2O$  emissions [6.5].

29. In forests, the C:N ratio of the forest leaf litter or top mineral soil is a good indicator of  $N_r$  status related to nitrate leaching. At C:N above 25, mineral  $N_r$  is usually retained, whereas below 25, nitrate leaching increases with increasing  $N_r$  deposition [6.5] (Table TS.1).

## Reactive nitrogen turnover and transfer along the aquatic continuum

30. Major sources of  $N_r$  in the aquatic environment include households and sewage discharges together with diffuse pollution losses from agricultural practices.

31. Nitrate retention by riparian wetlands is a frequent justification for conservation and restoration policies of these systems. However, their use for mitigating  $NO_3$  contamination of river systems must be treated with caution, since their effectiveness is difficult to predict, and side effects observed include

increased dissolved organic matter and  $N_2O$  emissions, together with loss of biodiversity [7.5].

32. Release of dissolved organic nitrogen has often been neglected, while it can play a significant role, particularly in upland semi-natural catchments, but is not determined in most routine European water quality monitoring programmes [7.3].

33. The effects of increased  $N_r$  loadings to aquatic environments include acidification and loss of biodiversity in semi-natural environments, and eutrophication in more disturbed ecosystems. Standing waters are particularly sensitive to both acidification and eutrophication, since the longer residence time in these systems leads to greater interaction between the biota and changing water chemistry [7.4].

34. The richest submerged plant communities in lakes have been observed to be associated with winter nitrate concentrations not exceeding 2 mg N/l, and this has been proposed as an appropriate target concentration for enriched shallow European lakes to reach 'good ecological status' [7.5].

35. Although phosphorus is often the main limiting element controlling primary production in freshwater systems,  $N_r$  has been reported as a factor limiting or co-limiting biological production in some eutrophicated lakes, and control of both  $N_r$  and P loading is needed in impacted areas, if ecological quality is to be restored [7.4].

36. The importance of storage and denitrification in aquifers is a major uncertainty in the global N cycle, and controls in part the response of catchments to land use or management changes. In some aquifers, the increase of N concentrations will continue for decades even if efficient mitigation measures are implemented now [7.5].

37. Nitrogen inputs from human activities have led to ecological deterioration in large parts of the coastal oceans along European coastlines, including harmful algal blooms and anoxia. The riverine  $N_r$ -loads are the most pronounced  $N_r$  source to coasts and estuaries, while atmospheric  $N_r$  deposition and  $N_2$  fixation also contribute significantly [8.8].

38. A large imbalance of  $N_r$  with respect to silica inputs causes the development of severe harmful algal blooms. Especially affected by eutrophication are the major European estuaries (e.g., Rhine, Scheldt, Danube and the coastlines receiving their outflow), North Sea, Baltic Sea, and Black Sea, as well as some parts of the Mediterranean coastline [8.10].

39. Marine biodiversity is reduced under high nutrient loadings, affecting nutrient recycling negatively. Recovery of communities may not be possible if eutrophication and anoxia persist for long time periods of several years [8.9].

40. The European coastal zone plays a major role in denitrification of  $N_r$  to  $N_2$ . Export of  $N_r$  to the sea is estimated at 4.5 TgN per year, most of which will be denitrified to  $N_2$ . Globally, coastal denitrification is estimated at 61 TgN per year, including 8 Tg per year in estuaries. Comparison with independent estimates of estuarine and sediment denitrification implies that coastal systems impart 54–197 TgN per year from the open ocean globally, compensating the  $N_r$  losses due to sediment denitrification [8.8].

**Table TS.1** Characteristics of coniferous forest ecosystems with low, intermediate and high N status, as grouped according to total N<sub>r</sub> input [6.5]

Nitrogen status	Low N status (N-limited)	Intermediate	High N status (N-saturated)
Input (kg N per ha per yr)	0–15	15–40	40–100
Needle N% (in spruce)	< 1.4	1.4–1.7	1.7–2.5
C:N ratio (g C per g N)	> 30	25–30	< 25
Soil N flux density proxy (litterfall + throughfall) (kg N per ha per yr)	< 60	60–80	> 80
Proportion of input leached (%)	< 10	0–60	30–100

## Reactive nitrogen turnover in the atmosphere

41. The main N<sub>r</sub> compounds emitted to the atmosphere by anthropogenic activities are NH<sub>3</sub> (~3.2 Tg N per year in EU-27), mainly from agriculture, and NO<sub>x</sub> including both NO and NO<sub>2</sub> (~3.5 Tg N per year). European NO<sub>x</sub> emissions arise mainly from transport (50%), power generation (25%) and other combustion sources (21%). Emissions of nitrous oxide (N<sub>2</sub>O) in Europe are much smaller (1 Tg N per year), mainly arising from soils, especially in agriculture [9.2, 16.3].

42. These N<sub>r</sub> forms have different fates in the atmosphere. The atmospheric chemistry of NH<sub>3</sub> is well known, undergoing irreversible reaction with sulphuric acid (H<sub>2</sub>SO<sub>4</sub>) and reversible reaction with nitric acid (HNO<sub>3</sub>) and hydrochloric acid (HCl). Previous assumptions of instantaneous equilibrium in the atmosphere are now believed to be incorrect due to kinetic constraints. Together with the effects of mixed aerosol chemistry (e.g., organic layers), these effects are not yet fully parametrized in models [9.3].

43. Very little is known either quantitatively or qualitatively about atmospheric organic nitrogen compounds, although they can contribute up to half of wet-deposited N<sub>r</sub>. Sources may include emissions of amines, amides, urea, amino acids. These represent an extra contribution to N<sub>r</sub> deposition and eutrophication, missing from current estimates [9.3].

44. The biosphere–atmosphere exchange of NH<sub>3</sub> and NO<sub>x</sub> is dependent on a combination of surface (canopy, soil, management) and environmental conditions. In the case of NH<sub>3</sub>, the environmental dynamics of emissions need to be better quantified, given the expectation that climate change may increase future emissions. Reductions in SO<sub>2</sub> are reducing rates of NH<sub>3</sub> dry deposition, tending to increase its atmospheric lifetime [9.4].

45. The potential for ‘pollution swapping’ has been illustrated by the effect of implementing policies to reduce nitrate leaching. By prohibiting winter manure application to fields, more manure is spread in spring, leading to a new peak in observed NH<sub>3</sub> concentrations [9.2].

46. Ammonia has substantial impact near its sources of emission due to high dry deposition rates to natural ecosystems, so that it may significantly affect natural ecosystems in agricultural areas. By contrast, NO<sub>x</sub> has little impact close to the sources, due to low dry deposition rates, until it is converted into nitric acid (about 5% per hour). Long-range transport of both components occurs in the form of aerosol phase compounds,

which are transported more than 1000 km. Abatement strategies need to take account of these differences when assessing the impact of N<sub>r</sub> deposition on sensitive ecosystems [9.4, 9.6].

## Part III Nitrogen flows and fate at multiple spatial scales

47. In managing our environment, humans have created complex ecosystem mosaics, the structure of which largely determines the flows and fate of N<sub>r</sub> from local to European scales.

### Nitrogen flows at farm and landscape scales

48. Farms represent the operational units at which local decisions on the use of nitrogen fertilizer are taken in accordance with policies to encourage food production and sustain farm incomes [10.1].

49. The basis of good nitrogen management in agriculture is to increase the temporal and spatial coincidence between N<sub>r</sub> availability in the soil and N<sub>r</sub> uptake by crop, thus increasing the N use efficiency (ratio of N<sub>r</sub> produced in final agricultural goods to N<sub>r</sub> introduced as fertilizer) and minimizing N<sub>r</sub> flows into water and the atmosphere [10.2].

50. Current management drivers often cause farms to be ‘open’ with substantial N losses, as the objective of high production often surpasses that of minimizing the emissions to the environment. Livestock farming presents particular problems with large potential losses associated with the management of manure. Animal excrements, historically the major fertilization resource for cropland, have often acquired the status of wastes to be disposed. The geographical dissociation of crop and livestock farming increases these problems [10.3].

51. Farm nitrogen budgets typical of north west Europe highlight the main N loss pathways according to farm type. Although there is a great variation in farm type, illustrative budgets show how the N losses per unit area from beef and organic dairy farms are typically lower than from more intensively managed pig and dairy farms. By contrast, the total N loss per unit N in products is highest for beef, dairy and organic dairy farms (2.7, 2.55, 1.92, respectively) as compared with pork farms (0.8) [10.4].

52. The fate and environmental effects of N<sub>r</sub> losses from agricultural systems are strongly dependent on the structure of the surrounding terrestrial and hydrological landscape. For

this purpose, the landscape is considered as that scale in which adjacent farms, fields, forests, nature areas and water courses interact, as affected by  $N_r$  dispersion through air and water, and by human transfers [11.1].

53. Landscape features, like the patchiness of land uses or presence of woods close to emission sources, can play a significant role in the fate of emitted ammonia, minimizing net emissions and helping to avoid longer distance transport. At the same time, such landscape features can affect the nature of surface runoff and deep infiltration, while the presence of active wetlands determine the occurrence of denitrification of leached nitrate, with possible secondary  $N_2O$  emissions [11.3]. These interactions highlight the potential to exploit spatial relationships at the landscape scale to minimize  $N_r$  losses and their environmental effects [11.6].

54. Integrated models are being established to assess the overall effect of landscape-scale decisions in relation to the local spatial distribution of multiple  $N_r$  effects (e.g., on biodiversity, greenhouse balance, water quality). The verification of spatially explicit landscape-scale models provides the basis to develop 'landscape planning' approaches to minimize  $N_r$  threats [11.6]. When designing and implementing environmental measures, greater attention should be given to the landscape scale in order to take into account local dispersion and buffering processes so as to maximize the efficiency of the measures.

## Nitrogen flows at the city scale

55. Cities, although they take only ~2% of land, dramatically affect the N cycle. Cities import  $N_r$  from rural areas the agricultural products needed to feed dense populations, then disperse the wastes resulting from its consumption: to surface water (2.3 Tg N per year in wastewater in the EU-27), to air (0.015 Tg N as  $NH_3$  and  $N_2O$  emissions from wastewater and solid waste treatment plants), and to soils (1.5 Tg N in sludge and solid wastes) [12.1, 16.4].

56. Because of their dense transportation networks, industrial facilities and energy production infrastructures, cities are the main sites of the 3.5 Tg N per year  $NO_x$  emission to the atmosphere through the burning of fossil and other fuels [12.3, 16.4].

57. A case study for Paris highlights the amplification of the N cycle from 1800. Today, the major part of  $N_r$  output is attributed to fuel combustion for transport and energy (50 Gg  $N_r$  per year mainly as  $NO_x$ ). Sewage water treatment plant denitrify 32 Gg  $N_r$  to  $N_2$ , with 12 Gg  $N_r$  released to water courses, and only 12 Gg  $N_r$  returned to soils [12.3].

58. By comparison to the present situation, late nineteenth century Paris recycled 50% of the  $N_r$  in solid/liquid waste for use in fertilizers, a system which came to the end with the advent of dilute sewage streams (e.g., the flushing toilet) and the Haber-Bosch process. The illustration highlights the potential for future sewage processing systems to process  $N_r$  (and phosphorus) for re-use as fertilizer, rather than wasting the  $N_r$  resource through denitrification [12.3, 23.5].

## Regional scale nitrogen flows through watersheds and the atmosphere

59. Many large watersheds such as the Danube, Rhine and the Scheldt cover several countries and, together with coastal water transfers and atmospheric dispersion, allow substantial trans-boundary transport of  $N_r$  pollution across Europe.

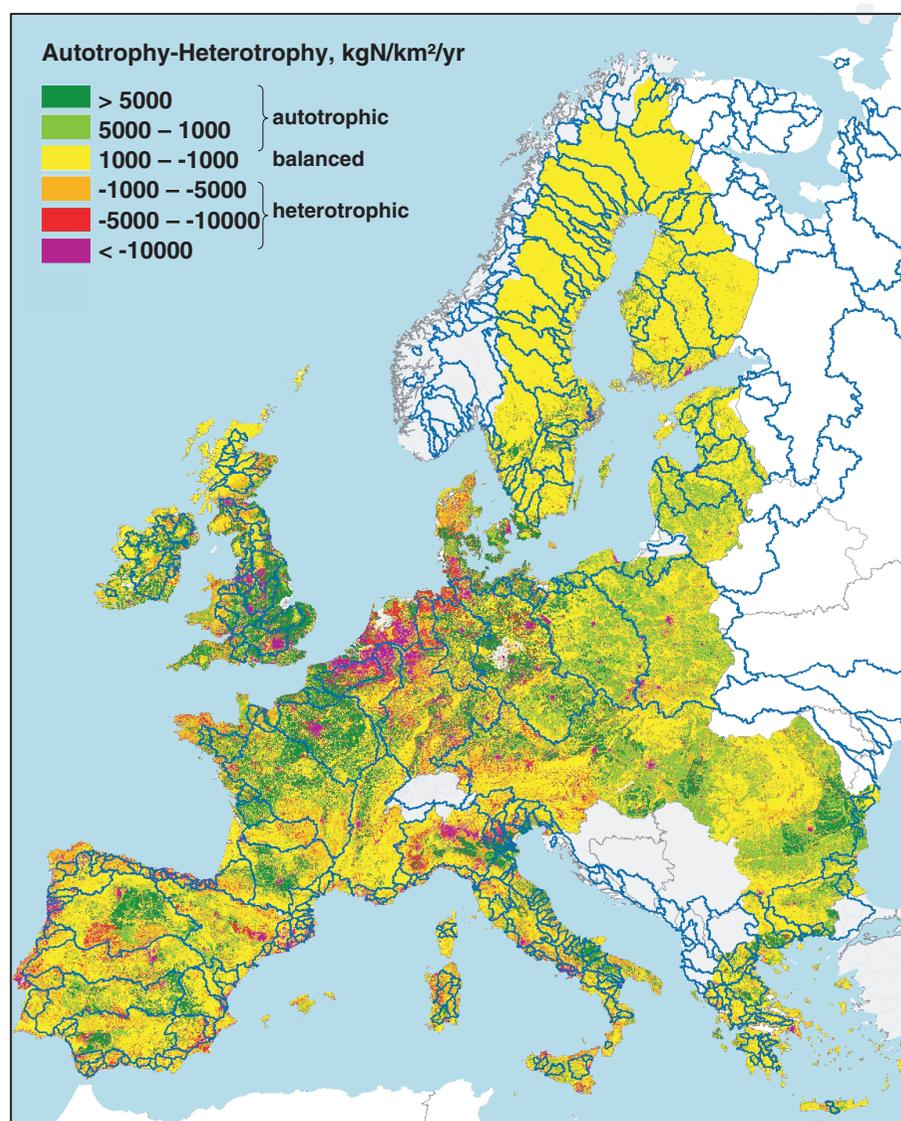
60. Regional watersheds represent territorial units for water resources management in relationship with both agricultural and urban activities. The strong regional specialization of agriculture and the concentration of urban habitat in Europe result in some basins exporting large amounts of  $N_r$  as food and feed (autotrophic basins), while others import them (heterotrophic basins). The difference between the two (i.e., net autotrophy) is illustrated in Figure TS.3. This map highlights the imbalance of the regional nitrogen cycle between watersheds, some of which import much more nitrogen (as food and animal feed) than their crop production can use [13.2].

61. Throughout Europe, net anthropogenic  $N_r$  inputs (fertilizer application, food and feed import, crop  $N_2$  fixation and atmospheric deposition) represent 3700 kg  $N_r$  per  $km^2$  annually (watershed ranges 0–8400 kg per  $km^2$ ), which is five times the rate of natural  $N_2$  fixation [13.2]. Of the anthropogenic  $N_r$  input, ~80% is stored (in soils, sediments or groundwater) or lost to the atmosphere along the drainage network as  $N_r$  or  $N_2$ . Only ~20% reaches the basin outlet and the marine coastal zones, at rates four times background. In such coastal areas with limited silica availability,  $N_r$  causes harmful algal blooms [8.9, 13.6].

62. The chain of processes leading to atmospheric emissions of  $N_r$ , atmospheric dispersion, chemical transformation and deposition is extremely complex and currently observations only address part of this chain. More observations are needed especially of gaseous nitric acid ( $HNO_3$ ), ammonia ( $NH_3$ ) and coarse nitrate aerosol concentrations. Concentrations of all compounds should be measured at the same site if the mass-balance of  $N_r$  is to be assessed, pointing to the need for integrated site measurements in monitoring networks [14.4].

63. Atmospheric models are routinely used to quantify the spatial patterns and trans-boundary fluxes of  $N_r$  across Europe, including inputs to coastal seas. Differences among European models can be 30% in some areas, and substantially more for specific locations. The major uncertainties indicate the need for further information on atmosphere-biosphere fluxes of  $N_r$  with sensitive ecosystems, dry deposition of particles, sub-grid fluxes of  $NH_x$  compounds, and effects of topography on wet-deposition. A balanced program of observations and models is critical to future understanding of atmospheric transport of  $N_r$  at local to global scales [14.6].

64. Considering the  $N_r$  inputs to the environment from multiple sources, and the inter-related flows between ecosystems, watersheds and the atmosphere, there is an obvious need for closer cooperation between existing monitoring activities. Strengthened monitoring programs and data integration are needed at the international level, which should improve harmonization between terrestrial, aquatic and atmospheric mon-



**Figure TS.3** Difference between nitrogen autotrophic and heterotrophic contributions for territories in the EU-27. Autotrophy is defined as the amount of nitrogen in crop and grass produced by agriculture. Heterotrophy is the amount of nitrogen feed ingested by livestock and food consumed by humans. The map shows overall net autotrophy (positive values) and net heterotrophy (negative values) [synthesis from 13.2 and 16.4].

itoring communities (terminology, methodology, units) and the exchange of information [synthesis of 13, 14].

## European scale nitrogen flows

65. European nitrogen flows and budgets have been estimated for terrestrial ecosystems (agriculture, forest and other ecosystems) and for all systems combined (also including urban, transport, industrial and aquatic flows).

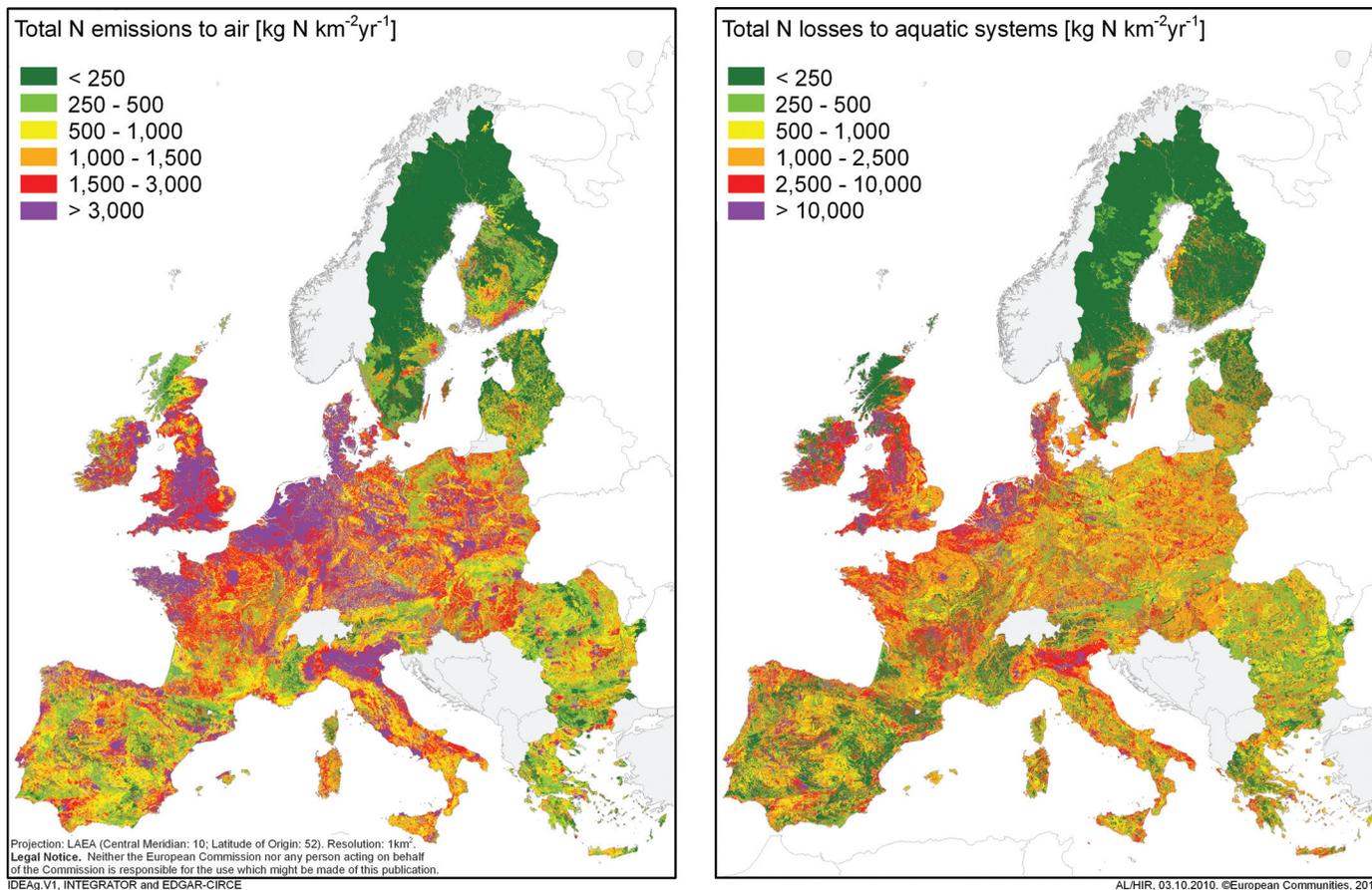
66. For the year 2000, a comparison of four mass balance models estimated  $N_r$  inputs to agriculture in the EU-27 at 23–26 Tg N, being mainly due to fertilizer and animal manure. For emissions from agriculture, the comparison showed similar relative estimates for  $NH_3$  (2.8–3.9 Tg N) and  $N_2O$  (0.35–0.46 Tg N), but diverging results for soil  $NO_x$  (0.02–0.20 Tg N). The largest absolute uncertainties were for  $NH_3$  [15.6].

67. Inputs of  $N_r$  to soils from both fertilizers and manure increased between 1970 and 2010 by ~20% for the EU-27. Although cattle numbers decreased, this trend was more than

offset by increased  $N_r$  excretion rates per cow (e.g., increased milk production per cow) [15.5]. These increases are consistent with overall estimated increases in  $NH_3$  and  $N_2$  emissions and  $N_r$  leaching by ~10% between 1970 and 2000, with emissions decreasing slightly since peak values around 1985. Emissions per unit agricultural area have increased by 20%–30% as a result of intensification over the period in western Europe [15.5].

68. The estimated distribution of overall  $N_r$  losses to the environment is shown in Figure TS.4. Emissions to the atmosphere reflect the distribution of livestock production (dominating  $NH_3$  emissions) and human population centres (dominating  $NO_x$  emissions) across Europe. The distribution of  $N_r$  inputs to aquatic systems is dominated by nitrate losses, which are largest in areas with high livestock density and precipitation excess, while more localized peaks are associated with urban waste-waters [16.3].

69. An overall budget of the present N cycle has been established (around 2000 for EU-27) and is summarized in



**Figure TS.4** Distribution of reactive nitrogen emissions across Europe ( $\text{kg N per km}^2$  for 2000) including emissions to air as  $\text{NO}_x$ ,  $\text{NH}_3$  and  $\text{N}_2\text{O}$ , and total losses to aquatic systems, including nitrate and other N, leaching and wastewaters [16.3].

Figure TS.5. The budget highlights the central role of crop production and livestock farming. The annual  $\text{N}_r$  brought to agricultural soils at 27.5 Tg N (consisting of 11.2 Tg N as synthetic fertilizers, 7.1 Tg N as manure, 2.4 Tg N as atmospheric deposition and 1.0 Tg N through biological nitrogen fixation and 5.8 Tg N as crop residues) is in surplus over the requirements of crop production (17.6 Tg N). The annual  $\text{N}_r$  surplus of 9.9 Tg contributes to substantial  $\text{N}_r$  leaching to surface and groundwater (6 Tg N), denitrification to  $\text{N}_2$  (4.5 Tg N), volatilization as  $\text{NH}_3$  (1.6 Tg N) and emission of  $\text{N}_2\text{O}$  and NO (0.5 Tg N). The overall balance for agricultural soils implies a small annual depletion of soil  $\text{N}_r$  stocks, although this term is considered to be very uncertain since it is calculated by difference of several uncertain estimates. It represents the regional average of a net loss of  $\text{N}_r$  in soils of autotrophy-dominated regions with arable farming, and a net gain of  $\text{N}_r$  in soils of heterotrophy-dominated regions with intensive livestock farming [16.4].

70. In order to provide an annual consumption of animal products by humans of 2.3 Tg N, livestock farming in Europe uses five times as much  $\text{N}_r$  from crops and imported feed (11.8 Tg), driving the overall European agricultural N cycle. By comparison, direct consumption by humans of crops grown in Europe represents only 2.0 Tg N per year. The handling of livestock excreta also leads to direct gaseous emissions of 1.5 Tg N per year. Overall, in order to produce 4.3 Tg N of food

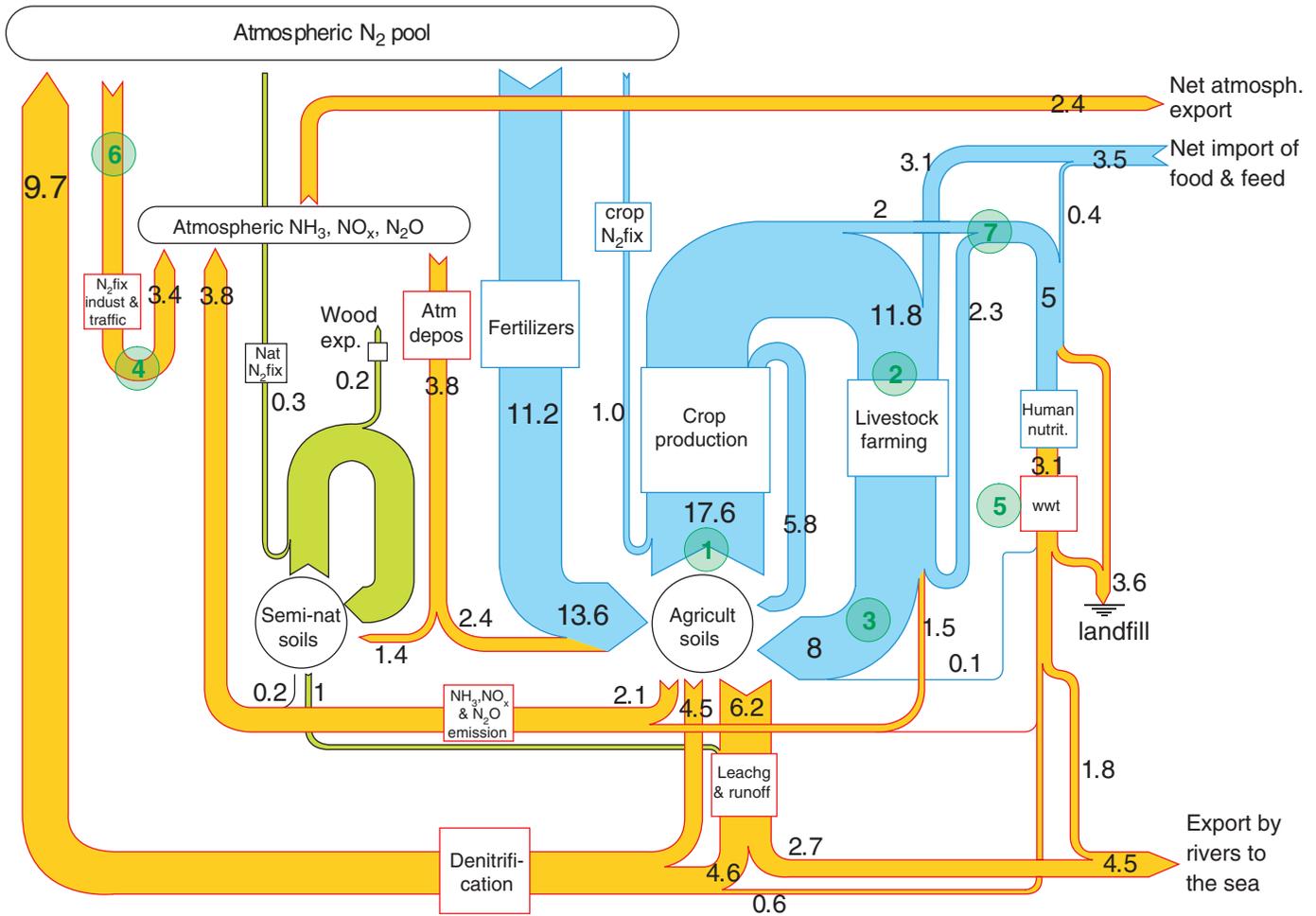
annually for the European population (not including 0.4 Tg N in imported food and feed), three times as much  $\text{N}_r$  is emitted to the environment, which corresponds to a nitrogen use efficiency of 30% (as compared with a global average of 50%) [16.4].

71. To these emissions should be added 3.7 Tg N per year of wastewater discharged to surface waters and 3.4 Tg N per year of  $\text{NO}_x$  emitted through fossil fuel combustion by the energy, industry and transportation sectors [9.2, 16.3].

72. As a whole, at around the year 2000, the EU-27 is a net commercial importer of animal feed and human food of 3.5 Tg N per year (mainly due to feed). Conversely, Europe is a net exporter of  $\text{N}_r$  to the environment: by atmospheric transport (2.4 Tg N per year) and by river export to marine systems (4.5 Tg N per yr). The largest single sink for  $\text{N}_r$  is denitrification to  $\text{N}_2$  in soils, river sediments and the sediments of European shelf regions, estimated at 9.3 Tg N per year, which is one of the most uncertain estimates [16.4, 8.8]. A net transfer from anthropogenic to natural systems is also implied, as estimated annual losses (0.7 Tg N) are less than half the estimated inputs (1.6 Tg N) mainly from atmospheric  $\text{N}_r$  deposition.

73. A similar budget has been reconstructed for the same territory (the present EU-27) in the beginning of the twentieth century (Figure TS.6). The estimates are necessarily less certain than for 2000, and the hypotheses leading to this

## Europe (EU27), around 2000. N fluxes in TgN/yr



**Figure TS.5** The N cycle at the scale of EU-27 [simplified from 16.4] for the year 2000. Fluxes in green refer to 'natural' fluxes (to some extent altered by atmospheric N<sub>2</sub> deposition), those in blue are intentional anthropogenic fluxes, those in orange are unintentional anthropogenic fluxes. The numbered green circles indicate a package of seven key actions for overall integrated management of the European nitrogen cycle (see para. 111) [23.5].

retrospective reconstruction are described in the ENA supplementary material [16].

74. By comparison with present-day amounts, the use of mineral fertilizers around the year 1900 (mainly from Chilean salpetre, guano and coal) was very small. The primary source of new N<sub>r</sub> in agriculture was biological N fixation by legume fodder crops, typically grown once every three years in triennial rotations. The nitrogen fixed by legume crops was brought to arable soils by the incorporation of crop residues and application of animal manures. Losses of N<sub>r</sub> from agricultural soils, though already significant, were only one third of the current level (Figure TS.6).

75. Annual atmospheric deposition of N<sub>r</sub> was around 1.9 Tg N in 1900, roughly half of which was deposited each to agricultural and semi-natural land. This was roughly half of the atmospheric deposition in 2000 at 3.8 Tg N, of which only 37% is deposited to semi-natural land, reflecting the overall reduction in area of semi-natural land (Figure TS.6). As with 2000, the overall difference between N<sub>r</sub> removals and gains for

agricultural soils is not considered significant, while the average gains to semi-natural soils including forests (1.1 Tg N) were larger than the removals (0.6 Tg N), but less so than in 2000.

## Part IV Managing nitrogen in relation to key societal threats

76. For each of the five key societal threats of excess reactive nitrogen, the Assessment examined the scale of the concern, including, where possible, information on temporal trends and the progress made through any existing policy measures.

### Water quality

77. Anthropogenic increase of N<sub>r</sub> in water poses direct threats to humans and aquatic ecosystems. High nitrate concentrations in drinking water are considered dangerous for human health, as they might cause cancers and (albeit rarely) infant methaemoglobinaemia. There is also evidence for benefits of nitrate

Europe (EU27), around 1900. N fluxes in Tg N /yr

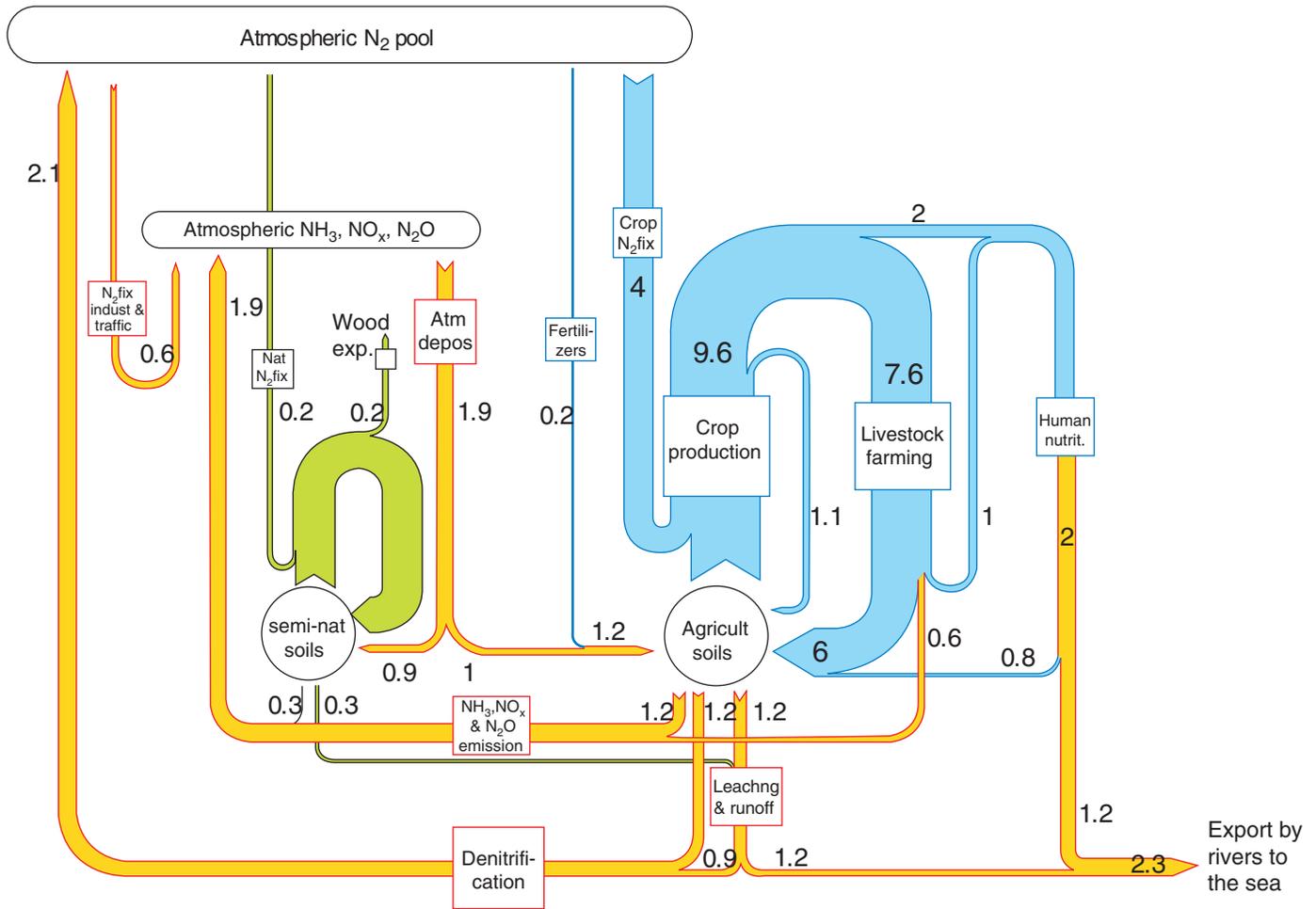


Figure TS.6 Reconstruction of the N cycle at the scale of current EU-27 for the year 1900 (16.4, supplementary material). Same colour code is used as in Figure TS.5.

for cardiovascular health and protection against infections. In aquatic ecosystems the  $N_r$  enrichment produces eutrophication, which is responsible for toxic algal blooms, water anoxia, fish kills and biodiversity loss [8.8, 17.3].

78. In addition to high  $N_r$  concentrations in European waterbodies, increasing nitrate in groundwaters threatens the long-term quality of the resource, as nitrate may have long residence time in the aquifers, and it can be expected that past fertilizer strategies will impact for many decades the quality of European groundwaters [7.5, 17.2].

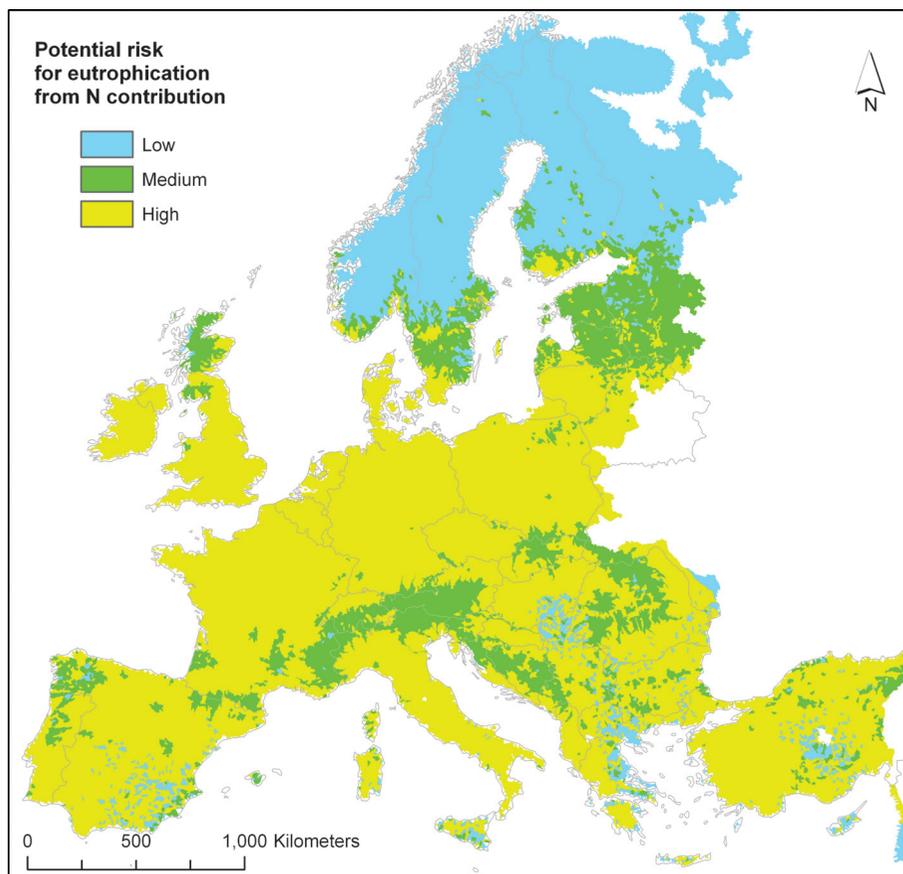
79. About 3% of the population in EU-15 using drinking water from groundwater resources is potentially exposed to concentrations exceeding the standard for drinking water of 50 mg  $NO_3/l$  (11.2 mg N/l), with 5% of the population is chronically exposed to concentrations exceeding 25 mg  $NO_3/l$  (5.6 mg N/l), which may double the risk of colon cancer for above median meat consumers [17.3].

80. A value of 1.5 mg N/l has been considered as the total  $N_r$  limit above which freshwater bodies may develop loss of

biodiversity and eutrophication. Except in Scandinavia and in mountainous regions, this level is already exceeded in most European freshwater bodies (Figure TS.7) [7.5, 17.3].

81. With  $N_r$  inputs to the coastal zone at four times the natural background (para. 61), large areas along Europe's coastline are suffering severe eutrophication problems, with anoxia and/or the proliferation of undesirable or toxic algae. Particularly affected are the south-eastern continental coast of the North Sea, the Baltic Sea (except the Gulf of Bothnia), the coasts of Brittany, the Adriatic Sea and the western coastal Black Sea [8.10, 13.7, 17.3].

82. Although eutrophication is decreasing, existing international policies have not been fully implemented, and even under favourable land use scenarios,  $N_r$  export to European waters and seas is anticipated to remain a problem in the near future. Achieving substantial progress at the European scale requires integration of sectoral policies, reducing overall inputs of  $N_r$  to watersheds, e.g., through changes in agriculture and other N flows [4.5, 13, 17.5].



**Figure TS.7** Indication of the nitrogen threat to water quality. Potential risk of eutrophication for surface freshwater based on estimated total  $N_r$  concentrations. The three classes of risk are: low,  $<0.5$  mg/l; medium 0.5–1.5 mg/l; high  $>1.5$  mg/l as total  $N_r$  concentration in water [17.3.3].

## Air quality

83. Emissions of  $NO_x$  and  $NH_3$  contribute to several negative effects on human health and ecosystems. In addition to effects of  $NO_2$ , secondary pollutants play key roles. These include ground level ozone ( $O_3$ ), formed photochemically in the presence of  $NO_2$  and volatile organic compounds (VOC), and inhalable particulate matter (PM), formed from oxidation of  $NO_2$  to  $HNO_3$  and reaction with  $NH_3$  to form ammonium nitrates.  $NO_x$ ,  $O_3$  and PM cause or aggravate asthma, reduced lung functions and bronchitis. Chronic exposure may increase the probability of respiratory or cardiovascular mortality and cancers [18.2].

84. Direct  $NO_2$  and ozone damage to vegetation has been recognized for a long time, as well as to materials, buildings and objects of cultural heritage. There is a difficulty of ascribing health effects to  $NO_2$  *per se* at ambient levels rather than considering  $NO_2$  as a surrogate for a traffic-derived air pollution mixture [18.2].

85. The role of particulate ammonium and nitrate in human health effects is still under discussion. Current approaches assume damage on a mass basis for PM with a median diameter less than  $2.5 \mu m$  ( $PM_{2.5}$ ).  $N_r$  compounds contribute up to 30–70% of  $PM_{2.5}$  mass in Europe [18.5]. Overall, models estimate a loss of statistical life expectancy due to PM of 6–12 months across most of central Europe (Figure TS.8). There has been a low success in controlling  $NH_3$  emissions in Europe

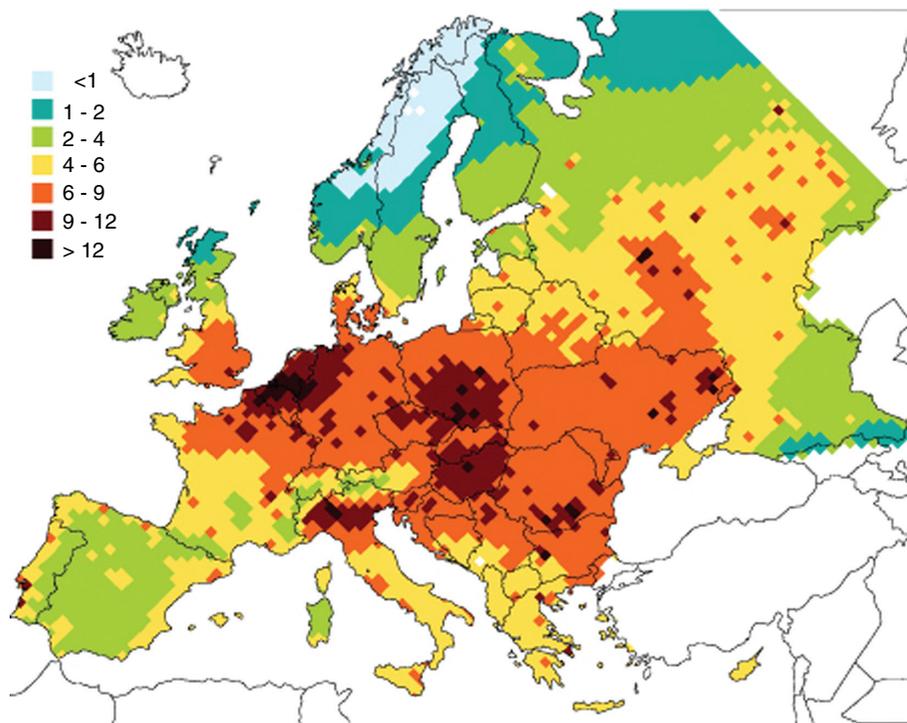
which needs to be further assessed, in particular in connection with the development of new agricultural policies [18.6, 4.5].

86. In the EU-27 countries, 60% of the population lives in areas (mainly urban) where the annual EU limit value of  $NO_2$  is exceeded. Levels have decreased since 1990, although the downward trends have been smaller or even disappeared after 2000. Although episodic  $O_3$  levels have decreased since 1990 due to VOC and  $NO_x$  control, continental background concentrations have increased, with  $O_3$  levels remaining a threat to human health and ecosystems [18.5, 4.5].

## Greenhouse balance

87. European anthropogenic  $N_r$  emissions have a complex effect on climate by altering global radiative forcing. They directly affect the greenhouse gas balance through  $N_2O$  emissions and indirectly affect it by increasing tropospheric  $O_3$  levels, altering methane ( $CH_4$ ) fluxes, and by altering biospheric  $CO_2$  sink (including atmospheric  $N_r$  deposition and  $O_3$  effects). Aerosol formed from  $NO_x$  and  $NH_3$  emissions also has a cooling effect [19].

88. A first assessment has been made of the overall effect (between 1750 and 2005) of European  $N_r$  emissions on radiative forcing. The main warming effects of European anthropogenic  $N_r$  emissions are estimated to be from  $N_2O$  (17 (15 – 19)  $mW/m^2$ ) and from the reduction in the biospheric  $CO_2$  sink by tropospheric  $O_3$  (4.4 (2.3 – 6.6)  $mW/m^2$ ). The main cooling



**Figure TS.8** Indication of the nitrogen threat to air quality. Across Europe  $N_r$  typically accounts for up to 0.3–0.7 of total particulate matter load with a median diameter less than  $2.5\ \mu\text{m}$  ( $\text{PM}_{2.5}$ ) on a mass basis [18.5]. Assuming that health effects are proportionate to mass [18.2.2] the map indicates the loss in statistical human life expectancy (in months) attributable to total  $\text{PM}_{2.5}$  [18.6, supplementary material].

effects are estimated to be from increasing the biospheric  $\text{CO}_2$  sink by atmospheric  $N_r$  deposition ( $-19$  ( $-30$  to  $-8$ )  $\text{mW}/\text{m}^2$ ) and by light scattering effects of  $N_r$  containing aerosol ( $-16.5$  ( $-27.5$  to  $-5.5$ )  $\text{mW}/\text{m}^2$ ) (Figure TS.9) [19.6].

89. Overall, European  $N_r$  emissions are estimated to have a net cooling effect, with the uncertainty bounds ranging from substantial cooling to a small net warming ( $-15.7$  ( $-46.7$  to  $+15.4$ )  $\text{mW}/\text{m}^2$ ) [19.6].

90. The largest uncertainties concern the aerosol and  $N_r$  fertilization effects, and the estimation of the European contributions within the global context. Published estimates suggest that the default  $\text{N}_2\text{O}$  emission factor of 1% used by the Intergovernmental Panel on Climate Change (IPCC) for indirect emissions from soils following  $N_r$  deposition is too low by at least a factor of two [6.6, 19.6].

91. Industrial production of  $N_r$  can be considered as having permitted increased livestock and human populations (and associated food, feed and fuel consumption). The expected substantial net warming effect of these wider  $N_r$  interactions remains to be quantified. Although individual components of  $N_r$  emissions have cooling effects, there are many opportunities for ‘smart management’ linking N and C cycles. These can help mitigate greenhouse gas emissions, while reducing the other  $N_r$ -related environmental threats [19.7].

## Terrestrial ecosystems and biodiversity

92. Atmospheric  $N_r$  deposition is a significant driver of biodiversity loss in terrestrial ecosystems. Rates of  $N_r$  deposition have substantially exceeded critical load thresholds for natural and semi-natural areas since the increase in agriculture-, energy- and transport-related emissions from the 1950s, resulting in a considerable loss of biodiversity in Europe [5.1, 8.2, 20.4].

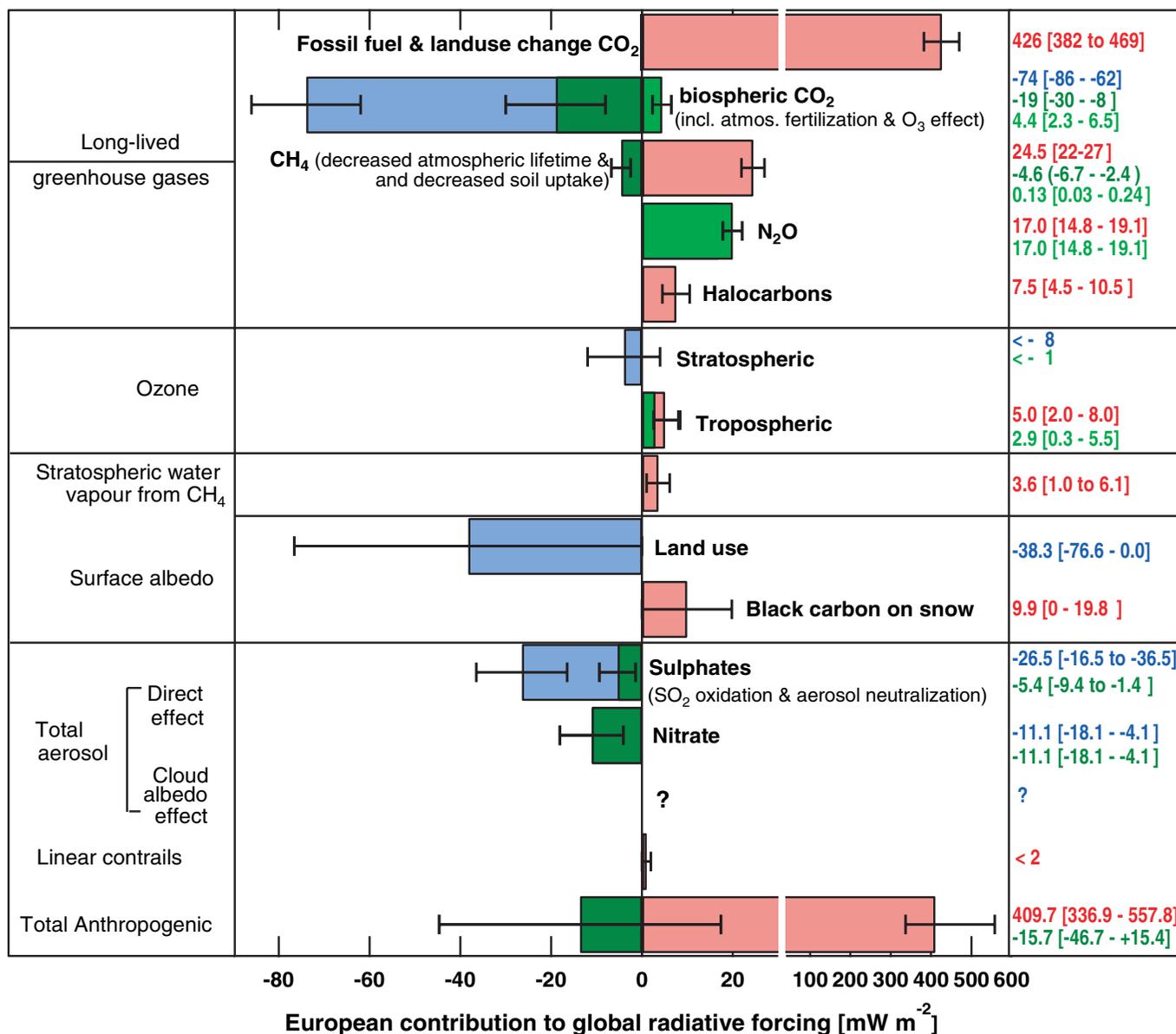
93.  $N_r$  deposition affects vegetation diversity through direct foliar damage, eutrophication, acidification, and pathogen susceptibility. The most vulnerable habitats are those with species adapted to low nutrient levels or sensitive to acidification, and include grassland, heathland, wetlands and forests [20.3]. First estimates of the overall reduction in biodiversity due to  $N_r$  deposition across Europe have been made (Figure TS.10), while the reductions in  $N_r$ -sensitive species will be even greater [20.4].

94. Although it is not yet clear to what extent oxidized versus reduced  $N_r$  (e.g.,  $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ) have different effects on biodiversity, gaseous ammonia ( $\text{NH}_3$ ) can be particularly harmful to vegetation, especially lower plants, through direct foliar damage [20.3]. Changes in plant communities may also indirectly affect faunal biodiversity. It is likely that  $N_r$  deposition acts synergistically with other stressors, in particular climate change, acid deposition and ground-level ozone, but these synergies are poorly understood [20.2].

95. Because of cumulative effects of  $N_r$  inputs, it is likely that biodiversity has been in decline for many decades due to  $N_r$  deposition. This implies that recovery after a reduction in deposition is likely to be slow, and in some cases may require active management intervention in the habitats affected [20.5].

## Soil quality

96. The major  $N_r$  threats on soil quality for both agricultural and natural soils are soil acidification, changes in soil organic matter content and quality and loss of soil biodiversity linked to eutrophication. Application of N fertilizers and manure and atmospheric  $N_r$  deposition cause soil acidification, which leads to a decrease in crop and forest growth and increases leaching of components negatively affecting water quality, such as heavy metals [21.3].



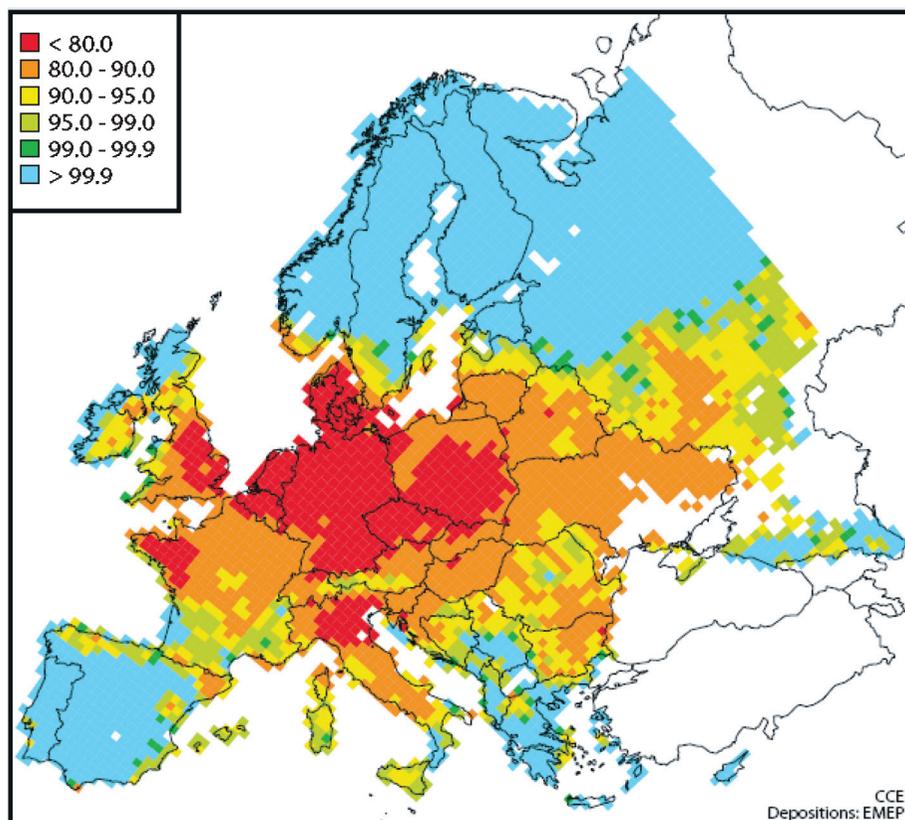
**Figure TS.9** Estimate of the change in global radiative forcing (RF) due to European anthropogenic reactive nitrogen (N<sub>r</sub>) emissions to the atmosphere [19.6]. Red bars: positive radiative forcing (warming effects); light green bars: positive radiative forcing due to direct/ indirect effects of N<sub>r</sub>; blue bars: negative radiative forcing (cooling effects); dark green bars: negative radiative forcing due to direct / indirect effects of N<sub>r</sub>. For biospheric CO<sub>2</sub>, the dark green bar represents the additional CO<sub>2</sub> sequestered by forests and grasslands due to N<sub>r</sub> deposition, while the light green bar represents the decrease in productivity due to effects of enhanced O<sub>3</sub> caused by NO<sub>x</sub> emissions. For CH<sub>4</sub>, the positive (not visible) and negative contributions represent the effects of N<sub>r</sub> in reducing CH<sub>4</sub> uptakes by soil and the decreased atmospheric lifetime, respectively. Other contributions include the positive effect of tropospheric ozone from NO<sub>x</sub> and the direct and indirect cooling effects of ammonium nitrate and sulphate containing aerosol.

98. Nitrogen stocks of agricultural land are often depleted compared with semi-natural systems, because of soil disturbance, crop removals and increased N<sub>r</sub> losses (para. 25). Addition of nitrogen generally has a positive effect on the quality of agricultural soils, by enhancing soil fertility, soil organic matter and conditions for crop growth, especially when added with carbon in manures [6.4, 21.3].

99. Some soil fungi and N fixing bacteria are reduced by high N availability, although the effect of N on diversity of soil organisms and the effects of changes of soil biodiversity on soil fertility, crop production and N<sub>r</sub> emissions are not fully understood [21.3].

100. In widespread pyrite containing soils, nitrate removal from groundwater by pyrite oxidation increases concentrations of cations, heavy metals and sulphate, causing problems for its use as drinking water [21.2].

101. Model simulations indicate that most of the European forest soils could recover from their acidified state within a few decades, as a result of recent and possible future reductions in SO<sub>2</sub> and NO<sub>x</sub> emissions. Although NH<sub>3</sub> emissions have only decreased slightly and can contribute to soil acidification, the effects of NH<sub>3</sub> on plant and soil biodiversity by eutrophication appear to be of more concern [21.4, 20.3].



**Figure TS.10** Indication of the nitrogen threat to terrestrial ecosystems and biodiversity. The map shows modelled similarity in species composition in European forests between un-impacted conditions, and under long-term enhanced N deposition (1990 scenario) [20.4].

## Part V European nitrogen policies and future challenges

### Costs and benefits of nitrogen

102. Cost–benefit analysis can provide guidance for the setting of policy priorities to abate  $N_r$  emissions with an integrated perspective. Social cost estimates have been derived based on available estimates of ‘willingness to pay’ for human life and health, ecosystem services and greenhouse gas emission reduction [22].

103. Total annual N-related damage cost in the EU-27 is estimated at around €70–€320 billion (i.e., €150–750 per capita per year), representing 1%–4% of the average European income [22.6]. Comparison of the overall costs, for each  $N_r$  and threat, is summarized in Table TS.2, which shows that health damage and air pollution cause the largest costs. A provisional ranking of damage from N-emissions in terms of cost, expressed as € per kg N released to the environment, is provided in Table TS.3.

104. Although the unit social costs of  $N_2O$  effects on climate (€5–€15 per kg N) are similar to several of the other effects (Table TS.3), because total  $N_2O$  emissions are much smaller, the overall cost to Europe of  $N_2O$  emissions (also from increased UV radiation) is much smaller than the other effects (Table TS.2). The further effects of  $N_r$  emissions on radiative balance (para. 87) are not expected to alter this overall conclusion, although future work is needed to extend the analysis.

105. The estimated social benefit of  $N_r$  for the farmer is €1–€3 per kg added N-fertilizer equivalent [22.3]. The total marginal environmental costs associated with N-emissions tend to exceed the marginal benefits for the farmer. Internalizing the environmental costs of N-fertilization would lower the optimal N-rate for arable production in north-west Europe by about 50 kg per ha per year [22.6].

106. The results provide support for N-policies on air pollution and human health, and on reducing ammonia emissions from agriculture, as the social benefits of abatement tend to exceed the additional costs. While strategies that give simultaneous emission reductions are attractive, the results point to the need to prioritize abatement of  $NO_x$ ,  $NH_3$  and  $N_r$  loss to water over the abatement of  $N_2O$  emissions [22].

### Integrated approaches to nitrogen management

107. The European Union has many policy measures aimed at decreasing unwanted N emissions from combustion, agriculture and urban wastes. Such sector-, media- and individual pollutant-based approaches have had varying success [4.5, 5.2, 22.1]. However, even under favourable land use scenarios the  $N_r$  export to European waters and seas is expected to remain problematic in the near future, as are atmospheric  $NH_3$  and  $N_2O$  emissions from agricultural activities, leading to ongoing threats to water quality, air quality, greenhouse balance, ecosystems and biodiversity and soil quality [17–21].

108. The lack of a holistic approach to the nitrogen cascade leads to risks of contradictory effects of policies dealing with

**Table TS.2** Estimates of overall social damage costs in the European Union (EU-27) as a result of environmental N<sub>r</sub>-emissions (billion € per year at 2000). Values are shown here rounded to the nearest 5 billion € to avoid over precision, explaining differences with the sums. The calculated value for N<sub>2</sub>O effects on human health is 1–2 billion € per year [22.6]

	NO <sub>x</sub> emission to air	NH <sub>3</sub> emission to air	N <sub>r</sub> loss to water	N <sub>2</sub> O emission to air	Total
Human health	35–100	5–70	0–20 <sup>a</sup>	<5	40–190
Ecosystems	5–35	5–35	15–50 <sup>a</sup>	—	25–115
Climate	—	—	—	5–10	5–10
Total	40–135	10–105	15–70	5–15	70–320

<sup>a</sup> The value for health effects is proportionately smaller than the value for ecosystems as not all leaching is associated with health effects (e.g., denitrified during the path from soil to sea).

**Table TS.3** Estimated cost of different N<sub>r</sub>-threats in Europe per unit N<sub>r</sub> emitted [22.6]

Effect	Emitted nitrogen form	Emission/loss to	Estimated cost € per kg N <sub>r</sub> emitted
Human health (particulate matter, NO <sub>2</sub> and O <sub>3</sub> )	NO <sub>x</sub>	Air	10–30
Ecosystems (eutrophication, biodiversity)	N <sub>r</sub> (inc. nitrate)	Water	5–20
Human health (particulate matter)	NH <sub>3</sub>	Air	2–20
Climate (greenhouse gas)	N <sub>2</sub> O	Air	5–15
Ecosystems (eutrophication, biodiversity)	NH <sub>3</sub> and NO <sub>x</sub>	Air	2–10
Human health (drinking water)	N <sub>r</sub> (inc. nitrate)	Water	0–4
Human health (increased ultraviolet radiation from ozone depletion)	N <sub>2</sub> O	Air	1–3

different aspects of the problem (para. 12) [4.6, 5.2]. The promise of integrated approaches to N management is that these are more effective (larger decreases in unwanted emissions) and/or more efficient (less side effects, less complexity) than the set of policies focused on individual sources and N<sub>r</sub> forms [23.2].

109. A conceptual framework developed here distinguishes five dimensions of integration: (i) vertical dimension, i.e., cause–effect relationships of N species; (ii) horizontal dimension, i.e., integration of all N species via for example N budgets; (iii) integrating N management with the management of other biogeochemical cycles, (iv) integrating stakeholders views, and (v) regional integration [23.2].

110. The toolbox for developing integrated approaches to N management includes systems analyses, communication, integrated assessment modeling, N budgeting, stakeholder dialogue and chain management [15, 16, 22, 23.3, 26]. Integrated approaches are especially applicable to agriculture, because of its multiple sources, multiple N species, and multiple inter-related actors [23.4].

111. It remains a challenge to define the optimum level of integration for various situations and cases. A package of seven key actions in four sectors is envisaged that should contribute to further developing integrated approaches to N management (Figure TS.5) [23.5].

### Agriculture

- (1) Improving nitrogen use efficiency in crop production.
- (2) Improving nitrogen use efficiency in animal production.

- (3) Increasing the fertilizer N equivalence value of animal manure.

### Transport and industry

- (4) Low-emission combustion and energy-efficient systems.

### Waste water treatment

- (5) Recycling nitrogen (and phosphorus) from waste water systems.

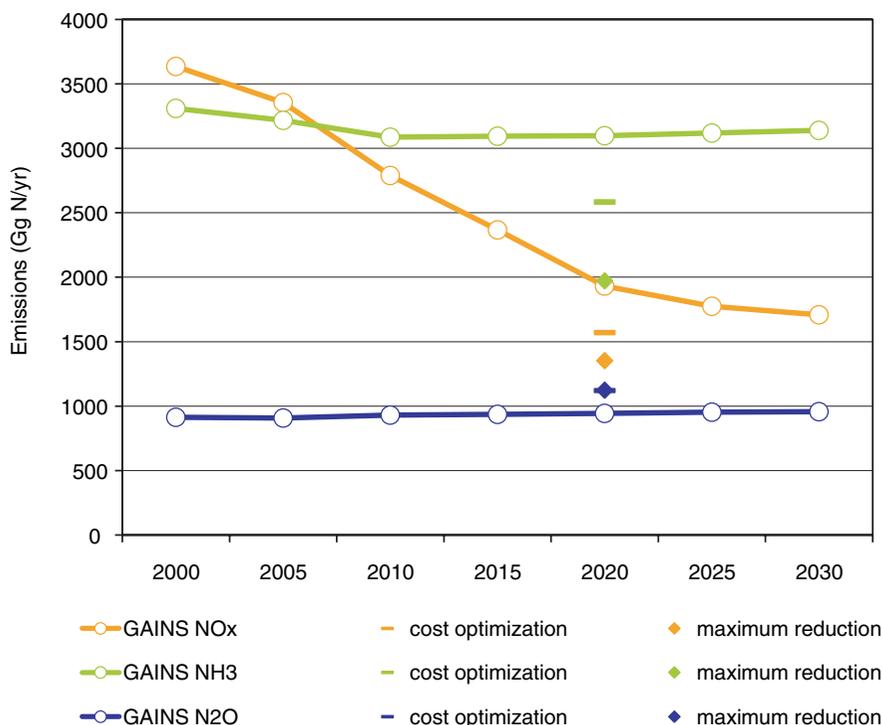
### Societal consumption patterns

- (6) Energy and transport saving.
- (7) Lowering the human consumption of animal protein.

### Future scenarios

112. Scenarios of nitrogen use follow the approaches currently used for air pollution, climate, and ecosystem projections. Short-term projections (to 2030) are developed using a ‘baseline’ path of development, which considers abatement options that are consistent with European policy. For medium-term projections (to 2050) and long-term projections, the European Nitrogen Assessment applies a ‘storyline’ approach similar to that used in the IPCC Special Report on Emission Scenarios (SRES) [24.4]. Beyond 2050 in particular, such storylines also take into account technological and behavioral shifts [24.5].

113. All scenarios agree in projecting a decrease in NO<sub>x</sub> emissions, while agricultural nitrogen use is expected to remain the leading cause of N<sub>r</sub> release to the environment. However,



**Figure TS.11** Total release of NO<sub>x</sub>, NH<sub>3</sub> and N<sub>2</sub>O to the atmosphere from the EU-27 as estimated by the GAINS model, including scenarios to 2030. The main lines show estimates according to current legislation ('Climate & Energy package'). Estimates for 2020 are given according to cost optimization by GAINS and according to the maximum feasible reduction based on technical measures only, as included in the GAINS model. The small increase in N<sub>2</sub>O emissions under the optimized and maximum reduction scenarios is related to estimated pollution trade-offs incorporated into the model [24.6]. Such potential trade-offs highlight the need to prioritize approaches that lead to reductions across the nitrogen cascade [23.5].

integrated assessment modelling including available technical measures indicates that cost-optimized NO<sub>x</sub> and NH<sub>3</sub> emissions for 2030 are substantially smaller than current reduction plans (Figure TS.11), highlighting the case for further emission reductions [24.6].

114. Major reductions in agricultural N<sub>r</sub> emissions will occur only if the extent of agricultural production changes, for example linked to changing human populations or per capita consumption patterns. Such a scenario is examined based on a healthier 'low meat' diet leading to lower N<sub>r</sub> losses [23.5, 24.5, 26]. The scenario, consisting of 63% less meat and eggs, would reduce NH<sub>3</sub> emissions from animal production by 48%. The associated land use changes need to be further explored. For example, a possible increase in intensively fertilized bio-fuel crops would lead to 'pollution swapping', with N<sub>2</sub>O and N<sub>r</sub> losses to water [24.5].

### The role of international conventions

115. International treaties, such as multilateral environmental agreements (MEAs), including conventions and their protocols, have done much to protect the global environment through promoting intergovernmental action on many environmental issues. MEAs and inter governmental organizations (IGOs) between them have targeted most environmental problems, but none has targeted nitrogen management policy holistically [4.3, 25.2].

117. Scientific and technical cooperation between MEAs has proved especially important in identifying the many links between reactive nitrogen threats, with the international scientific community able to provide an important role in

harmonizing information (para. 64) and promoting coordination as a foundation for action [25.3].

118. A new international treaty targeted explicitly on nitrogen could be a powerful mechanism to bring the different elements of the nitrogen problem together. While a new convention targeted on nitrogen would be complex to negotiate and could compete with existing structures, a joint protocol between existing conventions could be effective and should be explored [25.3, 25.4].

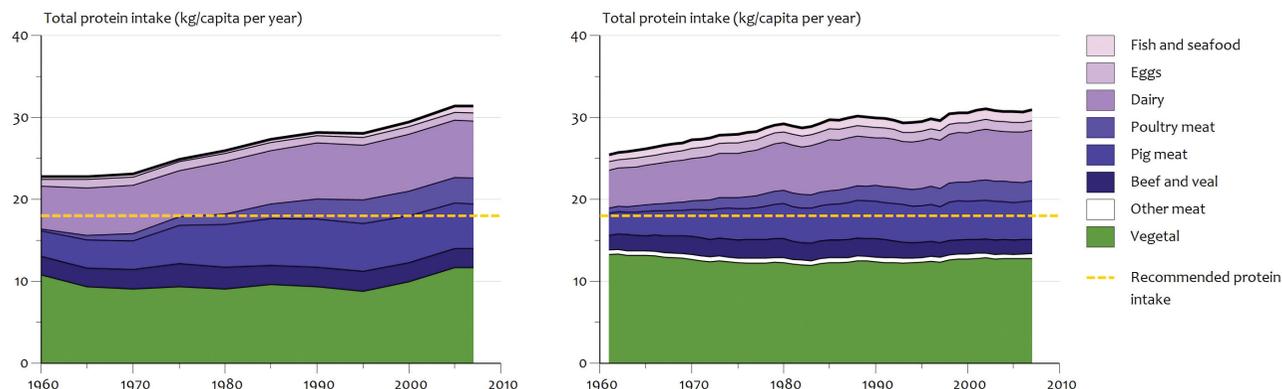
119. The immediate recommendation is to exploit established mechanisms and institutions to develop new coordinating links on nitrogen management between MEAs and IGOs, including the Global Partnership on Nutrient Management facilitated by the United Nations Environment Programme, the Task Force on Reactive Nitrogen of the UNECE Convention on Long-range Transboundary Air Pollution and the links with other UNECE Conventions. There is the opportunity for the UNECE Committee on Environmental Policy to develop the nitrogen management links between the UNECE Conventions, while the European Union and its Member States have important roles to play in harmonization and coordination [25.4].

### The role of societal choice and public awareness

120. Public and institutional awareness of the many benefits and threats of nitrogen remains very low. Public understanding is not helped by the complexity of the nitrogen cycle, while it has been insufficiently emphasized how nitrogen links many global change challenges. Increased public awareness has the potential to improve the efficacy of nitrogen

Intake of protein in the Netherlands

Intake of protein in EU-27



**Figure TS.12** Per capita protein consumption by source in the Netherlands and the EU-27 between 1960 and 2007 (PBL-calculations based on FAO). Reproduced with permission of PBL [26.3].

policies, while reducing the risk of antagonisms with other issues [5.4, 23.2, 26.1].

121. Food production, consumption and wastage represent key sectors where societal choice can greatly influence N use efficiency, benefits and threats. As an example, because of the low conversion efficiency of plant to animal products, the production of animal proteins releases at least seven times more reactive nitrogen into the environment than the production of the same amounts of plant proteins [26.3]. At the same time, many European citizens are increasingly eating more animal products than is necessary for a healthy diet (Figure TS.12). Even a limited reduction of the share of meat and milk in the European diet would substantially affect the overall N budget of Europe (Figure TS.5) [23.5, 24.5, 26.3].

122. Although this issue belongs to the field of personal choice, public initiatives, as for instance through encouraging healthy eating and reducing food waste in institutional and school catering, can play a significant role in changing behaviours [23.4, 26.3].

123. Awareness of nitrogen communication tools for analysts, media and European citizens should be increased, for example considering ‘nitrogen footprints’ alongside those for carbon. Succinct messages that convey the nitrogen challenge facing Europe are required. Only by joining efforts between policy makers, producers and the public will we be able to take effective action in managing our ‘nitrogen inheritance’ [1.1, 25.4].

