Guidance Document on Nitrogen Impact Assessment Methods

International Nitrogen Management System

INMS Guidance Document Series

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About the International Nitrogen Management System (INMS)
INMS is a global science-support system for international nitrogen policy development established as a joint activity of the United Nations Environment Programme (UNEP) and the International Nitrogen Initiative (INI). It is supported with funding through the Global Environment Facility (GEF) and over 80 project partners through the 'Towards INMS' project (2017-2024). INMS provides a cross-cutting contribution to multiple programmes and intergovernmental conventions relevant for the nitrogen challenge. These include the Global Partnership on Nutrient Management (GPNM) and the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA), the UN Convention on Biological Diversity (CBD) and the UNECE Convention on Long-Range Transboundary Air Pollution (Air Convention), through its Task Force on Reactive Nitrogen (TFRN). INMS receives major additional funding through the work of the GCRF (Global Challenge Research Fund) South Asian Nitrogen Hub supported by the UK Research & Innovation (UKRI).

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Cover photo View of Rocky Mountain National Park, USA © Hideaki Shibata 2018

INMS Guidance Document on Nitrogen Impact Assessment Methods

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Foreword

There is an odd contradiction that almost everyone has heard of nitrogen, yet so few people know about nitrogen as an environmental challenge. Some have even doubted how nitrogen pollution can be a problem since it makes up 78% of the air we breathe, as di-nitrogen gas (N_2). It becomes obvious that robust scientific guidance is needed to inform actions by society.

The contradiction is easily explained by the existence of different nitrogen forms. Di-nitrogen gas is abundant because it is unreactive, so that it cannot be used by most organisms. But if energy is used to force apart the N_2 molecule, an array of reactive compounds can be produced, such as ammonia, nitrogen oxides, amines, amino acids and proteins. Nitrogen becomes the stuff of life, as every organism's DNA is a unique nitrogen compound, just as chlorophyll and haemoglobin have nitrogen at their heart.

Collectively, scientists refer to these compounds as reactive nitrogen (N_r). It is here that things get exciting, because N_r compounds are naturally in short supply, and because there are so many interactions between the different N_r forms and effects. On the one hand, humans have deliberately increased rates of 'nitrogen fixation' converting N_2 into usable N_r forms. This has included accelerating rates of biological nitrogen fixation and large-scale industrial formation of ammonia and nitrogen oxides. The resulting reactive nitrogen flows from these processes have increased food production, helping to sustain a growing world population, but, together with N_r mobilized by combustion including from fossil fuels, have led to a cocktail of pollution with multiple effects on the environment.

The present *INMS Guidance Document on Nitrogen Impact Assessment Methods* has been prepared in recognition of this wide range of different nitrogen impacts, with the aim of providing clarity on the assessment approaches. The document is targeted particularly to the scientific community supporting governments, such as National Focal Points to the UNEP Working Group on Nitrogen, which is following up the United Nations Environment Assembly Resolutions 4/14 and 5/2 and the Colombo Declaration on Sustainable Nitrogen Management. As part of this emerging process, INMS is providing scientific support that can help countries as they start to develop their first National Nitrogen Action Plans. The diversity of nitrogen threats addressed in this guidance document, together with examples of nitrogen-related policies, can help build a shared understanding across major nitrogen issues, such as water, air and soil quality, biodiversity, climate and stratospheric ozone depletion. The guidance document will also benefit business, civil society and citizens themselves, informing about the key nitrogen impacts by distinguishing the most important Drivers, Pressures, States, Impacts and possible policy and practice Responses, applying the well-known DPSIR framework.

The document makes an important contribution to the INMS Guidance Document Series. It builds on the approach already established in the UNECE Air Convention, as illustrated by the guidance documents on Ammonia Abatement and on Integrated Sustainable Nitrogen Management. In the present case, the goal is to consider the nitrogen challenge in a global context, linking water, air, biodiversity, climate, health, food production, and commerce. Other complementary INMS guidance documents focus on nitrogen flows and concentrations, nitrogen use efficiency, nitrogen budgets and mitigation methods.

Most importantly, we hope that governments and others will find the guidance useful in mobilizing action to towards the ambitious UN goals to reduce nitrogen waste and pollution by 2030 and beyond.

Mark A. Sutton[1], Martha Schlegel[1], Jill S. Baron[2] and Hans J.M. van Grinsven[3]

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Executive summary

Hideaki Shibata, Jill Baron, Azusa Oita, Allison Leach, Daniel Liptzin, Timothy Weinmann

This document provides guidance for assessing nitrogen (N) impacts on the environment and humans at all scales from local to regional to global and was compiled with input from scientists worldwide.

Nitrogen provides benefits to humans as an essential nutrient for food production and to the environment by stimulating ecosystem productivity. Too much or too little reactive N in the environment causes positive and negative impacts to water quality, air quality, greenhouse gas balance, ecosystems and biodiversity, and soil quality (Sutton et al. 2013). In this document, we describe impacts on greenhouse gases, human health, terrestrial ecosystems, agricultural products, aquatic ecosystems, cultural services and non-agricultural products. Nitrogen takes multiple forms in the environment and behaves dynamically and interactively at different spatial and temporal scales depending on its source (industry, power generation, agriculture, transportation, natural), its forms (oxidised, reduced), and its receptors (humans, natural terrestrial, freshwater and marine ecosystems, the atmosphere, managed ecosystems including agriculture, human-made structures). Reactive N is highly mobile and can move from its original sources through many receptors and impacts before being chemically transformed back to the chemically inert di-nitrogen (N_2), a process described as the nitrogen cascade (Galloway et al. 2003).

The general background and overall concepts of N impacts are described in Chapter 1. The 'Pathways to Reactive Nitrogen Impacts' model is introduced as a general framework showing the transformation processes of drivers, pressures and impacts to describe and analyze the positive and negative effects of altered reactive N cycles in different environments (Figure 1.4). It is based on the concept of DPSIR (Drivers, Pressures, States, Impacts, Responses) described in Chapter 3 (Kristensen 2004). In Chapter 2, the functions across N pressures, states and impacts are summarised. Integrated methodologies, including DPSIR, to assess reactive N impacts are described in Chapter 3. Several other integrated methods, including Input-Output Budgets (Section 3.2), Nitrogen Footprint approaches (Section 3.3), Nitrogen Use Efficiency (NUE) assessments (Section 3.4), planetary boundary approaches (Section 3.5) and Critical Loads (Section 3.6) are also described. Chapter 4 presents assessment methods of individual N impacts on water quality, air quality, greenhouse gas balance, ecosystems and biodiversity, soil quality, food and non-food agricultural products, and (bio)energy. Each subchapter includes a summary, background on the consequences of N for each response variable, methods, and, where applicable, interactions of N with other types of disturbances. Chapter 5 provides cases where N impact methodologies have been applied for policy or management issues, receptors and scales.

One of the central features of this guidance document is to link the multiple dimensions, sources, impacts and policy responses for nitrogen. This breadth highlights new connections, such as between ecosystems, human health, climate and materials. It also highlights the importance of emerging nitrogen flows, such as industrial nitrogen flow, where ongoing development could increase future global flows by a factor 3-4 (Section 4.7).

It is our intent that this guidance document provides background support for research and assessment programs that describe and analyze the multiple positive and negative effects of reactive N at multiple scales (from local, national, regional to global). It serves as the foundation for improved integrated assessment and policy support, including support to the regional demonstration activities of the GEF/UNEP project, 'Towards an International Nitrogen Management System', as a contribution to the wider INMS process.

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Chapter 1: Background

1.1 General introduction

All living things need nitrogen (N). It is an essential nutrient for growth and survival, yet throughout much of the biological history of Earth, reactive nitrogen N (often presented as N_r but herein referred to as N for simplicity) has been in short supply, creating N limitations. Prior to human interventions in the N cycle, the major source of N for agriculture was biological nitrogen fixation and lightning. A select group of microorganisms can convert di-nitrogen (N_2) gas from the atmosphere into forms accessible to living organisms. The industrial Haber-Bosch process developed in the early twentieth century enabled the growth of a global industry to supplement the biological nitrogen fixing process as a source of N. The Haber-Bosch process resulted in the contemporary abundance of N fertilizer for food production in many, but not all, parts of the world. Industrially produced fertilizers have been estimated to be responsible for feeding half of the current global population under certain assumptions (Erisman et al. 2008). Large amounts of N are also produced unintentionally by fossil fuel combustion in manufacturing and transportation to produce food, energy and many other goods and services.

Once N has been produced it is difficult to control its fate because most chemical forms are easily transported in air and water. Reactive N can take many chemical forms as it moves through the Earth system, causing a variety of effects as it transforms from one chemical form to another. The N-cascade concept, introduced in 2003 and depicted in Figure 1.1, describes the many pathways by which N can move through and influence humans and the environment (Galloway et al. 2003).

While some N ends up in intended food and fibre products, the remainder of N, along with N in waste/residue streams and that N produced unintentionally, ends up cascading through the environment in a variety of chemical forms (Galloway et al. 2003; Sutton et al. 2011; Figure 1.1). Excess N affects human health, the natural environment, the built environment and climate stability (Sutton et al. 2013; Jones et al. 2014). In most developed countries, there is an overabundance of N and the above-mentioned threats are documented. However, N is not abundant everywhere. Some parts of the world lack access to N from fertilizers or biological N fixation, limiting food production and contributing to food insecurity. Even in regions that experience food insufficiency, negative impacts to health and the environment occur due to the inherent mobility of N, lack of enforced regulations and ignorance of best management practices. Globally, excess N load to the environment has been recognised as one of the most urgent global environment problems, potentially exceeding the planetary boundary — the current safe-operating capacity of our planet (Rockström et al. 2009; Steffen et al. 2015; UNEP 2019; Persson et al. 2022).

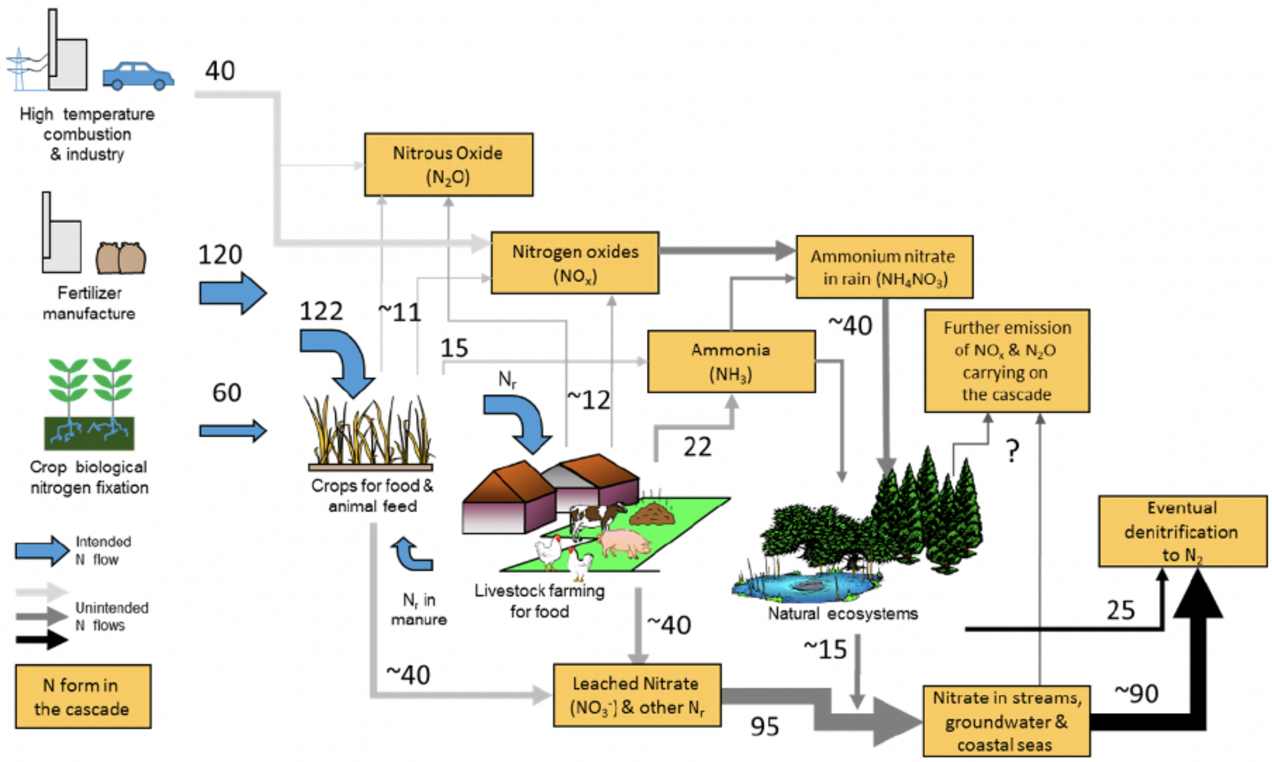


Figure 1.1. Global nitrogen (N) flows (Tg year^{-1}) expressed in the form of the nitrogen cascade. Intended flows are shown with blue arrows, while unintended flows are shown in grey-black arrows. The diagram emphasizes the magnitude of nitrogen wasted through losses of different nitrogen forms, with the overall global system efficiency is around 20%, with 80% of nitrogen inputs wasted (Sutton et al. 2019). © UKCEH 2019.

Impacts of intended use and unintentional losses of N are characterised by Pressure-Impact relationships. To quantify pressure-impact relationships, one must follow the pathways by which different forms of reactive nitrogen are released into the environment for intended products and unintended consequences. Pathways to impacts conceptualise how different reactive N compounds lost to the atmosphere influence human health, climate, stratospheric ozone, ecosystem and structural integrity (Figure 1.2). The Haber-Bosch industrial process — where N_2 is transformed to ammonia (NH_3) — and the subsequent chemical transformation to ammonium (NH_4^+) and nitrate (NO_3^-) compounds, entering terrestrial and aquatic ecosystems from atmospheric inputs or agricultural fertilizers promote agricultural and forest products, carbon sequestration, but also contribute to river, groundwater and coastal pollution. Haber-Bosch N is also used in a wide variety of manufactured products.

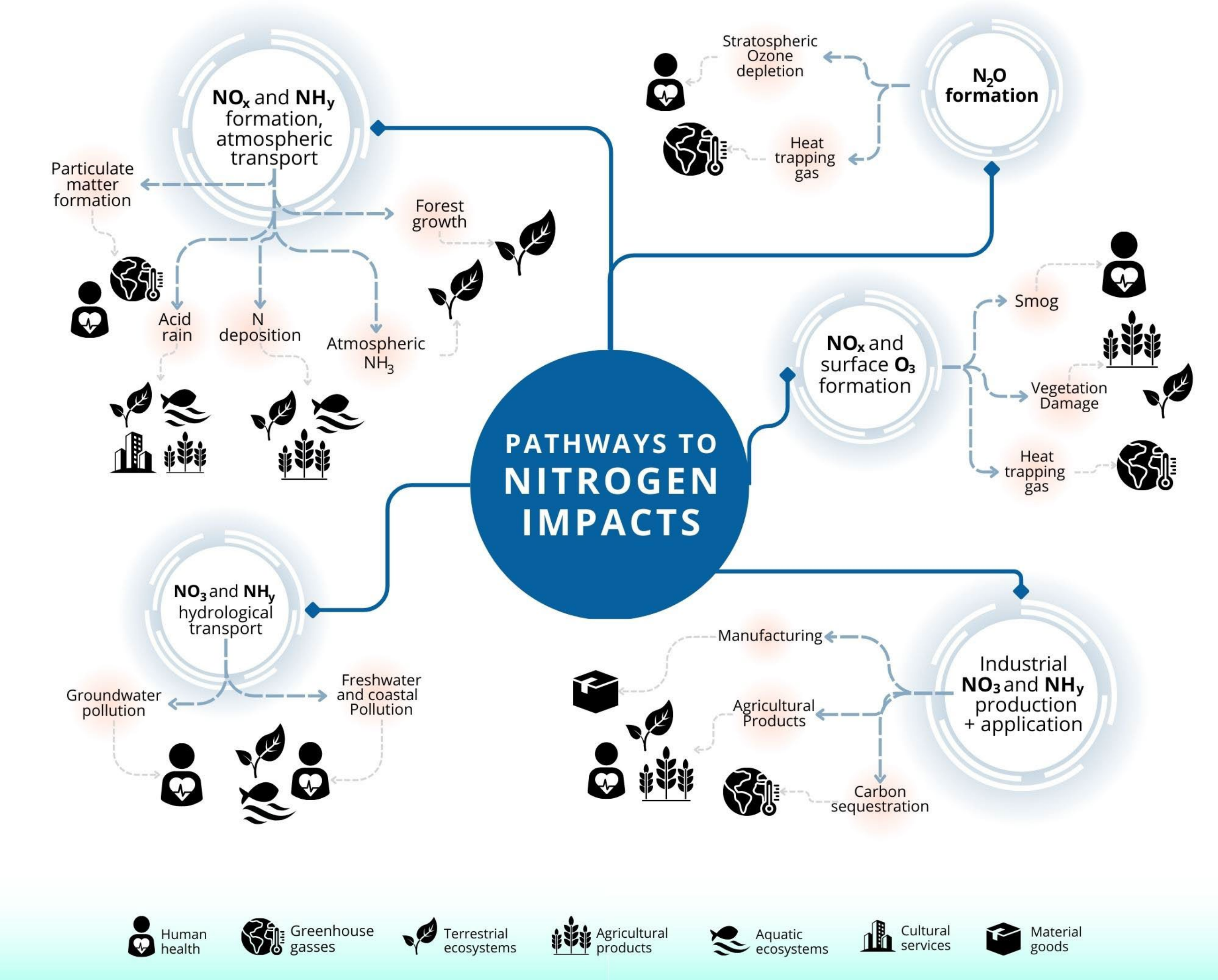


Figure 1.2. Pathways to positive and negative nitrogen (N) impacts. The five white circles are transformation processes leading to categories of impacts, depicted as icons. Original graphic produced for this document © UKCEH 2025.

This guidance document is a contribution to the International Nitrogen Management System (INMS) process, which has been supported since 2017 by the GEF/UNEP Towards INMS project. INMS represents scientific experts from all major world regions who have come together to provide a robust foundation upon which international and national governing bodies can build to develop better ways to manage N to increase the efficiency of beneficial effects and to improve global nitrogen management. Among other products, INMS includes a series of guidance documents as background references. We present here positive and negative impacts assessment methodologies that contribute to the first of four major components of INMS: Tools for understanding and managing the global N cycle.

The primary audiences for this guidance document are academic communities and the intergovernmental organisations that make up parts of the United Nations. These include the UN Environment Programme, the UN regional Economic Commissions, the UN Department of Economic and Social Affairs (including its Division for Sustainable Development Goals), as well as national and other decision-making bodies that are looking for descriptions of how reactive N helps or harms different elements of humans and the environment, as well as indicators that are commonly used to quantify change. We hope other communities of policy makers, as well as other stakeholders with responsibility for N management locally and at all levels of governance within and across nations, will also find this document to be a useful reference. References for all sections are found at the end of the guidance document and should be used to acquire additional details and methodologies.

Many of the UN Sustainable Development Goals (SDGs) that were adopted by United Nations member states in 2015 can be furthered by understanding, assessing and managing N impacts. The SDGs are part of the agenda for sustainable development that provides a blueprint for peace and prosperity for people and the planet (UNDESA 2023). INMS supports progress toward many of the SDGs, including:

- 1: No poverty
- 2: Zero hunger
- 3: Good health and well-being
- 6: Clean water and sanitation
- 7 Affordable and clean energy
- 12: Responsible consumption and production
- 13: Climate action
- 14: Life below water
- 15: Life on land

Scientific findings from INMS can also contribute to international conventions, policy frameworks and programs, including:

- The UN Framework Convention on Climate Change (UNFCCC)
- The UN Convention on Biological Diversity (CBD)
- The Convention on Long-range Transboundary Air Pollution (CLRTAP)
- The UNEP Global Programme of Action on the Protection of the Marine Environment from Land-based Activities (UNEP/GPA)
- The Montreal Protocol on Substances that Deplete the Ozone Layer (Montreal Protocol)
- The Committee on World Food Security (CFS)
- The UN High-level Political Forum on Sustainable Development
- The UN Environment Assembly (UNEA)
- The Organization for Economic Co-operation and Development (OECD)

1.2 DPSIR as a conceptual framework

In this guidance document, we use the Driver-Pressure-State-Impact-Response (DPSIR) approach to guide a quantitative accounting of the causes and consequences of N production and use. The DPSIR framework was developed as a systems-based approach that captures significant relationships between humans and the environment (Figure 1.3; Atkins et al. 2011). The framework is useful for structuring and communicating policy-relevant research about the environment to decision-makers. Social N demand for food, energy and goods are drivers of the nitrogen DPSIR framework. Pressures can cause a change in state, such as an increase in NO_3^- concentrations in water or nitrogen oxides (NO_x) in air. The state change can have an impact on human health or the environment, such as increased growth of harmful algae or increased rates of certain cancers or respiratory issues. Because of these impacts, there may be policy or management responses that change the drivers, like regulating fertilizer addition or power plant emissions, or responses that change the pressures and states, like creating wetlands to remove NO_3^- from agricultural runoff.

To connect N emissions with effects on the environment and human health, we created a more specific DPSIR diagram to capture the distinct N compounds that serve as pressures on states (different parts of the biosphere and the built environment). The impacts occur as effects related to climate change, human health, terrestrial/aquatic ecosystems and biodiversity, agricultural and non-agricultural products, and cultural services (Figure 1.4).

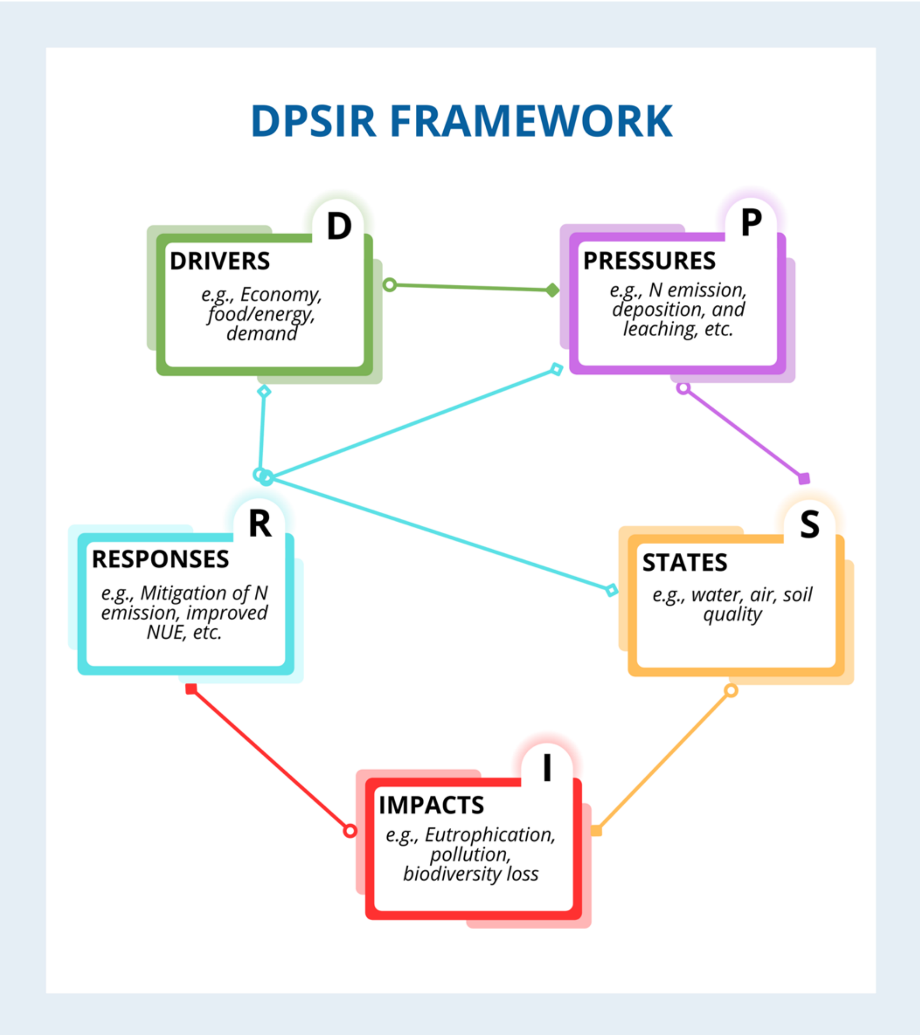


Figure 1.3. Conceptual diagrams of the drivers, pressures, states, impacts and responses (DPSIR) framework for nitrogen (N) impacts assessments (based on Atkins et al. 2011). Nitrogen (N), especially reactive nitrogen; Nitrogen Use Efficiency (NUE). Original graphic produced by Shel Evergreen for this document © UKCEH 2025.

1.3 How to use this document

The conceptual DPSIR framework used in the guidance document illustrates the different N species as pressures for the whole of the nitrogen cascade (Figure 1.4). Different compounds of N move from drivers — economic activities that include industrial, energy, agricultural sources of N — often via transformation of these compounds to impact specific environmental states or pools where N resides or passes through, including air, land, water and materials. From these states, N moves in multiple directions to cause impacts, and these are presented in this guidance document in several topic areas: Climate changes, human health, terrestrial ecosystems, agricultural products, aquatic ecosystems, cultural impacts and non-agricultural products. Policy and management responses can intervene to mitigate N impacts or enhance N products at state, pressure or driver levels.

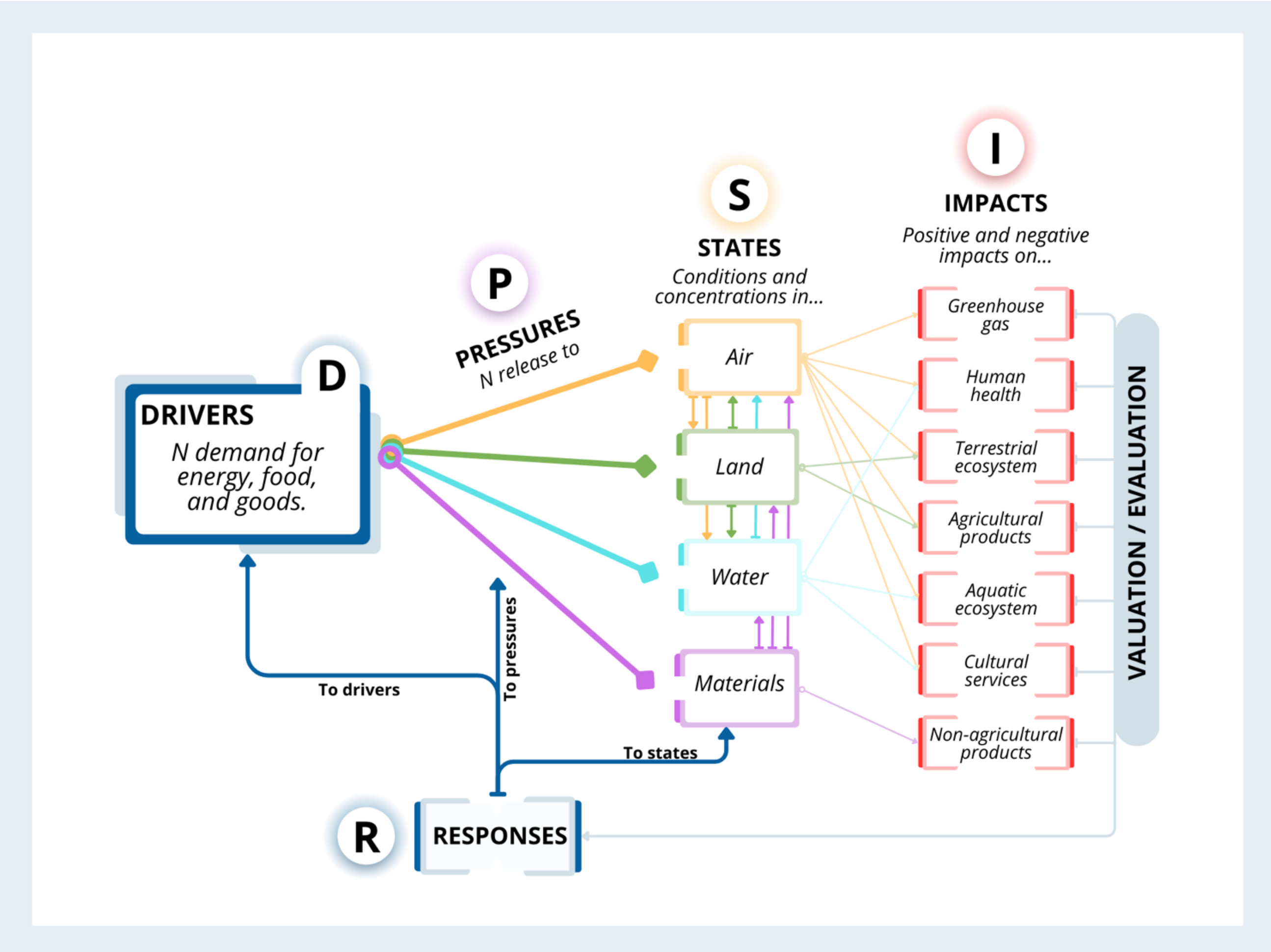


Figure 1.4. Expanded Drivers, Pressures, States, Impacts and Responses (DPSIR) diagram for examining nitrogen (N) flows and their consequences. Original graphic produced for this document © UKCEH 2025

The chapters in this document are organised to present methods by which drivers, pressures and states influence impacts and how results from the different methods can be used to provide information that informs policy, science and management decisions. Specifically, this guidance document focuses on pressure-states-impacts relations in the DPSIR framework (Figure 1.3). Chapter 3 introduces integrated methods that consider more than one type of influence and produces a type of index of response that can be used for information. Most of the integrated methods were developed specifically to analyze pressure-state linkages (including indicators of performances and thresholds), while others are more general with applicability to, or inclusion of, N impacts. Chapter 4 introduces scientific methods by which threats and benefits from changes in states to the seven impact categories can be identified (Figure 1.4). Each method follows the format of background information, published methods and interactions with other pressures that influence the impacts. Pressure-impact relations (i.e., dose-response functions) can exhibit linear, exponential, curvilinear or sigmoid patterns (Figure 1.5), and we include pictorial depictions for each.

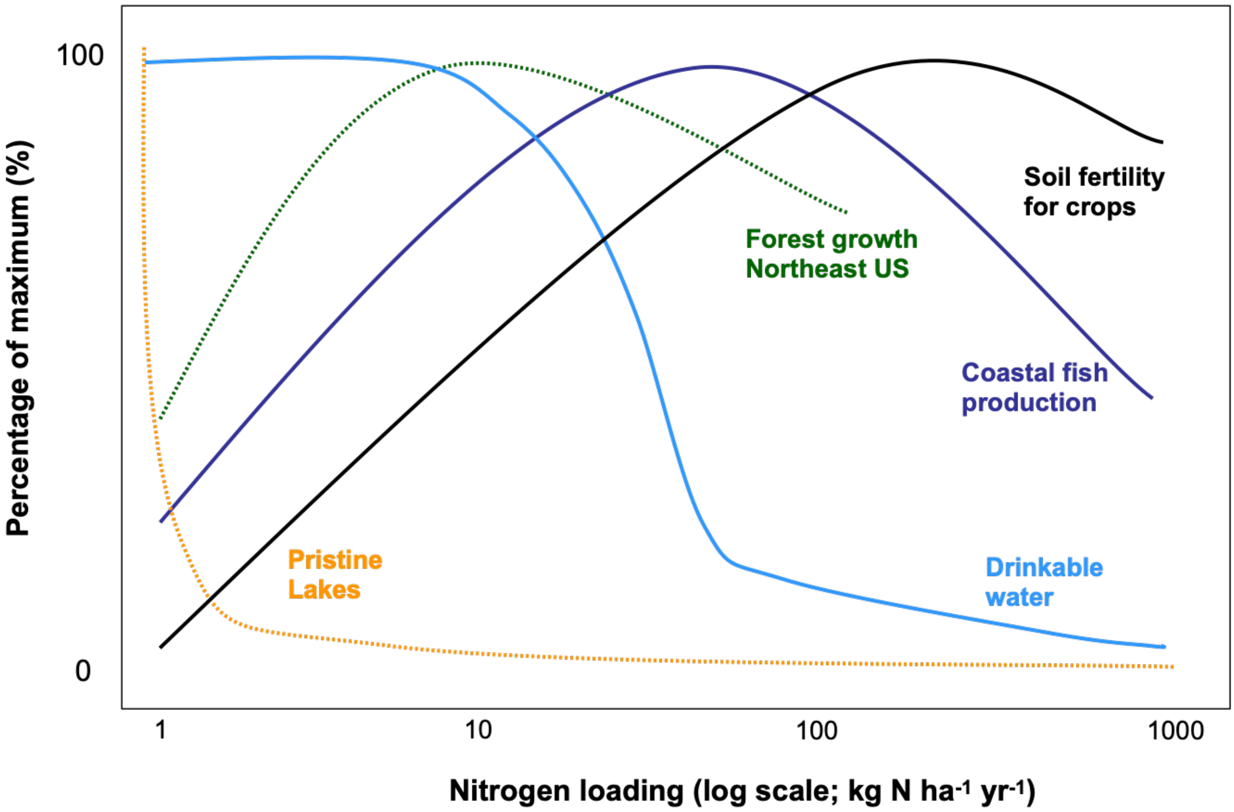


Figure 1.5. Examples of pressure-impact relations of N impacts to ecosystem processes, productivities and environment qualities (Source: Compton et al. 2011). Curvilinear, sigmoidal and exponential relations are shown.

The content of Chapter 5 focuses on the response aspect of the DPSIR framework by providing descriptions and examples of how methods for assessing threats and benefits can and have been applied for improved N management.

All methods are linked to the Nitrogen Matrix of Impacts and Pressures (N-MIP) — a searchable tool for identifying causes and effects, including indicators. This can help to structure and support applications of the methods in developing policy responses, including use of modeling and valuation. The N-MIP was created in a spreadsheet with searchable tools and attached as a supplementary file (N-MIP, Nitrogen Matrix of Impacts and Pressures to this guidance document (Figure 1.6).

Go to search <-- Click to start Provide your comments

Sheet tab	Contents
Search Indicators	Searchable tool of the various information and indicators
P-S-I functions -Summary	Summary table of Pressure - State - Impact functions
Indicators - Full	Overview of the indicators
Impacts matrix full	The master sheet for all items. All information (except model outputs) are listed. Useful for an overview of the whole contents, but better to refer the individual sheet below for the specific purpose.

Items	Descriptions	INMS C1 activity
Mechanisms	Underlying mechanisms for each impact	Common
Category	Classification of each impact by different category (i.e. A1.2, A1.4 and the WAGES (Water, Air, Greenhouse gas, Ecosystem/Biodiversity and Soil) -FE (Food and Energy))	Common
Pressures	Pressure indicators of each impact	A1.3
States	State indicators of each impact	A1.1 and 1.3
Impacts	Impacts indicators of each impact	A1.2 and 1.4
Valuation	Endpoints and intermediate indicators for the economic valuation	A1.4
Modelling	Available global model for each impact	A1.5 and 2.1
Model outputs	Output valuables of the available global model	A1.5 and 2.1

Figure 1.6. Contents of the searchable N-MIP (see supplementary file to this Guidance document), Nitrogen Matrix of Impacts and Pressures. Source: searchable N-MIP see supplementary file to this Guidance document. The right-hand column specifies the relationship to different parts of the Towards INMS project, especially Components 1 and 2 (C1, C2), and specific activities (A1.1-A1.5) contributing to the Components.

N-MIP is a useful tool to find N impacts with brief information of underlying mechanisms and determine the category of the impacts. It contains 38 individual responses to N (positive and negative) linking to relating indicators of pressures, states and impacts defined in the DPSIR framework, and to performance indicators — nitrogen use efficiency, critical load, nitrogen footprint and input-output budget (Chapter 3; Figure 1.7). It also has information on how to link to economic valuations (i.e., endpoint categories and intermediate indicators) for the purpose of external costing and cost-benefit analysis and link to global modeling (e.g., IMAGE-GBM, Global NEWS) to determine each indicator. The impacts are organised into seven categories: Human health, greenhouse gas, terrestrial ecosystem, agricultural productions, aquatic ecosystem, cultural impact and non-agricultural products. Assessment methods for these can be found in Chapter 4.

Pressures indicators
Emission to atmosphere
N_2O emission (kgN_2O/year)
NO_x emission (kgNO_x/year)
NH_3 emission (kgNH_3/year)
Atmospheric deposition
N deposition (kgN/ha/year)
Inputs to terrestrial system
N fertilizer (kgN/ha/year)
Inputs to water bodies
N-runoff (kgN/ha/year)
NO_3 leaching (kgN/ha/year)
NO_2 leaching (kgN/ha/year)
Input to materials
Manufactures N addition (MgN/year)

States indicators			
Air			
NOx (ppb_v)	HNO_3 (ppb_v)	NH_3 (ppb_v)	Acidity (pH in rainwater)
N_2O (ppb_v)	CO_2 (ppm_v)	CH_4 (ppb_v)	O_3 (ppb_v)
PM (ppb_v)	Org-N (ppb_v)	Volatilized org-N (VON) (ppb_v)	Peroxyacetyl Nitrate (PAN) (ppb_v)
Land			
Soil N (mgN/kg soil)	Soil pH (no unit)	Soil CN (ratio)	Soil Al (mg/kg soil)
Plant N (mgN/g plant)	C stock (MgC/ha)	NEP, NPP (MgC/ha/year, Mg/ha/year)	
Phytotoxic Ozone Dose (POD) (mmol/m^2)	W126 (ppm•hours)	Food quality index (Relative value)	
Animal N (mgN/kg animal)	Food N content (mgN/kg food)	Protein content (mg/kg food)	Food N intake (mgN/kg food)
Accumulated Exposure Over Threshold of 40 ppb (AOT_{40}) (ppb•hr)			
Accumulated stomatal Flux above a flux threshold Y (AFstY) (nmol/m^2)			
Water			
NO_3, NO_2 (mgN/L)	NH_4 (mgN/L)	PO_4 (mgP/L)	
Biological Oxygen Demand (BOD) (mg/L)	Dissolved Oxygen (DO) (mg/L)	Chlorophyll-A (µg/L)	Algal species (# of species/m^3)
Materials			
N content in commodities (mgN/kg materials)			

Impacts indicators	
Human health	
Harmful algal bloom (HABs) (mg/L of Cyanobacteria)	Premature death (number of premature death)
Production rates of food, fiber and fuel (kg/year)	
Human health & Cultural services	
Disability-Adjusted Life Years (years)	
Cultural services	
Energy yield (J/year)	Cost of repairing ($/year)
Willingness to pay (WTP) to prevent ($/year)	Number of haze days (days/year)
Greenhouse gas balance & Terrestrial ecosystem	
Global warming potential (GWP) change (relative value to CO_2)	
Terrestrial Ecosystem	
Lichen and moss (# of species/m^2)	Soil fauna (# of species/m^2)
Biodiversity and production loss (Relative values)	Land-use change (ha/year)
Aquatic Ecosystem	
Fish yield (Mg/year)	Chlorophyll (µg/L)
Macrophytes (# of species/m^3)	Macrofauna (# of species/m^3)
Ecosystems & Agricultural products	
Growth rate of plant and animal (kg/ha/year, kg/head/year)	
Agricultural products	
Crop yield (Mg/ha/year)	Wood yield (Mg/ha/year)
Animal yield (Head/year)	
Non-agricultural products	
Production rates of materials (Mg/year)	

Figure 1.7. Examples of pressures, states and impacts indicators in the Nitrogen (N) Matrix of Impacts and Pressures (N-MIP) Source: searchable N-MIP see supplementary file to this guidance document.

Chapter 2: Summary of pressure-state-impact functions

This guidance document focuses on pressures, states and impacts, as these are the direct target processes that determine responses (Figure 1.4). The pathways to nitrogen (N) impacts on human health, environment and structures are complex due to several contributing factors. Impacts are governed by geochemical, photochemical and biological reactions, and exposures via air, water and soil that define the nature of the responses. Ecological processes such as interactions between organisms and confounding factors like the presence of other nutrients or climatic controls constrain the rates of biological N processing. Despite this complexity, and even though there are many effects across seven impact categories, the methods for determining effects are fundamentally similar as described in Chapters 3 and 4. In addition, methods found in the literature and documented in the other guidance documents of INMS, include various monitoring, experiments, models and spatial studies across gradients. Human health effects studies additionally include clinical evaluations, population and epidemiological studies.

Confounding factors may require methodologies that incorporate a synthesis of multiple measures. Such integrated measures and thresholds are also common and can be effective tools for policy determination. These include input-output N budgets, N use efficiency, N footprints and critical loads, all of which are described in more detail in Chapter 3.

2.1 Pressure to states

The oxidised or reduced assemblage of N species are pressures that influence states of air, land, water and materials. These pressures are the vectors by which N species move into the environment to implement changes in states (Table 2.1). For example, when N fertilizer is applied in excess of crop demand, the balance of N lost to the environment creates a pressure that may change the state of N in the soil, the air and/or the downstream waterways. The nature of the state change determines the resulting impact.

Table 2.1. Overview of pressures-to-states connections. Nitrogen (N), Organic (Org). DON is dissolved organic N VON is volatile organic N. Nitrogen oxides (NO_x) are a combination of nitric oxide (NO) and nitrogen dioxide (NO_2). Total oxidized nitrogen (NO_y) is a combination of NO_x, nitrate (NO_3^-) and other oxidized N forms. Total ammoniacal nitrogen (NH_x) is a combination of ammonia (NH_3) and ammonium (NH_4^+).

Pressure(s)	N species	State(s)	Change in State
N fertilizer addition	NO_3^-, NH_4^+	Land	N availability
Emissions to air from fossil fuel combustion	NH_3, N_2O, NO_x	Air	Atmospheric concentration
Atmospheric N deposition	NH_x, NO_y, DON, VON	Land	N availability
Biological N fixation	NH_x	Land, Water	N availability
Emissions to air from land and water	NH_3, N_2O, NO_x	Air	Atmospheric concentration
Leaching to water	NO_3^-, NH_4^+, DON	Water	N availability
Industrial N sources	Organic N in solid materials, VON, DON	Materials	Products or waste products

2.2 States to impacts

Changes in pressures and states lead to impacts through an increasingly complex set of pathways and forces. As N circulates between air, land, water and manufactured states, it influences other chemical pathways such as ozone formation, or cycles such as carbon cycling, sometimes through a causal chain that leads to various effects. For example, while N_2O (as a greenhouse gas) and NO_3^- (in drinking water) have direct impacts on climate and health, respectively, most of the other positive and negative effects occur through intermediate pressures.

The causal chain between pressures and states depicts chemical and biological changes from N reactions, and these are enabled by atmospheric circulation and hydrologic transport processes. Atmospheric transport of airborne N products contributes to wet and dry deposition to ecosystems and materials removed from the source of origin. Hydrologic transport moves N, often as NO_3^-, into soils, groundwater, surface and estuarine waters. Hydrologic processes can transport N hundreds or thousands of kilometres from the source of origin and accumulate in groundwaters for many hundreds of years, leaving a legacy of N pollution that could take decades to remove (Van Meter et al. 2018).

Land use, climate, nutrient and trace metal availability, and species interactions, are some of the many conditions that govern the response of species, populations, communities to changes in N. They also govern how N affects ecosystem processes such as productivity, plant nutrient content, CO_2 respiration and other gas exchange. Difficulty in identifying impacts and assigning causality to them can be compounded by time lags between pressures, changes in states and impacts. This can be caused by system-level buffering, as with acid-base reactions, or delayed response times for long-lived species, such as trees. Terrestrial and aquatic ecosystems can be influenced by emissions of SO_2, by subsequent atmospheric deposition of SO_4 and by other pollutants.

In both terrestrial and aquatic ecosystems, N can alter acid-base status of soils and waters, as noted above, but also lead to a range of ecological effects through complex pathways that include changes in organismal, population and community dynamics through processes that affect competition, herbivory, and other disturbance factors. The impact of N is moderated by access to light, other nutrients and climate.

2.3 Functions of pressure-state-impact

Understanding pressure-state-impact pathways is the first step to assessing N impacts. We use the term "states to impacts" to relate to the Drivers-Pressures-States-Impacts-Response (DPSIR) diagram (Figure 1.4), but these effects are also often called "dose-response" or "cause and effect" relations. Some pathways are direct and obvious, but most are complex as N compounds move through and are altered by different environments and other environmental pressures. Pressure-state-impact responses fall into one or more functional shapes, where an increase in N causes a predictable response (Table 2.2). The response functions include linear, asymptotic, quadratic and sigmoid curves, and shapes can depend on temporal and spatial scales by which changes occur. They also can depend on the initial baseline state. The response of an individual plant to exposure under controlled laboratory conditions will be different from the response of the same plant species as part of an ecosystem, in view of buffer processes and interactions with environment and other species.

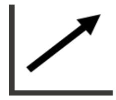
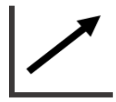
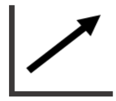
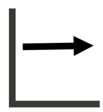
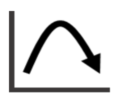
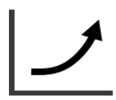
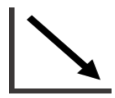
Linear impacts describe a condition where any increase in N via atmospheric, hydrologic or direct application (fertilizer) pathways causes a response; there is no threshold below which there are no effects. Some linear responses relate to human health, such as increased respiratory disease with increased smog and air pollution, and increased incidence of skin cancers with exposure to UV radiation in the absence of preventative measures.

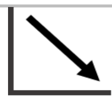
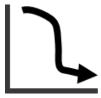
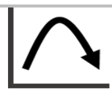
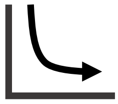
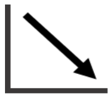
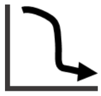
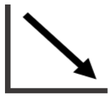
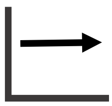
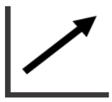
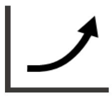
Examples of positive asymptotic impacts include forest C sequestration, crop yields, timber products and biofuels yield, where the increase is quite strong up to a maximum, at which point another environmental factor (e.g., climate, water, other nutrients, increased risk of pests, grain plants bending) limits their productivity (Horn et al. 2019; Cerrato & Blackmer 1990; Zhu et al. 2020). The gains in C storage and agricultural yields are not infinite; when yields level off the excess N begins to affect aquatic and terrestrial ecosystems, and human health (Galloway et al. 2003; Aber et al. 1998).

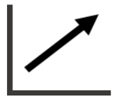
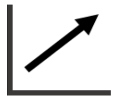
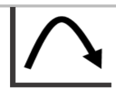
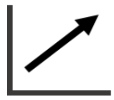
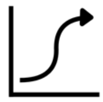
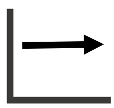
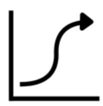
The response to elevated N of terrestrial and aquatic species richness can be characterised either by a quadratic or by a negative sigmoid function. Where ecosystems are initially highly oligotrophic and N limited, N addition stimulates species richness, portrayed by a positive quadratic shape at the beginning of the curve (Clark et al. 2019). With continued increase in N availability, species richness declines due to interspecies competition to lower species richness and dominance by nitrophilic species. In ecosystems, such as many grasslands, where species richness is initially high, species richness declines with N additions (Stevens et al. 2010; Tang et al. 2017; Seabloom et al. 2020). Fish catches also follow a quadratic curve function, where catch increases with nutrient addition to a highly oligotrophic system but decline subsequently due to altered trophic interactions and hypoxia (Breitburg et al. 2009a).

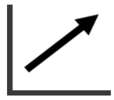
Changes in terrestrial and aquatic acid-base chemistry caused by excess N led to acidification through declines in soil base cations or increases in aluminum solubility. These changes in state indicators constitute thresholds that, once crossed, lead to changes in growth rates and species richness described by sigmoidal curves (Sverdrup & Warfvinge 1993).

Table 2.2. Examples of pressure-state-impact pathways, functions, spatial and temporal scales of effects. These examples are illustrative of outcomes described below for human health and marine eutrophication. *Harmful Algal Blooms (HABs), Haber-Bosch (HB), Biological Nitrogen Fixation (BNF).*

Category	Impact description	Pressure to state pathway	State indicators	Impact indicator (+/-)	State to impact function	Spatial and temporal scale of impact
Human health	Respiratory disease, cancers from NO_x, O_3, particulate matter	NH_3, NO_x emissions	NO_x, $NH_3 \dashrightarrow$ particulate matter (PM) and O_3 in troposphere	Incidence of respiratory disease (-)		Local to regional/years
Human health	Health issues from drinking water contamination	NO_3^- leaching	NO_3^- in surface & groundwater	Incidence of birth defects & cancers (-)		Local to regional/years
Human health	Wheat intolerance, allergies to gluten	Mineral and organic N fertilizer application to wheat	Gliadin (a storage protein)	Celiac disease		Regional to global/years
Human health	Disease associated with high protein & carbohydrate consumption	Mineral and organic N fertilizer application	Crop yield, livestock production, animal & caloric consumption	Incidence of diet-associated cancers, liver & heart disease		Global/years
Aquatic ecosystems	Fish yields	NO_3^- leaching	NO_3^- and DON in surface water $\dashrightarrow$ algal biomass	Fish catch (+/-)		Local to regional/Seasons to years
Aquatic ecosystems	Freshwater eutrophication	NO_3^- leaching	$NO_3^- \dashrightarrow O_2$, cyanotoxin, algal biomass	Biodiversity (-)		Local to regional/seasons to years
Aquatic ecosystems	Freshwater eutrophication	NO_3^- leaching	$NO_3^- \dashrightarrow O_2$, cyanotoxin, algal biomass	Harmful Algal Blooms (-)		Local to regional/seasons to years
Aquatic ecosystems	Freshwater eutrophication	NO_3^- leaching	$NO_3^- \dashrightarrow O_2$, cyanotoxin, algal biomass	Recreational use, property values (-)		Local to regional/seasons to years
Aquatic ecosystems	Marine and coastal eutrophication	NO_3^- leaching	$NO_3^- \dashrightarrow O_2$, cyanotoxin, algal biomass	Biodiversity, HABs, Recreational use,		Local to regional/seasons to years

Category	Impact description	Pressure to state pathway	State indicators	Impact indicator (+/-)	State to impact function	Spatial and temporal scale of impact
				property values (-)		
Aquatic ecosystems	Marine & coastal acidification	NO_3^- leaching, N deposition	NO_3^-, pH	Biodiversity, survival, calcification, growth, reproduction (-)	 	Local to regional/seasons to years
Terrestrial ecosystems	Change in plant biodiversity	Reactive N deposition	Soil N	Biodiversity indices (-)	 	Local to regional/Years to centuries
Terrestrial ecosystems	Changes to below-ground communities, ecosystem processes	Reactive N deposition	Soil N	Abundances & assemblages of soil biota, CO_2 flux (-)		Local to regional
Terrestrial ecosystems	Mortality, toxicity, reduced plant growth rates	Reactive N deposition	Soil N, Soil Ca/Al, Soil pH	Leaf/root damage, biomass/ biodiversity decline (-)		Local to regional/years to centuries
Terrestrial ecosystems	Plant productivity, mortality	NO_x emissions	NO_x, O_3, aerosols	Plant growth rate, drought tolerance (-)		Regional/Days to seasons
Terrestrial ecosystems	Biodiversity protection from agricultural intensification	Agricultural intensification reduced need for land clearing	Agricultural Yield/Area of protected lands	Correlation between intensification & conservation (no impact)		Local
Greenhouse gas balance	Climate warming from tropospheric N_2O	N_2O emissions	Soil N ⇢ N_2O	CO_2 equivalent warming (-)		Global/Years to centuries
Greenhouse gas balance	Climate warming from O_3 formation	NO_x emissions	NO_x ⇢ ground-level ozone	CO_2 equivalent warming (-)	Too spatially complex for a function	Regional/days to seasons
Greenhouse gas balance	Climate cooling from aerosols	NH_3, NO_x emissions	NO_x, NH_3 ⇢ aerosols	CO_2 equivalent cooling (+)		Regional

Category	Impact description	Pressure to state pathway	State indicators	Impact indicator (+/-)	State to impact function	Spatial and temporal scale of impact
Greenhouse gas balance	Climate cooling from CH_4 uptake	Reactive N deposition	Soil N ⇢ CH_4	CO_2 equivalent cooling (+)		Global/Years to centuries
Greenhouse gas balance	Climate warming from tropospheric N_2O	N_2O emissions	Soil N⇢ N_2O	CO_2 equivalent warming (-)		Global/Years to centuries
Agricultural productivity	Food, fiber, biofuels yield	HB fertilizer, manure, BNF	Soil N	Crop yields (+)		Local to global/seasons to years
Agricultural productivity	Food, fiber, biofuels yield	Reactive N deposition	Soil N	Crop yields (+)		Regional/seasons to years
Agricultural productivity	Wood production	Reactive N deposition	Soil N	Woody biomass (+)		Regional/Years to decades
Agricultural productivity	Livestock production	HB fertilizer, manure, BNF	Soil N ⇢ animal feed	Livestock production (+)		Local to global/weeks to years
Agricultural productivity	Food, fiber, biofuels yield	NO_x emissions	NOx ⇢ ground level ozone	Crop yields (-)		Regional/days to seasons
Agricultural productivity	Lodging of rice and other grain plants	HB fertilizer	Soil N	Grain yield (-)		Local/seasons
Agricultural productivity	Decreased baking quality in cereals grown with insufficient reactive N; decreased rice flavour in rice grown with excess reactive N	HB fertilizer	Soil N	Crop quality (-)	NA	Local/seasonal
Cultural services	Damage to buildings, monuments, other surfaces	Reactive N deposition	NO_x, NH_3	Corrosion & erosion of monuments & engineered materials (-)		Regional/years to decades
Cultural services	Haze	NO_x, NH_3 emissions	NO_x, NH_3 ⇢ ground-level O_3, PM	Perception of visibility at scenic vistas and airports (-)		Regional/Days to weeks
Cultural services	Odour	Volatile organic N and/or NH_3 emissions from livestock	Odour threshold concentrations	Perception of odour (-)		Local to regional/days to years

Category	Impact description	Pressure to state pathway	State indicators	Impact indicator (+/-)	State to impact function	Spatial and temporal scale of impact
		farms, wastewater treatment plants				
Non-agricultural products	Industrially produced N products	HB industrial	Industrial N	Industrial N fibers, plastics, rubber, resins, glues, explosives, pesticides, paints, rocket fuel, & medicines (+)		Local/years to centuries
Non-agricultural products	Environmental fate of industrial products	HB industrial	Industrial N waste, toxins	Respiratory disease, air toxins (-)	Variable	Local to regional/years to centuries

Chapter 3: Integrated methodologies to assess nitrogen impacts

3.1 Integrated methods overview

Integrated nitrogen (N) methodologies assess the environmental, human health and economic consequences of the flow of reactive N as it moves through the various components of the biosphere. These assessment approaches provide a framework for considering the negative and positive impacts of N at spatial scales ranging from local to global. Each approach provides policy makers and other practitioners a way of examining the pathways and tradeoffs involved with N fluxes. Below, we describe six integrated approaches toward quantifying the flow of N through the environment and evaluating the environmental responses (Table 3.1). Those six approaches are input-output N budgets, Nitrogen Footprint approaches, N use efficiency, planetary boundaries, critical loads and an environmental performance index. Economic assessments are described in by van Grinsven et al. (2024, see Figure 1.5 of that document).

In this guidance document, integrated N methodologies are defined as N assessment approaches that span several pressures-states linkages in the DPSIR framework (Figure 1.4). Further, those integrated N methods can address several N indicators (e.g., indicators of performance, efficiency, system and others). They can include economic assessments and information on the threshold of pressure (e.g., critical load). Nitrogen-specific integrated methods are included (input-output budget, N footprint approaches and N use efficiency), as well as methods that extend beyond N (e.g., planetary boundaries, critical loads, environmental performance index and ecosystem services).

Pressure-impact functions are ways of attempting to quantify the relation between cause and effect and are easiest to understand when there is a simple response of a variable to a single cause: For example, nitrate (NO_3^-) concentration in a stream for compliance with drinking water standards, and runoff from an agricultural field. Pressure-impact functions can become more complicated when the response depends on additional drivers of change (e.g., temperature, other nutrients or invasive plants for biodiversity; Porter et al. 2012) or different antecedent conditions or states (e.g., pre-existing respiratory and cardiovascular diseases make some people more susceptible to N-caused health issues; Peel et al. 2013). Where applicable, pressure-impact functions are described for the integrated methods in this chapter.

Table 3.1. Summary of the integrated methodologies to assess nitrogen (N) impacts. Drivers (D), Pressures (P), States (S), Impacts (I) and Responses (R).

Assessment: **Major relation to DPSIR**	**Purpose**	**Type**	**Scale(s)**	**Key reference(s)**
Input-output budgets (P, S)	Understanding N cycle, identifying storage & losses, comparison across sites	Quantitative	Farm, ecosystem, watershed, landscape, country, and global scale	Likens 2013
N footprints (P, S)	Link many kinds of entities of supply chains to nitrogen emissions	Quantitative	Individual, institutional, watershed to national and global	Leach et al. 2012; Table 3.4.1
NUE approaches (P, S)	Predict how efficient plants are in use of N, defined as yield (biomass) per unit input of N.	Quantitative	Farm, ecosystem, country, and global	Lassaletta et al. 2014
Planetary boundaries (P, S, I)	Elucidate the environment capacity (i.e., safe-operating space) for various global environmental issues with quantitative thresholds	Quantitative	Regional to global	Rockström et al. 2009; Steffen et al. 2015; Persson 2022
Critical N load & critical inputs and extent of exceedance (P,S, I)	Quantify the critical threshold of atmospheric N deposition (critical load) & N fertilizer inputs (critical inputs) below which there are no discernible effects according to current knowledge	Quantitative	Farm, watershed, ecosystem type	Nilsson and Grennfelt 1988; CLRTAP 2004, 2015
Critical level concentrations and extend of exceedance (P, S, I)	Quantify the extent to which atmospheric concentrations of reactive nitrogen compounds are above threshold concentrations, above which there are discernible effects according to current knowledge.	Quantitative	Landscape, national, regional, global.	CLRTAP 2004. 2015
Environmental Performance Index (S,I)	Provide foundation for effective policy making on a range of pollution control & natural resource management challenges	Quantitative	Country level	Emerson et al. 2012
Economic assessments / cost-benefit assessment (I, R)	Conversion of pressures, states, impacts & responses to social values (often monetary) to weigh multiple measures & effects to identify interventions with highest net benefit (or cost-effectiveness	Quantitative	Regional to global	van Grinsven et al. 2013

3.2 Input-output nitrogen budgets

3.2.1 Method descriptions

Input-output N budgets have been used as tools for integrated assessment of N cycles since the early 1960s (Likens 2013). Nitrogen budgets can be applied at a variety of scales and to a variety of systems. Common applications include watershed budgets, farm budgets and budgets for a defined region. Input-output budgets can provide important information about the overall balance of N inputs to N outputs in a defined system. Some budgets take a "black box" approach, comparing overall inputs to outputs, which means that N cycling within the system is not considered. More complex budgets may consider N cycling within a system, which can help identify sources of N surplus or deficit.

In the simplest version of these budgets, inputs from atmospheric N deposition, N fertilizer, food imports, and biological N fixation are compared with hydrologic and gaseous outputs and food exports. The N inputs minus outputs is commonly referred to as 'retention' for watersheds and 'N surplus' in farming systems. These metrics are frequently used as an index of overall system N status and have been found to be highly responsive to changes (both increases and decreases) in inputs and over a wide range of system changes (e.g., disturbance events, vegetation change, or management changes).

Input-output N budgets assess N pressures and states, but do not directly assess N impacts in the Driver-Pressure-State-Impact-Response (DPSIR) framework (Figure 1.4). This is because the metrics from input-output budgets, such as N retention or N surplus, reflect the estimated amount of N released to the environment, but do not connect directly to how that N affects human and ecosystem health. It is possible to connect N retention or N surplus from input-output budgets with impacts using site- and process-based models (e.g., PnET model).

3.2.2 Applications and discussion

Input-output approaches have been applied at multiple scales. For example, early watershed budget work (Likens 2013; Figure 3.1) focused on watersheds, but the approach is scalable to any size watershed at regional (Howarth et al. 1996; Boyer et al. 2002; van Breemen et al. 2002), national (Houlton et al. 2013) and global scales (Leip et al. 2011; Figure 3.2).

In non-agricultural ecosystems where there is concern about N saturation, watershed approaches are useful for assessing whole-ecosystem response to increases in atmospheric deposition. Much research historically has been focused on anthropogenic activities that increase inputs of N (i.e., pressure or dose) to forests through increases in emissions and atmospheric N deposition (Galloway et al. 2008), as well as the ability of these inputs to accelerate forests toward a condition of N saturation (i.e., impact or response), with adverse consequences on soils, plant growth and aquatic ecosystems (Aber et al. 2003; Aber et al. 1989; Stoddard 1994).

However, atmospheric N deposition is now declining over large areas of North America and Europe (Eshleman et al. 2013; Lloret & Valiela 2016). Watershed mass-balance approaches have been useful for evaluating ecosystem responses to these declines and have raised new questions about ecosystem responses to atmospheric N deposition. For example, there have been remarkable declines in N export from some, especially forest ecosystems, which cannot be explained by declines in atmospheric deposition alone (Bernal et al. 2012; Bernhardt et al. 2005; Driscoll et al. 2016; Fuss et al. 2015; Goodale et al. 2003; Likens 2013; Martin et al. 2000; McLauchlan et al. 2007; Rosi-Marshall et al. 2016; Yanai et al. 2013). This raises questions about how changes in climate and atmospheric chemistry (e.g., CO_2 levels, acidity) interact with N inputs to influence ecosystem response to N inputs. Over large areas of Europe and North America, concerns about N saturation and N supply are being replaced with concerns about N oligotrophication causing N limitation of forest productivity and diminished capacity of ecosystems to dynamically respond to disturbance and environmental change (Durán et al. 2016; Elmore et al. 2016; Groffman et al. 2018).

Nitrogen input-output budgets have highlighted key uncertainties about the N cycle that arise from large amounts of 'unaccountable N' that dominate N budgets at all scales. Inputs of N in fertilizer, atmospheric N deposition and N in sewage have been found to be substantially higher than hydrologic outputs of N in many studies and at many scales (Howarth et al. 1996; Boyer et al. 2002; Groffman 2008; Worrall et al. 2015). There is significant uncertainty about the fate of this excess N (van Breemen et al. 2002): Is it stored in soils or vegetation? Is it converted to gaseous nitrogen, and, if so, in which forms? This uncertainty is particularly compelling in agricultural systems that receive high rates of N input, causing concern about the air and water quality impacts of these N exports (Davidson et al. 2012).

Despite these uncertainties, mass balance approaches are useful for evaluating potential N excess problems. This is especially true in agricultural systems where mass balance analysis can highlight times and places where inputs in fertilizer or N fixation greatly exceed outputs in crop harvest, leaving a large pool of N available for export to the environment. For example, a comparative analysis showed that:

- Large excesses in China where population density drove demand for maximum cropland productivity;
- large N deficits in Africa where fertilizer use is rare;
- and relative balance in North America where there have been extensive efforts to reduce excess fertilizer application (Vitousek et al. 2009).

Further details on national nitrogen budgets can be found in the INMS Guidance document on development of national nitrogen budget approaches (Winiwarter et al. 2025) and in the earlier UNECE Guidance Document (ECE/EB.AIR/119, UNECE 2013). Further information on farm scale nitrogen budgets can be found in the INMS Guidance document on improving nitrogen management using nitrogen budgets for dairy farms (Gourley et al., 2025).

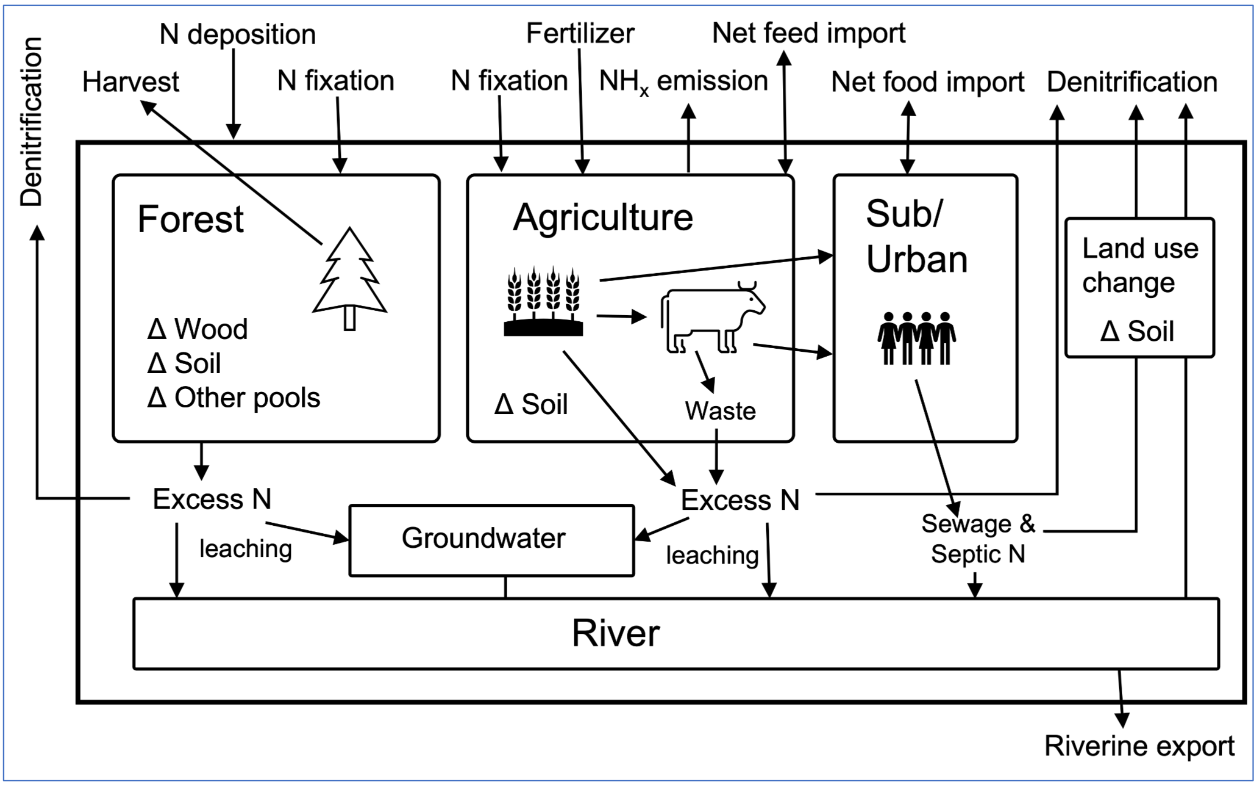


Figure 3.1. Watershed nitrogen (N) budget (redrawn from van Breemen et al. 2002). Original graphic produced for this document © UKCEH 2025.

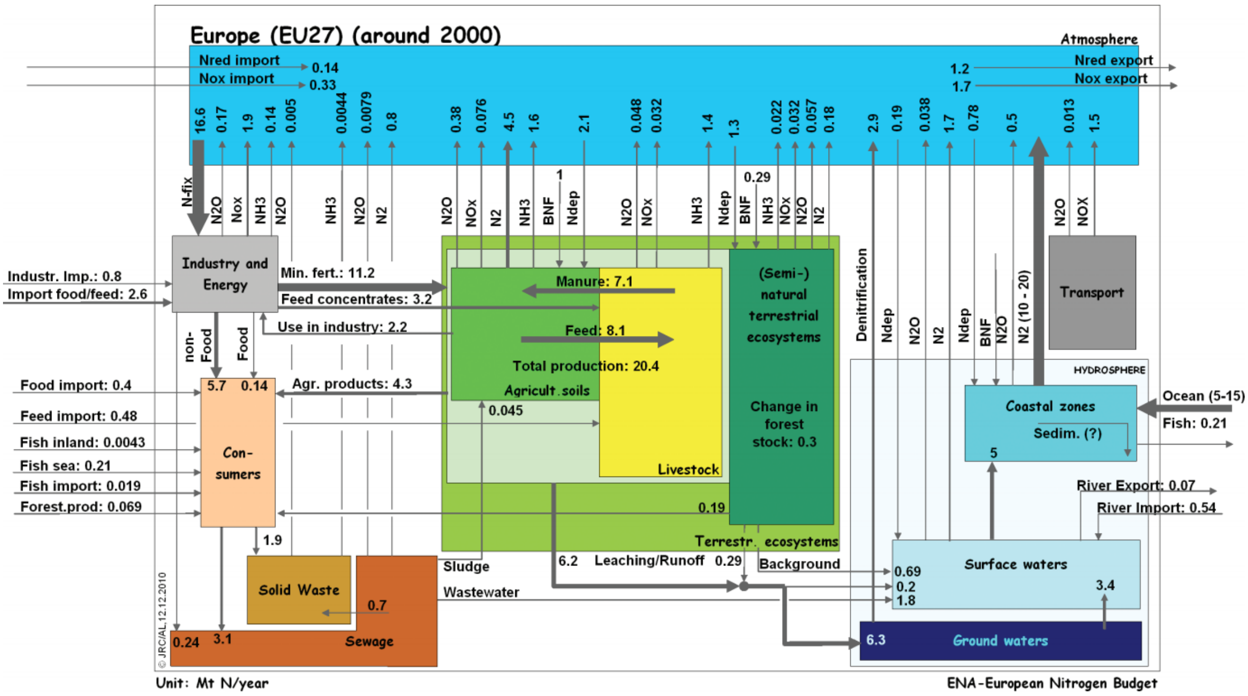


Figure 3.2. Input-output nitrogen (N) budget for Europe. Source: Leip et al. 2011. See details in the INMS Guidance Document on Development of National Nitrogen Budget Approaches (Winiwarter et al. 2025) and in the earlier UNECE Guidance Document (ECE/EB.AIR/119, UNECE 2013). © Cambridge University Press 2011.

3.3 Nitrogen Footprint approaches

The 'Nitrogen Footprint' is a metric that connects resource consumption with the associated N losses to the environment (Leach et al. 2012). Nitrogen footprints have been calculated at a variety of spatial and temporal scales (Table 3.2), and there are four existing methods typically used:

- N-Calculator (Leach et al. 2012)
- N-Input (Shindo and Yanagawa 2017)
- N-Output (Eguchi & Hirano 2019)
- N-Multi-region (Oita et al. 2016a).

Other methods that have been utilised to assess N footprints include:

- Coupled Human and Nature Systems (CHANS), a mass-flow model (Gu et al., 2013).
- Input-Output Material Flow Analysis (IO-MFA), which is related to the N-Multi-region method. This can examine both embodied N emissions and content flows between economic sectors that can calculate nitrogen use efficiency of economic sectors (Chapter 2.5; Oita et al. 2021; Katagiri 2018).
- Reactive Nitrogen Spatial Intensity (NrSI). This provides an indicator that can be used to calculate the spatial intensity of N footprints per settled area of country N footprints (Liang et al. 2018).

Nitrogen footprint approaches can be used for research to track trends over time, compare across different entities, and identify N reduction strategies. Some N footprint methods can also assess pressure indicators (e.g., N-Calculator, N-Input, N-Output, NrSI). Nitrogen footprint approaches can be used to establish N sustainability goals and track progress toward those goals (see Chapter 4).

Table 3.2. Methods to determine Nitrogen Footprints with various purposes. Bottom-up approaches use process-based data, and top-down approaches use statistics at a national level or the level of the target region.

Methods	Approach	Purpose	Item	Scale	Reference
N-Calculator	Bottom-up	Connection between consumer & N loss to the environment	Food & energy (i.e., transportation, housing, goods and service production)	Individual, institution, watershed	Leach et al. 2012
Coupled Human and Nature Systems (CHANS)	Top-down	Quantify different sources to the overall N footprint & connect to the country's N budget	Food, energy, & non-food	Country	Gu et al. 2013, 2015
N-Input	Top-down	Quantify the impact of food trade	Food	Country	Shindo & Yanagawa 2017
N-Output	Top-down	Quantify the N loss in domestic & food exporting countries separately	Food	Country	Eguchi & Hirano 2019
N-Multi-region	Top-down	Elucidate N loss (N-Multi-region) in economic sectors along supply chains across in-country regions or countries.	Food, non-food goods, & energy (all economic sectors)	Region, country, global	Oita et al. 2016a
Input–Output Material Flow Analysis (IO-MFA)	Top-down	Elucidate N loss and N content (IO-MFA)	Food, non-food goods, & energy (all economic sectors)	Region, country, global	Oita et al. 2021
Reactive Nitrogen Spatial Intensity (NrSI)	Bottom-up	Determine spatial intensity of the N loss by the N footprint	Food	Farm, watershed, country	Liang et al. 2018

3.3.1 Method descriptions

In this section, the methodology is described for the following seven nitrogen footprint approaches:

N-Calculator method

The N-Calculator N footprint methodology consists of two parts: Food and energy (Leach et al. 2012). The food N footprint considers food production and consumption steps with nutrient utilisation and recycling ratios, using Virtual Nitrogen Factors (VNFs) as summary indicators. A VNF for each food item is defined as N losses to the environment during food production including food processing and food loss divided by N included as a constituent of consumed food (Liang et al. 2021). The energy N footprint takes account of N emissions for energy use at home, personal transportation and purchase of goods and services by consumers and is complemented by emission data of a domestic input-output (IO) table.

N-Input method

The N-Input method is a top-down approach for estimating a country-wide N footprint. This method considers the N footprint of a country of interest, accounting for new N created in that country as well as N imported from other countries. The N-Input method accounts for new N inputs from chemical fertilizers and biological N fixation in the target country (i.e., direct N inputs) and new N inputs imported to a target country from exporting countries (i.e., indirect N inputs) (Figure 3.4; Shindo & Yanagawa 2017).

N-Output method

The food N footprint (i.e., N output during food production-consumption process) in a food-importing country can be regarded as the sum of N output to the domestic environment (direct N output) and that to the foreign environment in the food-exporting countries (indirect N output). The N-Output method (Eguchi & Hirano 2019) is a top-down approach that focuses on the other side of the N-Input method (Shindo & Yanagawa 2017), i.e., separately evaluating the direct and indirect N outputs. The direct N output is equal to the N balance (surplus N) that is calculated from the conventional material flow analysis for the agro-food system of a target country of analysis. The indirect N output is estimated for importing food and feedstuffs based on the concept of N-Input method or that of N-Calculator method with VNFs derived by the N-Input method calculation (dVNFs).

N-Multi-region method

The N-Multi-region method is an N footprint model for global analysis that takes account of complete supply chains and consumption, based on an environmentally extended quantitative economic model based on input-output (IO) analysis. These IO tables have been often extended to describe energy flows with generated pollutants, including N oxide (NO_x) emissions, such as for embodied energy and emission intensity in Japan (Nansai et al. 2012). The N-Multi-region method integrates a global Multi-Region Input–Output (MRIO) table and not only N emissions from energy and industrial production, but also N emissions in relation to N input for both food and non-food goods.

Input–Output Material Flow Analysis method

Input Output Material Flow Analysis (IO-MFA) is another technique based on the IO method for estimating the flows of target materials (e.g. different forms of N) among economic activities (Oita et al. 2021; Matsubae & Nagasaka 2015). IO-MFA can comprehensively trace the flow-paths of multiple target materials in inter-industrial transactions to final destinations described in an IO table. At the same time, this method considers the generation of losses and the need for elimination of immaterial transactions from the original monetary IO table.

By using physical IO tables, waste input–output material flow analysis (WIO-MFA), provides a higher resolution of IO-based material flow than other IO-MFA and can provide material composition of products present in the IO table (Nakamura et al. 2007).

Reactive Nitrogen Spatial Intensity method

A new indicator, Reactive N Spatial Intensity (NrSI), was developed to directly connect human resource-use to environmental impacts of the spatial distribution of N losses (Liang et al. 2018). The methodology starts with the N-Calculator framework for the N footprint and incorporates land area. The NrSI was calculated for two systems with contrasting land uses within a country: An agricultural system with edible crop, meat, and dairy production, and a settled system with residential areas, industrial infrastructure and transport-works in cities and towns). The NrSI framework maps the geographical locations of anthropogenic N losses, while the metric reports the N loss per unit area per year (units commonly used are kg N $ha^{-1}yr^{-1}$).

3.3.2 Applications and discussion

In this section, the applications of the following seven nitrogen footprint approaches are discussed.

N-Calculator applications

Nitrogen footprint tools based on the N-Calculator method have been developed for individual consumers, meals, institutions of higher education, communities and watersheds (Leach et al. 2012, 2016; Castner et al. 2017; Dukes et al. 2020). Nitrogen footprints can also be connected to other environmental footprints (and social indicators like N damage costs and income (Leach et al. 2016, 2017; Oita et al. 2020; Compton et al. 2017; Dukes et al. 2020). As with any other index, there is uncertainty in calculation and application, but, when used comparatively, N footprint tools provide valuable information.

As of 2021, consumer N footprint tools have been developed for 10 countries: Australia, Austria, Germany, Japan, the Netherlands, Portugal, Taiwan, Tanzania, the United Kingdom and the United States (Figure 3.3; Liang et al. 2016; Pierer et al. 2014; Shibata et al. 2014; Oita et al. 2018; Hayashi et al. 2018; Eguchi and Hirano 2019; Oita et al. 2020; Hutton et al. 2017; Stevens et al. 2014; Leach et al. 2012). In addition, the N-Calculator method has been applied to China, Egypt, India, 40 Sub-Saharan African countries and Yemen (Gu et al. 2013; Guo et al. 2017; Oita et al. 2020; Elrys et al. 2019, 2021; Dhar et al. 2021; Alnadari et al. 2021). These tools are country-specific to account for different consumption patterns, food production methods and fuel types in each country.

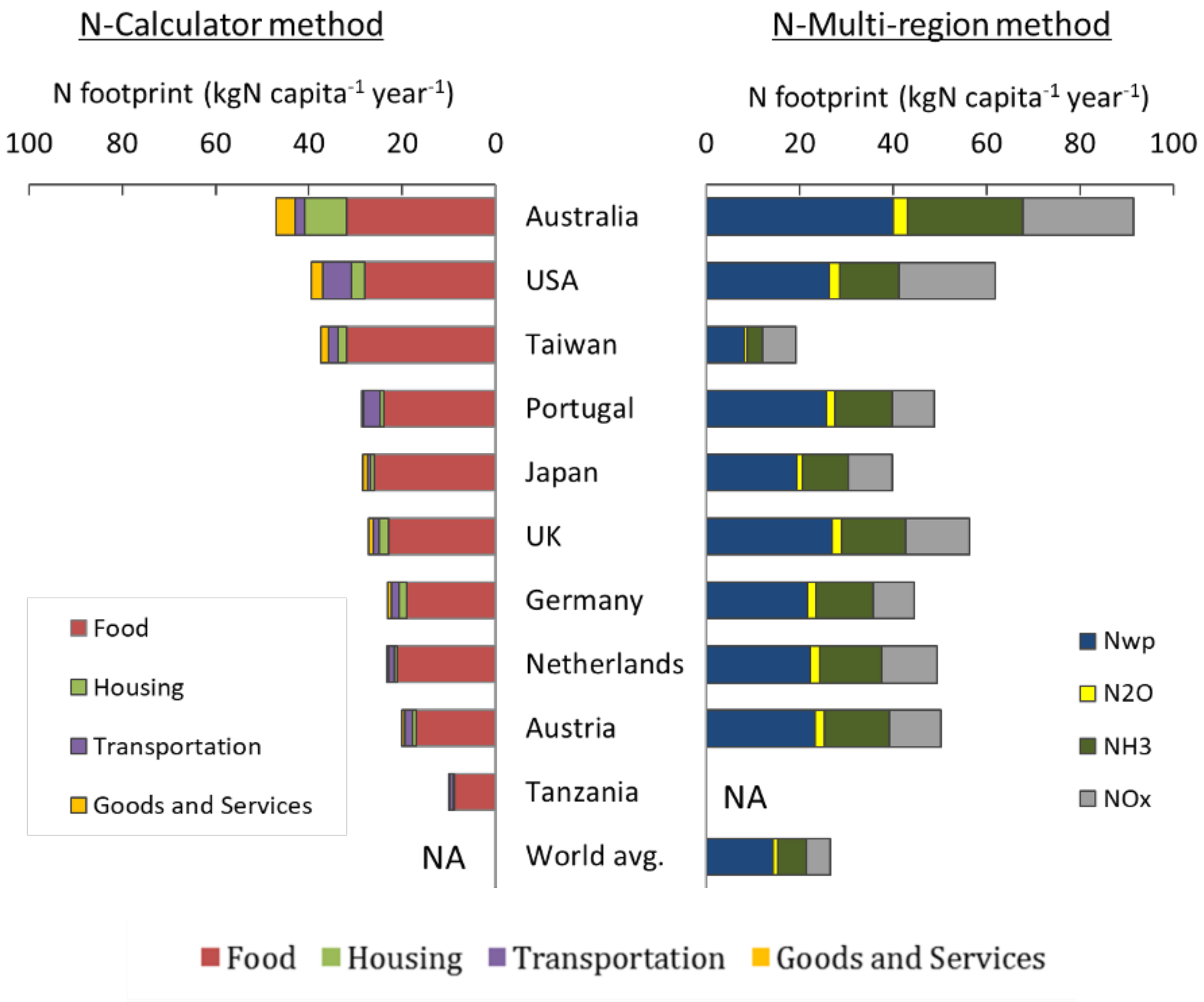


Figure 3.3. Comparison of the N-Calculator method (Shibata et al. 2017) and the N-Multi-region method (Oita et al. 2016a) on per capita annual N footprints. Nwp is reactive nitrogen potentially lost to water bodies, mainly being NO_3^-. Original graphic produced for this document © UKCEH 2025.

The campus N footprint tool was designed for institutions of higher education (Leach et al. 2013, 2017) and can be used to set a reduction goal (e.g., 25% by the year 2025 at the University of Virginia) (Castner et al. 2017). The campus N footprint was integrated with the campus carbon footprint in a web-based tool: SIMAP (Sustainability Indicator Management and Analysis Platform; unhsimap.org). A community N footprint model was developed for Baltimore City, Maryland (Dukes et al. 2020), and it expands the scope of an N footprint by including all resource consumption within that community, such as individual consumers, business and municipalities. The analysis used U.S. census survey data sets for consumption patterns. Another type of individual consumer N footprint is the watershed N footprint, which assesses how much of a consumer's N footprint makes it to a body of water (https://secure.cbf.org/site/SPageNavigator/bay_footprint.html).

N-Input applications

The N-Input method was developed to estimate national average Virtual Nitrogen Factors (VNFs) of various food items using a top-down approach by calculating VNFs of the target country and VNFs from the countries from which N-containing foods are imported (Shindo & Yanagawa 2017). Using the N-Input method, the Japanese food N footprint was estimated as 16.5–18.1 kg N capita^{-1} yr^{-1}, consisting of one-third direct N input (approx. 5.5 kg N capita^{-1} yr^{-1}) and two-thirds indirect N input. The N footprint estimate is at around the lower end of the range of the values of Japan estimated by the N-Calculator (15.2–28.1 kg N capita^{-1} yr^{-1}; Figure 3.4; Shibata et al. 2014; Oita et al. 2018; Hayashi et al. 2018; Eguchi & Hirano 2019). The advantage of the N-Input method is that it is based on accessible published data, making it easier to apply to countries that import large amounts of food.

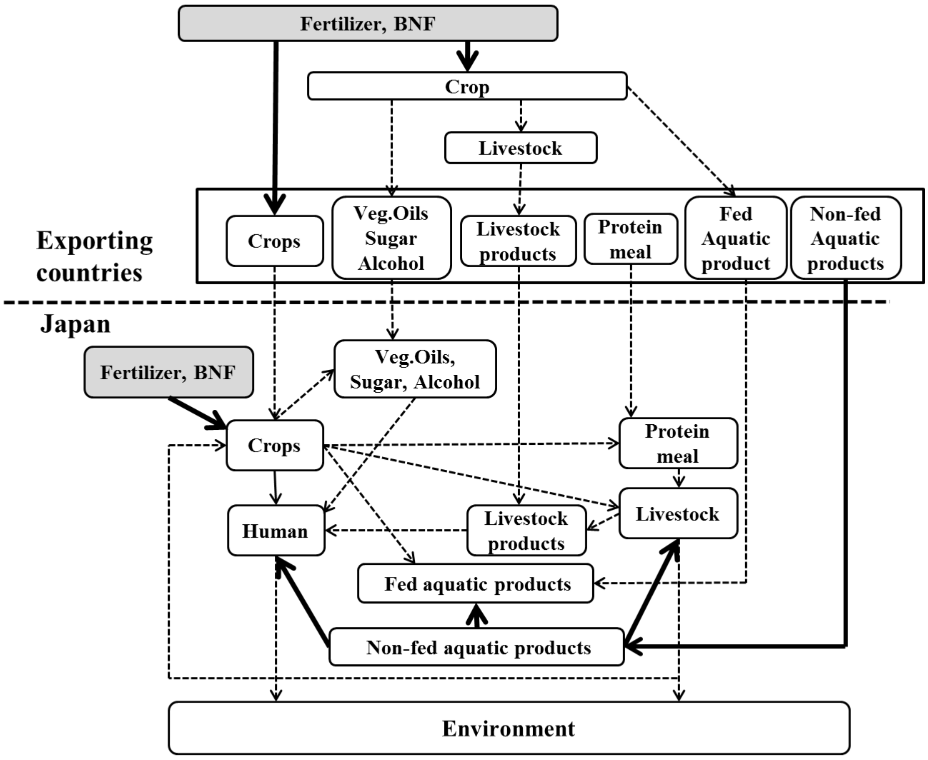


Figure 3.4. N-Input method: Nitrogen flows of food production and consumption. Bold arrows represent external nitrogen inputs evaluated in the N-input method. Biological N Fixation (BNF). Source: Redrawn from Shindo and Yanagawa (2017).

N-Output applications

The estimated food N footprint of Japan using the N-Output method was 21.4–23.8 kg N capita^{-1} yr^{-1} (Eguchi & Hirano 2019) during 1995–2007, consisting of direct N output of about 60% (13.5–14.8 kg N capita^{-1} yr^{-1}) and indirect N output of about 40%. The estimation is larger than or comparable to those using the N-Calculator approach (15.2–28.1 kg N capita^{-1} yr^{-1}; Shibata et al. 2014; Oita et al. 2018; Hayashi et al. 2018; Eguchi & Hirano 2019). The difference seems to be mainly due to the estimates of livestock and forage production efficiencies. The advantage of the N-Output method is in the accuracy of estimated N output to the domestic environment by using detailed national data while also estimating N output to the foreign environment in the food-exporting countries.

N-Multi-region applications

There are two approaches that consider international trade of embodied virtual N emissions/input: Bilateral trade-based and input-output (IO) based (Lassaletta et al. 2019). Bilateral trade-based approaches are sensitive to each imported food item (e.g., Burke et al. 2009; Oita et al. 2016b, 2020; Shindo & Yanagawa 2017; Shibata et al. 2014). This approach derives N emissions/input from the production of each food item or uses weighted average VNFs with self-sufficiency ratios to describe trade. An IO-based approach, the N-Multi-region method, was developed to overcome limitations of the bilateral trade-based approaches in analyzing the higher tier of supply chains. This method can be applied to connect to impact, such as eutrophication potential (Hamilton et al. 2018).

The N-Multi-region method estimated the global average N footprint at 27 kg N capita^{-1} yr^{-1}, ranging across 188 countries in 2010 from 7 kg N capita^{-1} yr^{-1} to 100 kg N capita^{-1} yr^{-1} (Figure. 3.5; Oita et al. 2016a). An advantage of the N-Multi-Region method is that it reveals flows of N emissions embodied in raw products along the supply chain, from the producing country to the countries of final consumption, enabling us to analyze economy-wide embodied virtual N emissions (Figure. 3.5; Oita et al. 2016a). It was found that top net importers are mostly developed countries, while top net exporters are mainly developing countries.

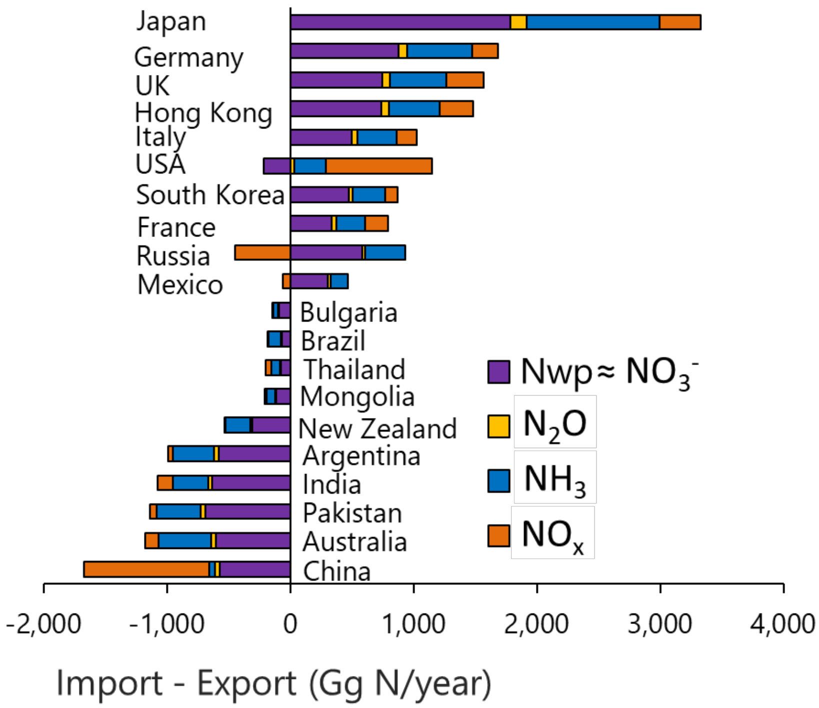


Figure 3.5. Top 10 net exporters and importers of embodied virtual nitrogen (N) emissions. Net exporters (shown bottom) pollute other countries through imported goods consumption, and net importers (shown top) are more polluted due to production of exporting goods. (Source: Oita et al. 2016a). Nwp is reactive nitrogen potentially lost to the water bodies, mainly NO_3^-. Copyright © 2016, Nature Publishing Group / Copyright © 2016, Royal Swedish Academy of Sciences.

Input-Output Material Flow Analysis (IO-MFA) applications

Although further research is needed to fully apply IO-MFA for N, some studies made early steps. One analysis demonstrated the detailed structure of the N cycle in the target economy (Singh et al. 2017), while another revealed representative compositions of the N sources physically embedded in a unit production for each industrial sector (Katagiri 2018). The nutrient-extended input-output (NutrIO) approach shows the N sources used in all economic sectors, including detailed agricultural sectors (Oita et al. 2021). An approach focused on solid waste materials WIO-MFA, has been applied to metals, plastics and carbon (Ohno et al. 2014, 2018; Nakamura et al. 2009). When applied to N flows, WIO-MFA can show the composition of different N species in products, such as newly fixed or recycled N. It can also be used to show different purity levels of N: For example, Purity of 99.9% ammonia is often enough for fertilizers, but semiconductors often need purity of 99.9999% or greater.

A global nitrogen IO-MFA could be performed, drawing on national N budgets of countries. This could have policy implications on the use of related technologies for N emissions from life cycles of food and energy in relation to industrial activities and household consumption.

Reactive Nitrogen Spatial Intensity (NrSI) applications

The Reactive Nitrogen Spatial Intensity (NrSI) approach has been applied to seven of the countries that have quantified their N footprint (Australia, Austria, Germany, Japan, the Netherlands, the United Kingdom and the United States; Figure 3.6; Liang et al. 2018). The pattern of NrSI among these nations was very different from that of their N footprint. Australia had the lowest NrSI (6 kg N ha^{-1} yr^{-1}), while the Netherlands had the highest (217 kg N ha^{-1} yr^{-1}).

The NrSI provides new insights into sustainable N management. It connects the total N loss with the land area over which it is lost, which can indicate the potential for environmental impacts, identify N emission hotspots and lead to management recommendations. Reducing total N footprints and improving nitrogen use efficiency (NUE) are important, but the connection between these actions and the environment should also be addressed. Given the pressure from a growing global population and increasing food and energy consumption, incorporating the NrSI into the N footprint framework is critical for identifying N pollution hotspot sectors and locations.

Further information on linking between efficiency and effect indicators is provided by a specific INMS guidance document (Leach et al. 2025).

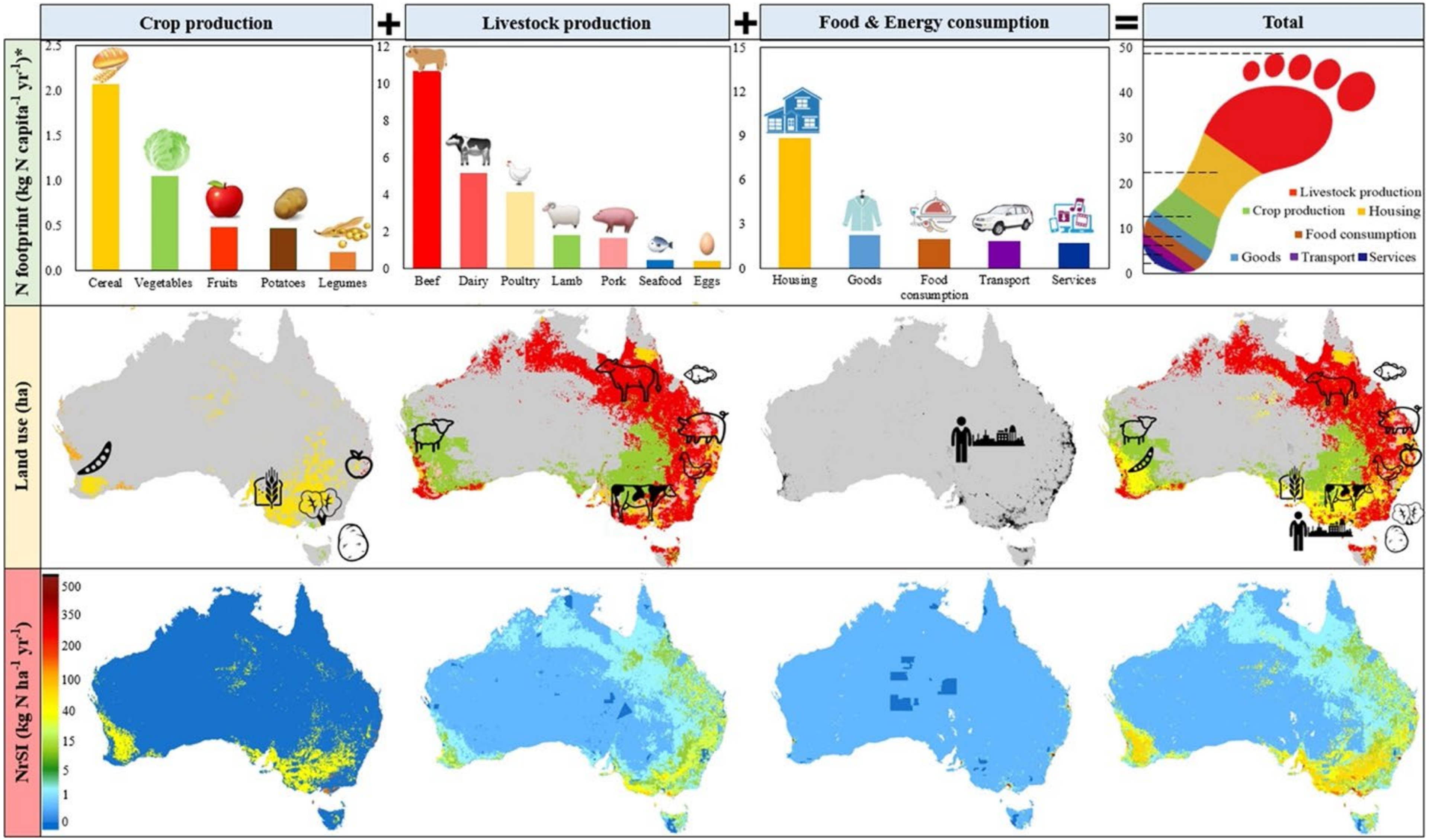


Figure 3.6. A simplified schematic representation of the information flow in calculating reactive nitrogen spatial intensity (NrSI). Nitrogen footprints (green row of boxes at top) quantify, per capita, N released from food and energy production and consumption to assess the contribution of human activities to N pollution (Leach et al. 2012; Liang et al. 2016). Combined with the land use of each system (yellow row of boxes in the middle), NrSI (red row of boxes at bottom) depicts, per area, N released from food and energy production and consumption to identify the hotspots of N pollution by human activities (Liang et al. 2018).

3.4 Nitrogen use efficiency (NUE)

3.4.1 Method descriptions

Nitrogen use efficiency (NUE) is a widely used indicator for evaluating the nitrogen performance of a system. NUE can be applied to agricultural systems (cropping, livestock, mixed systems and farms) and to evaluate food production–consumption chains, economic sectors, a specific region or a country. In general, NUE describes the ratio of N outputs to N inputs for a defined system. A higher NUE indicates higher efficiency and lower potential N losses, whereas a low NUE indicates lower efficiency and higher potential N losses. Since NUE is a ratio, it is important to report it with an indicator that can describe the magnitude of N use, such as N output or N surplus. The N surplus is calculated as the difference between the inputs and the outputs.

There are many ways to calculate crop NUE (Ladha et al. 2005), but at the cropping system or farm scale the output/input ratio is probably the most common (Lassaletta et al. 2014; Zhang et al. 2015; Quemada et al. 2020; Jones et al. 2025). For crop NUE, the inputs generally considered are synthetic fertilizers, manure, crop biological fixation and atmospheric N deposition. Outputs correspond to harvested crops and, if possible, removed crop residues (Oenema 2015). The N surplus can then be used as a proxy of the N that will be potentially emitted to the environment (as nitrate, ammonia, nitrous oxide, or non-reactive N_2) by N retained in the soil organic matter (Billen et al. 2013). A detailed description of NUE as a sustainability indicator of agro-food systems at different scales is provided in the INMS Guidance Document on nitrogen use efficiency indicators across multiple scales (Lassaletta and Sanz Cobena 2025) with an overview provided by Lassaletta et al. (2024). NUE is not directly a nitrogen pressure or impact indicator since it is a ratio that does not indicate the total flux of N released. However, NUE can be used to track efficiency changes over time and help inform pressure and impact indicators.

3.4.2 Applications and discussion

NUE can be calculated for any agricultural system, such as for a cropping system, livestock system, mixed used system or farm. NUE can also be calculated for a specific region or country and for the economy as a whole (Lassaletta and Sanz-Cobena 2025). Improvement of the NUE in cropping systems is desirable and generally associated with a decrease of N surpluses and their associated environmental impacts. A higher NUE also implies greater economic value, since more of the inputs go toward their intended use instead of being wasted (Sutton et al. 2021). Similar NUE figures with different levels of inputs and outputs are common. It is therefore important to consider all aspects to properly evaluate sustainability and thus to design efficient agro-environmental strategies capable of reducing impacts.

For example, in Spain a wide range of NUEs and surpluses were found for rainfed and irrigated barley and maize cropping systems at the regional level (Figure 3.7; Lassaletta et al. 2019). For barley, the NUE ranges from 85% to less than 10%. A reference maximum surplus value of 80 ± 20 kg N ha^{-1} yr^{-1} was used, which must be adapted to local situations (Oenema 2015; De Vries et al. 2013). In most regions, barley performance can be an example for both the environmental and agronomic perspectives (low surpluses and adequate yields). There were also situations where similar NUE values produced very different surpluses. In other cases, similar surpluses above the threshold were associated with different NUEs and yields. In those cases, higher NUEs can be linked to better yields, but not necessarily with acceptable environmental impacts. In the case of maize, most regions are above the reference surplus value. It is clearly shown how better NUEs are associated with better performance, but again many intermediate situations were found.

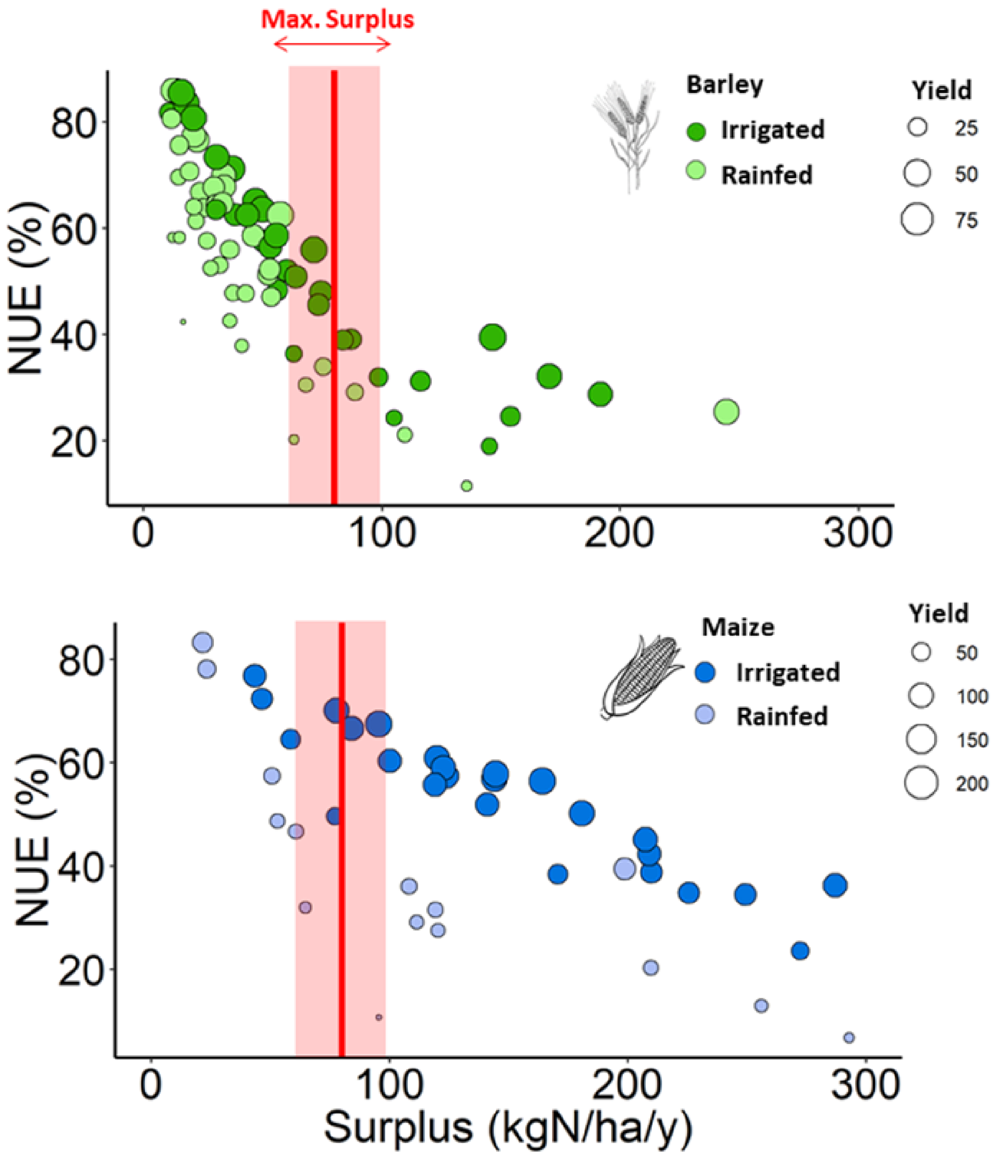


Figure 3.7. Relationship between nitrogen (N) surpluses, nitrogen use efficiency (NUE) and crop yield for rainfed and irrigated barley and maize crops in Spanish regions in 2008. The red line represents a reference maximum surplus that should not be crossed, here set at 80 kg N ha^{-1} y^{-1}. This threshold is only illustrative and must be adapted to local conditions based on potential impacts. The size of the circle represents yield expressed as kg N $ha^{-1}y^{-1}$. Original graphic produced for this document © UKCEH 2025.

The NUE approach can be extended to the national level to analyze how countries have evolved over the last 60 years following contrasted pathways (Figure 3.8; Lassaletta et al. 2016; Zhang et al. 2015). For example, in the Netherlands important surplus reductions are estimated related to yield and NUE increases. This improvement has not yet achieved a good environmental performance, because water and air pollution are still a problem in many areas of the country (van Grinsven et al. 2019). Egypt followed an inverse trend with increasing inputs that produced lower yield responses with dramatic NUE decreases and remains very far from a sustainable production system (Elrys et al. 2019). In contrast, in France, after a period of environmental degradation, improved performance has pushed the country to a much better agro-environmental condition. This general situation is not observed in some intensively managed areas where excessive surplus continues to impair water quality (Romero et al. 2016). Finally, NUE in Brazil has stagnated, and substantial intensification with slow yield responses has produced a significant rise in N surpluses.

In conclusion, independently of the considered scale, NUE is a useful indicator to monitor the resource and environmental performance of a given system or to benchmark comparable systems. NUE improvements are therefore positive from both agronomic and environmental perspectives. However, the picture is not complete if surplus (impacts) and yields (benefits) are not analyzed at the same time. Nitrogen use efficiency has an optimum range: Values too low exacerbate losses and higher values risk mining of soil N stocks (Oenema 2015).

Examples of NUE over different system boundaries, such as for crops, livestock, food chain and the overall economy are provided in the INMS Guidance document on nitrogen use efficiency indicators across multiple scales (Lassaletta and Sanz-Cobena 2025).

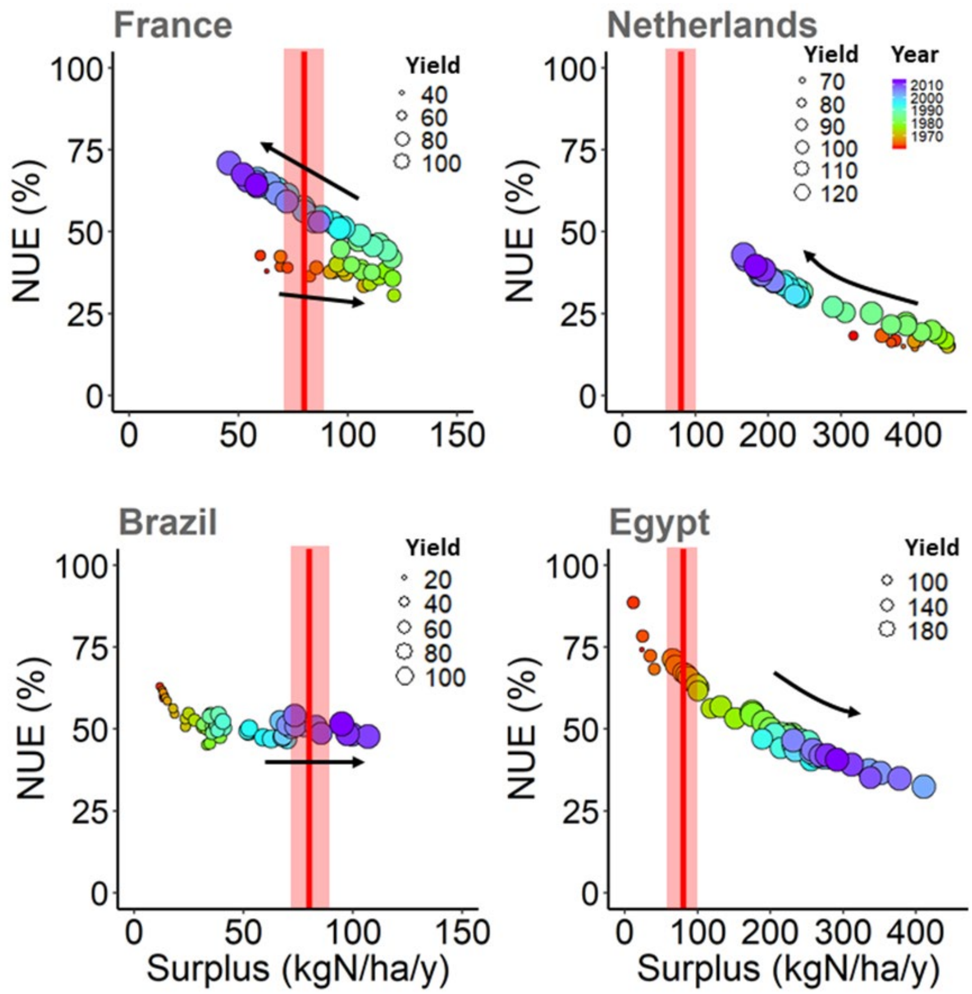


Figure 3.8. Historical evolution (1960-2013) of crop nitrogen use efficiency (NUE), surplus and yield in four contrasting countries (aggregated figure including all the cropping systems). These values represent a weighted country average; therefore, even if they indicate a general country trend, they do not represent the reality of local or regional areas where situations can vary. The red line illustrates a reference maximum surplus that should not be exceeded, here set at 80 kg N ha^{-1} y^{-1}. This threshold is only illustrative and must be adapted to local conditions based on potential impacts. The size of the circle represents yield expressed as kg N ha^{-1} y^{-1} (methods to prepare these figures are based on Lassaletta et al. 2016) Original graphic produced for this document © UKCEH 2025.

3.5 Planetary boundaries

3.5.1 Method descriptions

The planetary boundary framework was developed to assess the risk of major global environmental stresses overwhelming the capability of Earth's socio-ecological systems to respond through resilience or resistance. The stresses include nitrogen (N) and phosphorus (P) pollution, climate change, biodiversity loss, ocean acidification, stratospheric ozone depletion, land-system changes, freshwater use, atmospheric aerosol loading and chemical pollution (Rockström et al. 2009; Steffen et al. 2015).

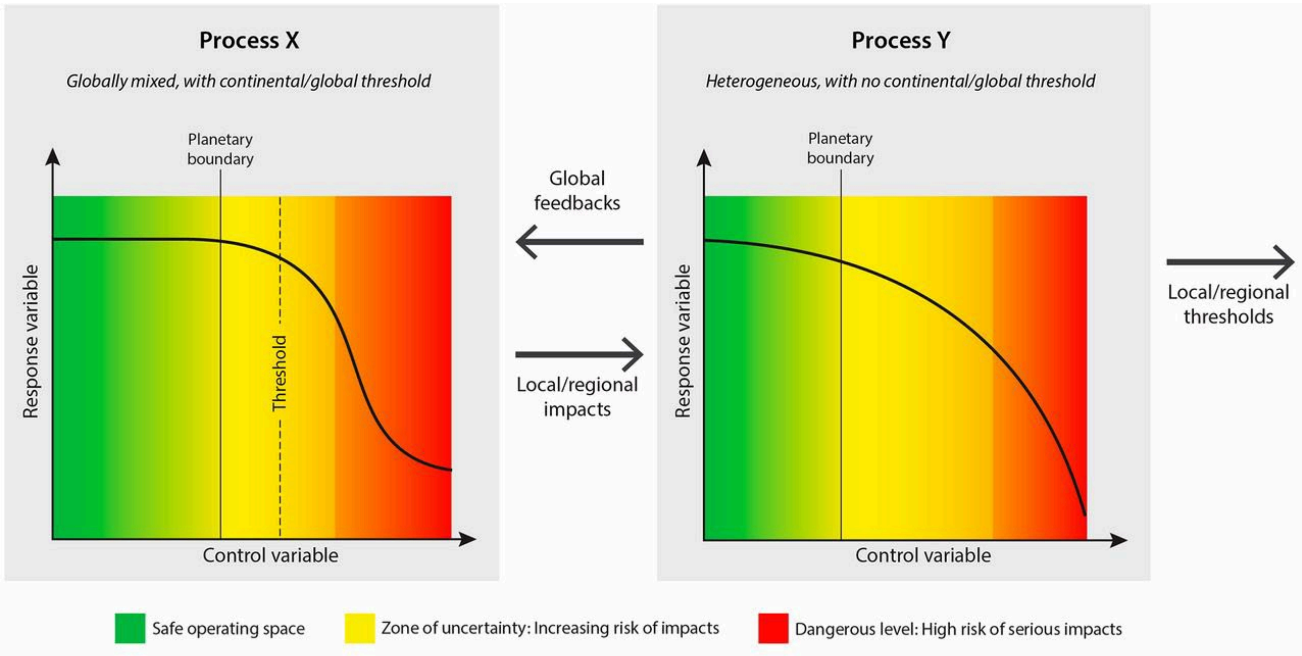


Figure 3.9. Conceptual framework for the planetary boundary approach, showing the 'safe operating space', the zone of uncertainty, the position of the threshold (where one is likely to exist), and the area of high risk (Steffen et al. 2015). Copyright © 2015, American Association for the Advancement of Science.

For each stress, planetary boundaries indicate the hypothetical threshold of impact or 'safe operating space' below which there is minimal environmental impact for the entire planet. Some of these thresholds have been quantified, while others have not. Ranges are identified for the 'zone of uncertainty (increasing risk)' and 'beyond zone of uncertainty (high risk)' (Figures 3.9 & 3.10). Biodiversity loss and nitrogen pollution (along with phosphorus pollution) are identified as the only two environmental issues already considered high risk (Figure 3.10).

The concept of deriving planetary N boundaries has been challenged due to the spatial variability of N impacts, both in terms of N limitation and N overuse (Rockström et al.2009; De Vries et al. 2013). These authors thus derived a planetary N boundary based on several factors:

i. Identification of multiple threat N indicators and setting critical limits for them.
ii. Back-calculating critical N losses from critical limits for N indicators while accounting for the spatial variability of indicators and their exceedance.
iii. Back-calculating critical N fixation rates from critical N losses.

The included indicators of this approach by Steffen et al. (2015) were:

i. Atmospheric NH_3 concentrations in view of adverse biodiversity impacts above 'critical levels' (see Section 3.6) of 1 μg m^{-3} and 3 μg m^{-3} for impacts on lichens and higher plants.

ii. Nitrate (NO_3^-) concentrations in groundwater related to health effects according to the World Health Organization drinking water limit of 50 mg NO_3^- l^{-1} or 11.3 mg NO_3^--N l^{-1}.

iii. Dissolved inorganic N concentrations in surface water to prevent aquatic ecosystems from developing eutrophication or acidification in the range of 1.0- 2.5 mg NO_3^--N l^{-1} (De Vries et al. 2013).

The results of the approach were better quantified and included in Steffen et al. 2015). Planetary boundaries have been useful in raising awareness and informing policy, but in relation to N with regional variation in DPSIR needs to be related to regional or national boundaries (Springmann et al. 2018; Willet et al. 2019). The N losses above planetary boundaries also interact with and impact other environmental planetary boundaries, such as climate change, biodiversity loss and stratospheric ozone depletion, which are all described in Chapters 3 or 4 (cf. Lade et al. 2020).

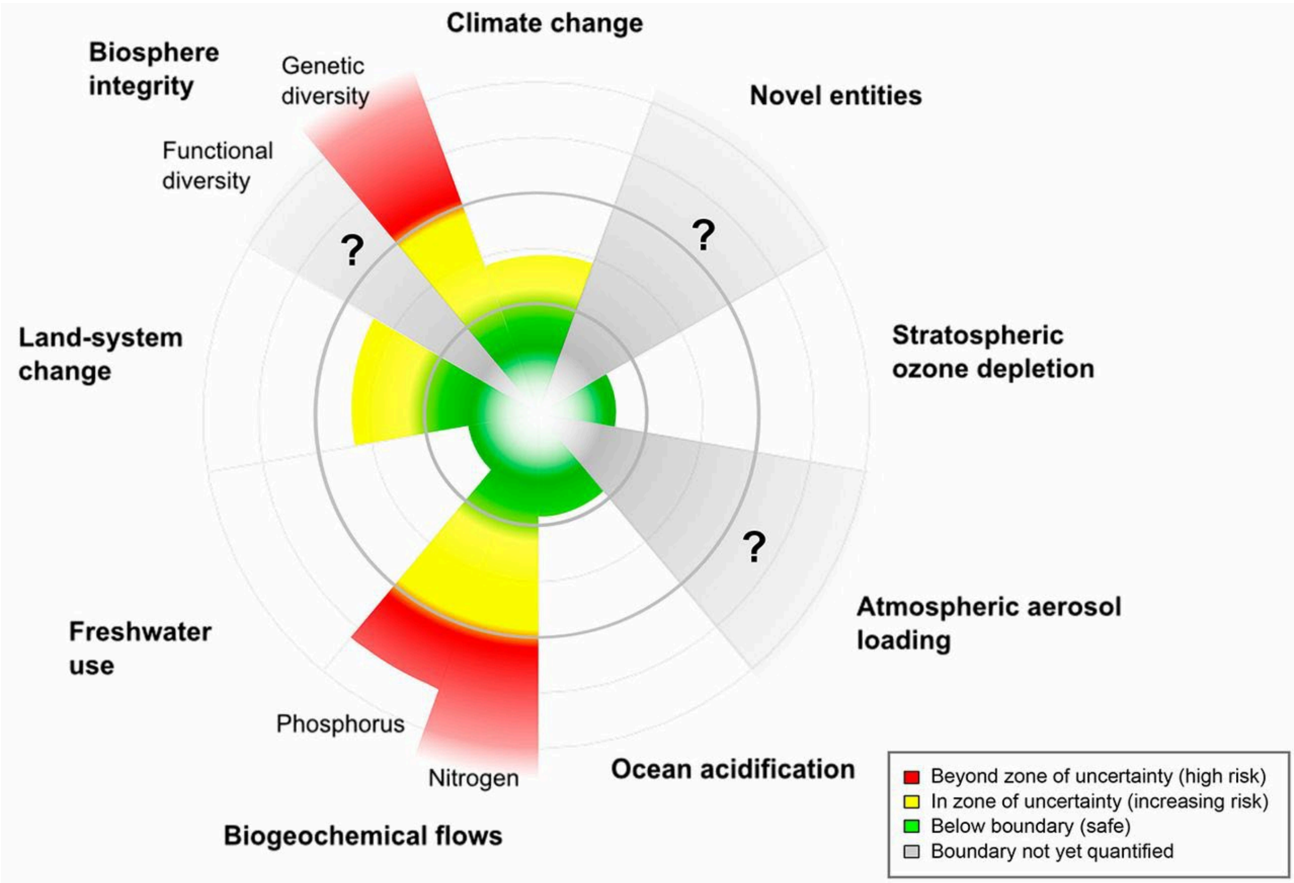


Figure 3.10. Illustration of the exceedance of nine planetary boundaries. The inner green shading indicates the proposed 'safe operating space' for the different systems. The red sectors represent an estimate of the present position for each variable (Steffen et al. 2015). Those authors emphasised that proposed phosphorus (P) and nitrogen (N) boundaries may be larger for an optimal allocation of Nand P around the globe. Copyright © 2015, American Association for the Advancement of Science.

3.5.2 Applications and discussion

Planetary boundaries can be used as targets or as thresholds for a goal for a given environmental issue. The threshold value of global N flow was originally set by Rockström et al. (2009) at 62 Tg N yr^{-1} (with the range of uncertainty of 62–82 Tg N yr^{-1}), intended to act as a global 'valve' limiting introduction of new reactive N to the Earth system (Steffen et al. 2015). The current estimate of the global N flow by these authors is ~150 Tg N yr^{-1}, which far exceeds the safe operating space of nitrogen. The spatial distribution of fertilizer N is also critical for impacts of N locally and regionally. The planetary boundary framework is a useful approach for broadening public understanding and for policy makers to describe the relative threats posed by different environmental systems.

Planetary boundaries were originally applied at the global scale, but subsequent analyses have identified nitrogen boundaries for smaller regions, such as a national level (Geupel et al. 2021). One analysis used a spatially explicit N balance model to calculate where agricultural N losses within EU27 currently led to an exceedance of critical N emissions (De Vries & Schulte-Uebbing 2020; Figure 3.11). Those authors considered adverse impacts on terrestrial ecosystems, critical N concentrations in runoff to surface water in view of eutrophication impacts and critical nitrate (NO_3^-) concentrations in groundwater in view of current EU environmental legislation on drinking water quality. They then calculated the N inputs at which critical N emissions or concentrations are not exceeded ('critical' N inputs). Actual and critical N inputs were calculated for approximately 40,000 unique soil-slope-climate combinations throughout the European Union. This identified areas where significant efforts to reduce environmental impacts are needed. When actual N inputs exceeded critical inputs, they calculated the necessary reduction in ammonia emission fractions and necessary increase in NUE to reconcile current food production while also reaching air and water quality goals. This was also done for the target food production level (De Vries & Schulte-Uebbing 2020). A similar approach as for EU27 has recently been applied at global scale, showing large variations in areas where N inputs can be increased, to close yield gaps in regions where environmental thresholds are not exceeded (e.g. large parts of Africa) and need to be reduced (Schulte-Uebbing et al. 2022). Recent studies highlight that only a combined action from the agricultural systems to consumption patterns will allow feeding an increasing population without trespassing several N boundaries (Springmann et al. 2018; Gerten et al. 2020).

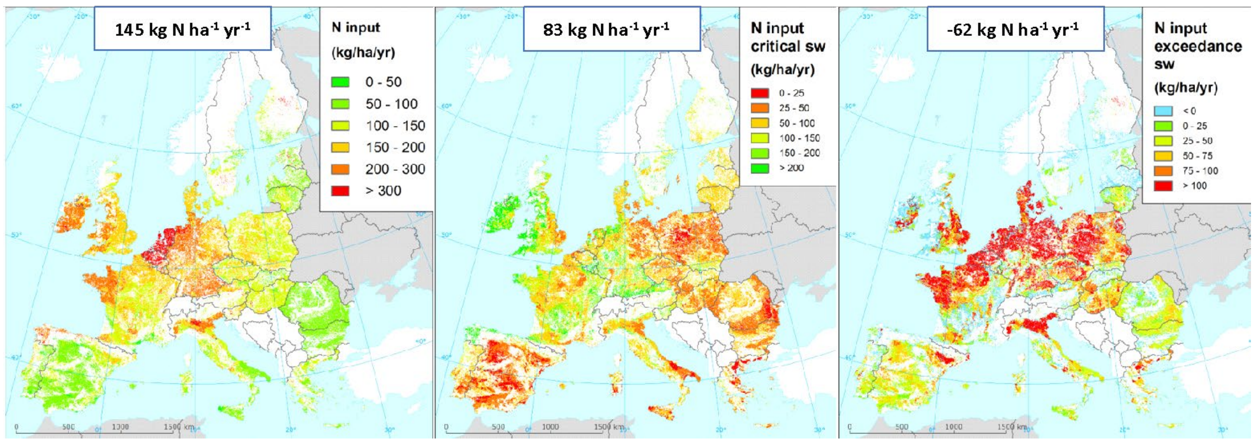


Figure 3.11. Geographic variation in actual nitrogen (N) inputs (left), critical N inputs in view of the protection of surface water quality (sw, middle) and the difference between actual and critical N inputs (right) as estimated by De Vries & Schulte-Uebbing (2020). Copyright © The Authors / International Fertilizer Society.

3.6 Critical loads and critical levels

3.6.1 Basic concepts and approaches

Critical loads and critical levels are useful scientific concepts that have been used since the 1990s to inform international policy development, especially within the UNECE Convention on Long-range Transboundary Pollution. These provide an indication of how much anthropogenic atmospheric nitrogen deposition (in the case of critical loads) and concentrations of nitrogen gases (in the case of critical levels) that natural ecosystems can tolerate before negative impacts begin to occur. Critical loads can also be applied for soil leaching and river loads.

A critical load is formally defined as the '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' (Nilsson & Grennfelt 1988). Conceptually, a critical load represents a numerical estimate of a threshold. In its policy application in Europe, the critical load is generally taken to be the threshold below which the most sensitive parts of the ecosystem are safe from damage. While in the United States, the same concept is most often applied for species rather than ecosystems. This threshold can be for any ecological endpoint of concern to researchers or decision makers, such as changes in soil acid neutralizing capacity (ANC) conditions, the onset of leaching, reductions in plant biodiversity, or anthropogenic eutrophication of water bodies (Blett et al. 2014; CLTRAP 2004). Historically, critical loads have been used in the context of atmospheric deposition and associated effects on terrestrial and aquatic ecosystems. However, there are close analogies in the water quality domain that relate the concentrations of N in lakes, rivers, and streams — which may be primarily from discharge and runoff of point and nonpoint sources — to changes in the aquatic ecosystem.

A critical level is formally defined as "the concentration, cumulative exposure or cumulative stomatal flux of atmospheric pollutants above which direct adverse effects on sensitive vegetation may occur according to present knowledge" (CLRTAP 2017). Critical levels have been defined for different averaging time periods for both nitrogen oxides and ammonia (WHO 2000; Cape et al., 2009; Sutton et al., 2009) and are in use in support of the Gothenburg Protocol of the UNECE.

Critical loads can offer an advantage over critical levels in integrating contributions of different nitrogen compounds to atmospheric deposition. Conversely, critical levels offer the advantage that atmospheric concentrations are easier to measure than nitrogen deposition and allow assessment specifically of NH_3 and NO_x impacts. The current critical loads methodology does not take account of the fact that, per unit nitrogen input, NH_3 can be up to five times more damaging than other forms of N input (Sutton et al., 2020), and such differences are currently only accounted for by the use of critical levels. A limitation is that critical levels for other nitrogen gases have not been defined, such as for nitric acid (HNO_3) or nitrous acid (HNO_2).

For the purpose of providing an overview in this section, the following paragraphs mainly focus on methods and application of critical loads. Further information is provided by the UNECE Mapping Manual for critical loads and levels (CLRTAP 2017).

3.6.2. Methods for critical loads assessment

To assess critical loads for atmospheric deposition of N or acidity, an endpoint that is affected needs to be identified (e.g., plant species diversity and soil biodiversity). An endpoint can be defined as a 'specified sensitive element of the environment' that one wants to protect (Nilsson & Grennfelt 1988). To quantify impacts, a certain endpoint indicator is needed, which can be defined as 'a quantifiable measure describing the status of an endpoint'. In the case of geochemical indicators, this is also called a chemical criterion. Finally, to assess risk, a critical limit is needed, which can be defined as a maximum or minimum value allowed for an endpoint indicator or chemical criterion. For example, an endpoint for N impacts is vegetation change, and the endpoint indicator (or chemical criterion) is the dissolved N concentration. Its critical limit is in the range 2.5-6.0 mg N l^{-1}, and a study provides an overview of those indicators (De Vries et al. 2015).

Critical loads have been estimated in varied ways, but generally fall into four approaches: Manipulation experiments, spatial gradient studies, space-for-time substitution studies and dynamic simulation modeling (Blett et al. 2014; CLRTAP 2004).

Manipulation experiments are where researchers add specific amounts of additional N and record the ecological response. There have been hundreds of manipulation experiments globally, especially in the EU, United States, and Asia (Bobbink et al. 2010, 2011; Pardo et al. 2011). These have the benefit of isolating the source of the effect to elevated N, but they have limitations in that they are difficult and costly to implement over large scales and for many locations. Furthermore, there are uncertainties whether the form, timing and manner of manipulation accurately simulates atmospheric N deposition (Cao et al. 1998; Midolo et al. 2019 Pardo et al. 2011). Conversely, the explicit comparison of different nitrogen forms offers the potential to take account of differences between wet and dry deposition and oxidized and reduced nitrogen inputs (Sheppard et al. 2014; Sutton et al. 2020).

Empirical critical N loads are mainly based on manipulation experiments. Despite the strong empirical evidence, there are several drawbacks of empirical critical nitrogen loads. For example, the endpoints which they aim to protect vary between ecosystems. Moreover, they are generally (but not always) based on comparatively short-term experiments and thus provide limited knowledge of the evolution of impacts over time. Finally, the variation in critical loads for a given ecosystem is not spatially explicit but only included by expressing the critical load as a range (De Vries et al. 2010, 2015). This highlights the need for long-term experiments over multiple decades, which are essential to test whether the expectations by spatial studies and models match out in practice. For example, the Whim Bog experimental manipulation study (Levy et al. 2019) has over 20 years of data and incorporates a local (60 m) spatial gradient, but such experiments are very rare.

Spatial field studies assess impacts across a gradient of N deposition (i.e., 'gradient studies'; Stevens et al. 2010; Simkin et al. 2016; Horn et al. 2019; Clark et al. 2019). Gradient studies are much more limited in number than manipulation studies. They have the advantage of assessing the effect of actual N deposition. However, other factors may covary with the deposition gradient (especially when performed at larger spatial scales of several km and above), lowering confidence in the assessment of the response of interest. For example, ozone and sulphur deposition may spatially covary with N deposition, resulting in an identified, but spurious, N response that may be driven by sulphur or ozone (Horn et al. 2019).

Space-for-time-substitution studies have plots that have been sampled at some time in the past (presumably with lower deposition), and those same plots are then resampled currently (presumably with higher deposition; Smart et al. 2005; Bennie et al. 2006). The difference between the two time points is assumed to be due to deposition. These have the same advantages and disadvantages as gradient studies, but with the covariation problem across time rather than space.

Finally, **modeling** can be used to estimate critical loads (CLTRAP 2004; De Vries et al. 2010; McDonnel et al. 2020; Bonten et al. 2016). Dynamic models can be used that simulate biogeochemical processes, plant communities or both. **Dynamic models** can also enable an assessment of the risk of adverse effects over time and space, indicated by the exceedance of critical limits for specifically chosen endpoint indicators. Alternatively, steady-state, **mass-balance models** can be used that estimate charge- or nutrient-balances for critical loads of acidification and eutrophication, respectively (Schulze et al. 1989; De Vries et al. 1994b). The advantages of modeling include the flexibility in exploring various hypotheses that may be difficult or impossible to empirically assess in the field, and in pointing researchers toward new avenues that may be under-appreciated. The limitations of modeling are that the models themselves are built from our current understanding of how ecosystems work, and so while they may shed light on sensitivities that are difficult to empirically test in the field, they are limited by our current level of understanding.

3.6.3 Applications and discussion

Examples of critical loads across ecosystems and taxonomic groups

There are numerous examples of critical loads (CLs), including: Forest soil eutrophication and acidification; surface water eutrophication and acidification; losses in plant biodiversity and individual species; changes in soil geochemistry; increases in leaching; and changes in lichen community composition. It is beyond the scope of this guidance document to compile those in any comprehensive manner, but we provide a snapshot of their diversity. Many examples of CL assessments across Europe, the United States, Canada and China are presented in *Critical Loads and Dynamic Risk Assessments for Nitrogen, Acidity and Metals in Terrestrial and Aquatic Ecosystems* (De Vries et al. 2015).

Broadly speaking, species and processes that have faster growth rates (e.g., diatoms and soil bacteria compared with trees) or biogeochemical rates (e.g., soil nitrate concentrations compared with total soil N), respectively, respond more rapidly to changes in N deposition and thus have lower critical loads (Figure 3.12; CDPHE 2017). However, it should be noted that critical load is defined as a 'harmful effect' not as 'any effect' (Nilsson & Grennfelt 1988), meaning any reaction is not necessarily evidence that a CL has been reached (Nilsson & Grennfelt 1988). Furthermore, species with low critical loads are often relatively unresponsive to the added N and are thus outcompeted by fast-growing species. Though these broad generalisations hold true on average, there are still many instances of slow growing species of herbs and trees that are sensitive to low levels of N deposition, and diversity of critical loads should be considered (Figures 3.12 & 3.13; Wilkins et al. 2016; Horn et al. 2019).

Utility in policymaking and regulation

Both critical loads (CLs) and critical levels (CLEs) have seen wide application in Europe to support agreements under the Convention on Long Range Transboundary Air Pollution (CLRTAP; Figure 3.14). In Europe CLs are also widely used to assess effects on biodiversity. By comparison, CLEs for nitrogen have been set to protect the most sensitive vegetation (NH_3) or average vegetation (NO_x), and currently do not go far in describing differences between species (WHO 2000; Cape et al. 2009; CLRTAP 2017). This points to the need for further experimental and observational studies to characterize CLEs for more situations, including for different vegetation types globally.

There are national and European indicators of CL exceedance, and both CLs and CLEs are used as a basis to inform local licensing of activities in some countries; if a CL or CLE is predicted to be exceeded, a licence for a project, a plant or an installation can be denied. An example is the SCAIL model (Simple Calculation of Atmospheric Impact Limits), which provides an initial screening tool for such assessments using both CLs and CLs, prior to more detailed site base assessments (Theobald et al. 2009; SCAIL 2024) in support of the wider 'Air Pollution Information System' (APIS 2024). Although these on-line tools were developed for the UK, they also provide useful material relevant internationally.

Critical loads have also been applied in Canada under the Canada-United States Air Quality Agreement, but are used to assess exceedance of CLs for acidity only. Critical loads have seen less direct use in the United States at the national level, although some states (e.g., New York, Colorado) use them to support air quality decisions, and some federal agencies (e.g., National Park Service, United States Forest Service, Bureau of Land Management) use them to inform decisions about improving air quality in protected areas (Blett et al. 2014). Nonetheless, in the United States CLs are currently being considered for use in setting a national secondary standard for ecological effects and public welfare under the Clean Air Act (EPA 2018). Thus, critical loads are increasingly used as a policy instrument to help make informed decisions on air quality to prevent damage to ecological end points of concern to decision makers and the public.

Values of empirical critical loads for nitrogen and of critical levels for NO_x and NH_3 are given in Chapter 4 (Tables 4.2.2. and 4.2.3).

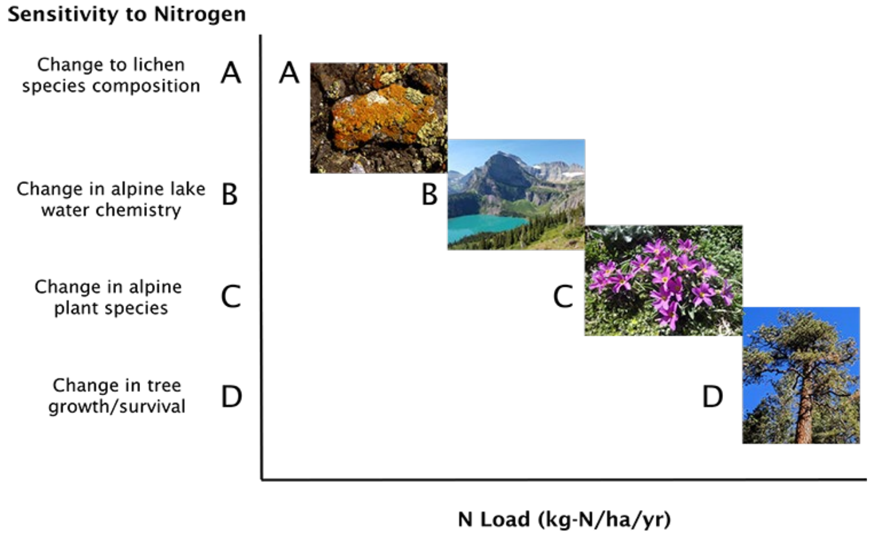


Figure 3.12. Conceptual diagram of how "fast processes" such as lichen communities respond more rapidly to nitrogen (N) deposition than "slow processes" such as trees (Based on data and concepts in Geiser et al. 2010; CDPHE 2017). Original graphic produced for this document © UKCEH 2025.

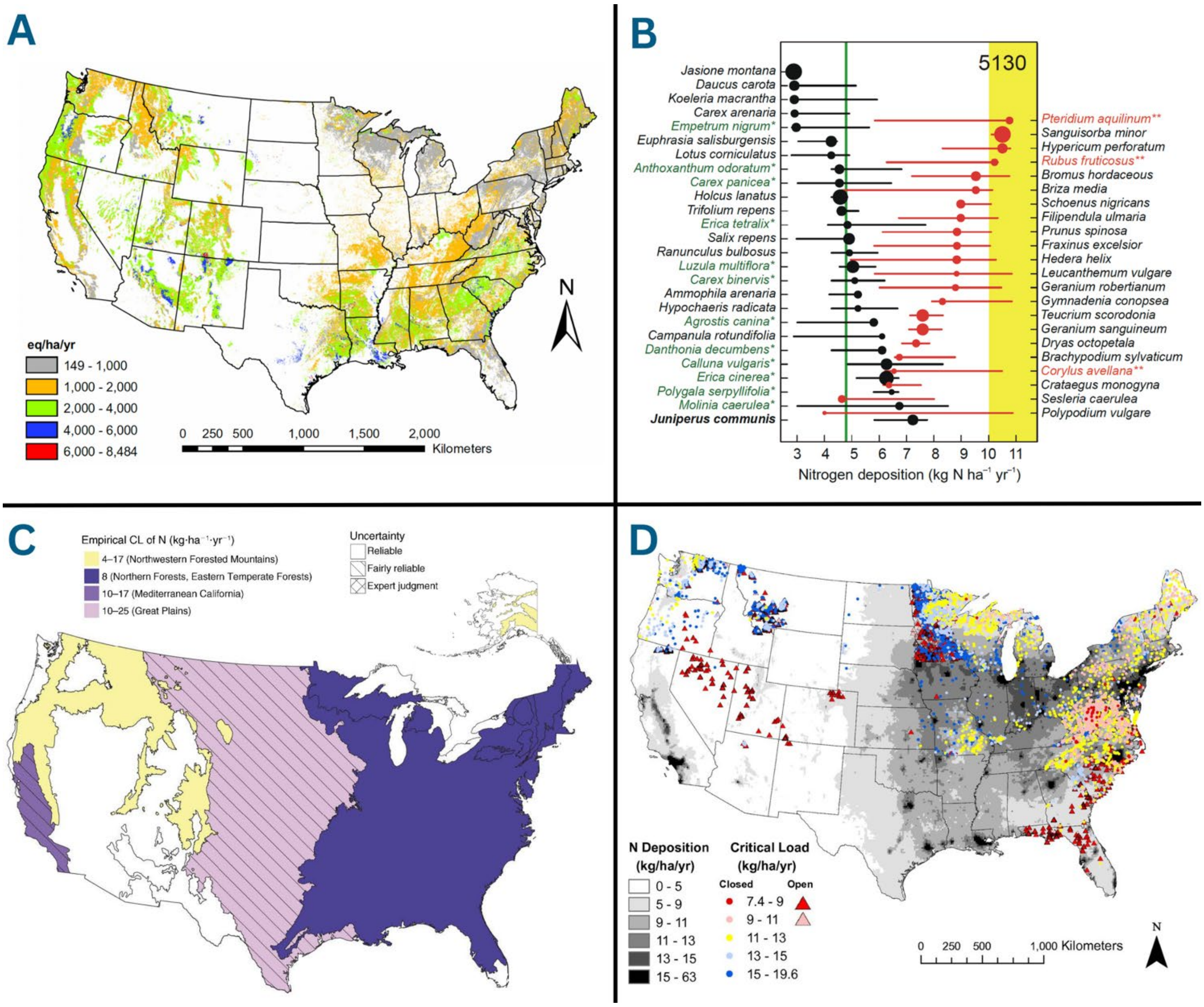


Figure 3.13. Examples of critical loads (CL) from various studies. **A**: Critical loads of forest soil acidification, which are influenced by nitrogen (N) and sulphur (S) and calculated from mass-balance-modeling (McNulty et al. 2007). Copyright © 2008 Elsevier Ltd. **B**: Empirical CLs for changes in abundance of different plant species for a habitat in Ireland using a TITAN analysis on plots across a gradient (Wilkins et al. 2016). Change points (dots) and bootstrapped intervals (whiskers) are shown separately for species that decrease (black), increase (red), or remain stable (green). The overall community change point (green line) and the historical estimated CL from earlier work are also shown (Bobbink et al. 2011). © 2016 The Authors. Published by Elsevier Ltd. **C**: Empirical CLs for nitrate leaching for Level 1 Ecoregions across the United States assessed with site-specific studies, then extrapolated to the ecoregion (hatching indicates confidence; Pardo et al. 2010). © 2011 by the Ecological Society of America. **D**: Empirical CLs for reductions in total species numbers for herbaceous communities of grasslands and forest understories, estimated using a gradient approach (Simkin et al. 2016). Copyright © 2016 National Academy of Sciences.

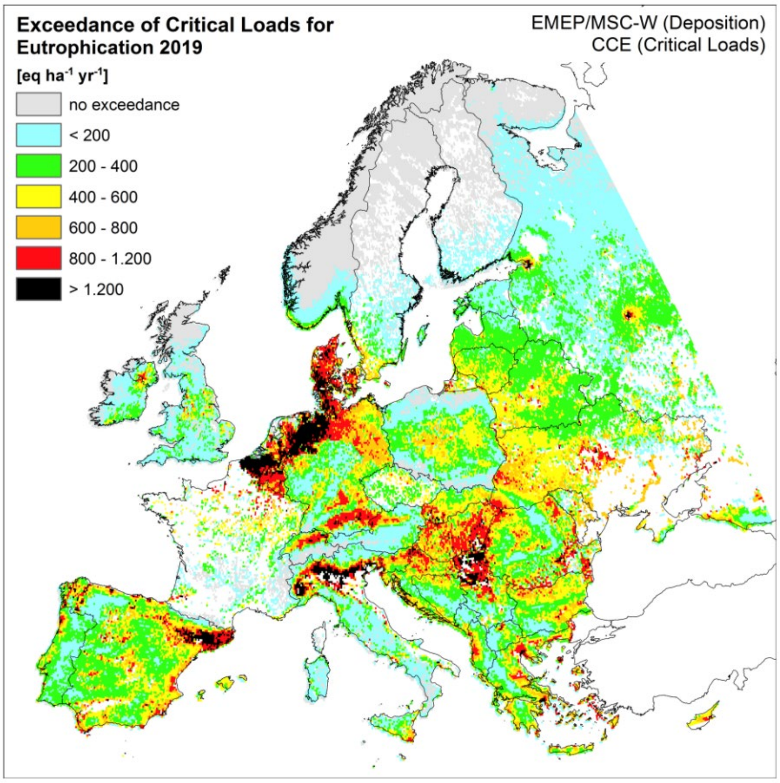


Figure 3.14. Exceedance of critical loads for eutrophication for 2019 (Fagerli et al. 2021).

Chapter 4: Assessment methods of nitrogen impacts

Introduction and general approaches for nitrogen impact assessments

This chapter includes specific descriptions drawn from the literature of how the positive and negative impacts of reactive nitrogen (N) in the environment are assessed. Specifically, this chapter focuses on state-impact linkages in the Driver, Pressure, State, Impact, Response framework (DPSIR; Figure 1.4). The narratives are purposefully short and include background of the impact; processes by which reactive N causes the impact, either by itself or interactively with other drivers; indicators of impact; and how the impact is measured. The descriptions below are related to the N-MIP - Nitrogen Matrix of Impacts and Pressures (see supplementary file "N-MIP matrix.xlsm").

There are a variety of methods to quantify pressure-state-impact relations: Experiments, monitoring over time, monitoring over space (i.e., gradient studies), meta-analysis of published papers, modeling and remote sensing. For human health impacts, epidemiological and longitudinal studies of people over time are described, along with models.

Experimental research most commonly involves adding different amounts and forms of N in liquid or solid form to croplands, natural systems, soils, or waters in controlled experiments with replication. Experiments can be conducted *in situ* or in controlled laboratory settings. The response is measured for the dependent variable, which can include plant biomass and abundance responses, soil carbon stocks and trace gas fluxes. Ecosystem parameters can also serve as independent variables, such as net primary productivity, plant species composition, invertebrate abundance, N mineralisation rate, respiration, soil chemistry change, nitrate leaching or gaseous emissions from soil. Experiments can also be conducted with ozone or other gases in chambers or free air enrichment experiments (FACE) plots to explore ecosystem responses to exposure.

Long-term and cross-site monitoring of N inputs (e.g., atmospheric deposition, fertilizer runoff, human waste and sewage) and N concentrations in aquatic systems are used to identify soil and water quality impacts. The results of monitoring can be compared to a measured or estimated baseline or to standard to determine deviation from background or desirable values. Monitoring of greenhouse gas (GHG) emissions from soils and waters can be achieved using chamber methods, flux gradient methods and eddy covariance techniques. Regional and country-level monitoring programs are used to assess the pollution level at which there are impacts on people, plants and materials, including for air quality (e.g., N oxide: NO_y, ammonia N: NH_x) and other air pollutants (e.g. ozone: O_3, sulphur oxides: SO_x). These programmes can also monitor how these levels change with regulation or other events that alter energy use or limit industrial activities due to restrictions on movements of workers (e.g. COVID-19). Monitoring of N concentration in food products is used to identify desirable or undesirable levels.

Gradient studies are most often used to detect changes in ecosystem properties and include field surveys along a gradient of N deposition. These have been used to identify patterns in biodiversity and soil quality across regions, nations or continents.

Meta-analyses are powerful ways of statistically summarizing the results of a number of isolated studies. They have been used to develop empirical functions of the pressure-impact relationship and to identify significant drivers and indicators that explain site-to-site differences.

Process-based and statistical models are useful for interpreting pressure-impact relations for all impacts described in this chapter. Models are valuable for developing hypotheses, testing hypotheses and projecting forward or backward in time. Such temporal comparisons aim to identify baseline conditions of resources or possible past/future conditions under different pressures. Models are applied at many different spatial scales. Parameterisation and validation with high-quality observed data is critical to reducing model uncertainties and increasing confidence in model outputs.

Satellite and aircraft remote sensing identify spatial patterns of eutrophication and related effects such as algal blooms induced by elevated N inputs to lakes, ponds, estuaries and coastal systems, on regional to global scales. Remote sensing with an advanced very-high-resolution radiometer (AVHRR) can be used to quantify N in terrestrial foliage. Satellite platforms provide an important means to quantify total columns of gaseous nitrogen compounds and have global scope, with absorption of NH_3 in the infrared range and absorption of NO_2 in ultraviolet and visible ranges. Such satellite measurements have been used to compare with or estimate spatial patterns of emissions (e.g., van Damme et al. 2018).

Human health effects are approached with epidemiological studies and individual clinical exposure studies. Epidemiology is the study of the distribution, causes and risk factors of health-related states and events. For example, epidemiological associations link exposure to air pollutants with the development of asthmatic and chronic inflammation in populations that have been exposed to certain levels of ozone (O_3), nitrogen dioxide (NO_2) or fine particulate matter ($PM_{2.5}$; i.e., inhalable particles with diameters 2.5 micrometers or less). The population exposure is the aggregate exposure for a specified group of people. The measure of the distribution of individual exposures typically includes a measure of the central tendency (e.g., mean exposure) and of its variability (e.g., variance). An accurate and statistically valid characterisation of population exposure may require personal exposure measurements. Air monitoring using fixed-site monitors and longitudinal (e.g., cohort) studies are required to assign causality, and biological measures can be used to assess effects. Questionnaires can be used for qualitative and (often retroactive) information.

4.1 Aquatic ecosystems

Increased loading of N has a range of impacts to water quality in coastal zones, fresh waters and drinking water, affecting both aquatic organisms and humans. Drivers that affect water directly influence human health, aquatic ecosystems and cultural services (Figure 1.4). The environmental N impacts associated with aquatic ecosystems are described in this section, and more detail can be found in the supplementary material (N-MIP):

- Eutrophication of marine and coastal environments (Sections 4.1.1. & 4.1.2)
- Eutrophication in freshwater systems (Section 4.1.3)
- Acidification of ecosystems and waters (Sections 4.1.3., 4.4.4 & 4.5.4)
- CO_2 emission from NO_3^- induced ocean acidification (Section 4.1.3)
- Human health impacts (Sections 4.1.4, 4.4.4 & 4.5.3)

4.1.1 Aquatic primary productivity

Background on the mechanisms by which excess N effects aquatic ecosystems

Many freshwaters and coastal zones worldwide are degraded from inputs of excess reactive nitrogen (N) from runoff, groundwater inputs and atmospheric N deposition, point sources (e.g., industry, sewage plant), as well as N that is either imported, manufactured or deposited in upstream catchments. Some major adverse effects include harmful algal blooms, hypoxia of fresh and coastal waters, ocean acidification, long-term harm to human health, and increased emissions of greenhouse gases (Baron et al. 2013). These effects occur when N, which throughout evolutionary history has been a limiting nutrient for growth, becomes readily available and stimulates aquatic plant growth (Conley et el. 2009). Similar effects can occur in lakes, where nitrogen contributes to an increase in primary productivity, stimulating macrophytes and algae. Freshwaters and estuaries can experience proliferation of cyanobacterial blooms. Cyanobacteria are highly undesirable because they can be toxic, cause hypoxia, and disrupt food webs (Conley et al. 2009). Also, the loss of water transparency due to water pollution is an important driver for aquatic biodiversity loss.

Effects of excess N on different aquatic habitats and taxonomic groups

Eutrophication is the process by which water becomes progressively enriched with nutrients. As N increases to the point where it is no longer limiting, it alters the linkage among pressure-state-impacts (Figure 1.4). Estuarine and coastal marine ecosystem eutrophication can cause loss of seagrasses and hypoxia, which disrupts food webs (Diaz & Rosenberg 2008). In that study, the World Resources Institute identified 415 major zones of coastal hypoxia and eutrophication worldwide in 2008, covering most major rivers and estuaries except those of the far northern hemisphere where human population and development are minimal. Nitrogen has also been shown to have adverse effects on estuaries and coral reefs, including:

- Increased biomass of phytoplankton and epiphytic algae
- Shifts in algal composition to taxa such as dinoflagellates that are toxic or inedible
- Increases in nuisance blooms of toxic zooplankton
- Death and losses of coral reef communities
- Increased probability of kills of fish, crustaceans and shellfish (Smith et al. 1999).

Eutrophication increases algal biomass in freshwater environments, including lakes, rivers and streams. In lakes, species richness tends to increase to an optimum based on the availability of nutrients, and then decline as some species exert dominance over others. The patterns for lake phytoplankton, rotifers, cladocerans, copepods and macrophytes were significantly unimodal in a survey of several well-studied lakes, with the highest biodiversity occurring in lakes with relatively low primary productivity (Dodson et al. 2000). Species richness generally peaked at levels of primary productivity in the range of 30–300 g C m^{-2} yr^{-1}. Bacterial community composition varies with phytoplankton productivity and with total N concentrations in lakes (Kolmonen et al., 2011). Macrophyte diversity declines with increasing N concentrations (James et al. 2005; Qin et al., 2013). Several studies demonstrate a loss of phytoplankton diversity with increased N (Proulx et al. 1996) and a shift from autotrophic to heterotrophic bacteria (Kolmonen et al. 2004). In rivers, an increase in N often leads to rapid abundant growth of filamentous green algae (Smith et al. 1999). In groundwater-fed wetlands, N can alter plant communities and increase the N status of the vegetation and soils (Rhymes et al. 2014)

Methods of measuring impacts of excess N on aquatic systems

Monitoring is used for identifying changes in water quality and in ecological responses over time. There are many standardised monitoring protocols available from countries and regions worldwide and from the World Health Organization (Bartram & Ballance 1996). Causality, however, must be determined with experiments conducted in the field, sometimes with mesocosms (Axler & Reuter 1999), or in the laboratory (Elser et al. 2009). Dependent variables can be primary productivity, as well as changes in species assemblages of algae or macrophytes, and higher members of food webs, including zooplankton, invertebrates and fish. Remote sensing is becoming more robust at reporting chlorophyll as a measure of lake primary productivity, especially for larger bodies of water (Bresciani et al. 2018).

Interactions of N with other factors

Primary productivity in aquatic ecosystems can be limited by phosphorus (P) or sometimes other nutrients, such as silica (Si) in addition to N. Phosphorus limitation status alters productivity, species assemblages and diversity. Empirical analyses of algal biomass (measured as chlorophyll *a*) relative to P and N levels in lakes and streams show these two nutrients act synergistically to promote eutrophication (Figure 4.1.1). Both N and P are essential to maintaining aquatic food webs and can have positive effects in water bodies such as supporting fish and shellfish production. However, an excess of N, P, or both, can lead to eutrophication, anoxia, harmful algal blooms and loss of biodiversity (Wurtsbaugh et al. 2019). The N to P ratio may be useful as a measure of which nutrient may limit algal growth (Redfield 1958). When the total N to total P ratio (TN:TP) in the environment is >14:1 (by mass), N may be present in excess and P often limits algal growth. When TN:TP < 14, N is often the limiting factor (Downing & McCauley 1992). However, the bioavailability of each nutrient and the nutrient needs of different algal taxa makes this relationship far from precise, and some authors have suggested using different ratios to assess nutrient limitation (Morris & Lewis 1988; Tank & Dodds 2003; Wurtsbaugh et al. 2019).

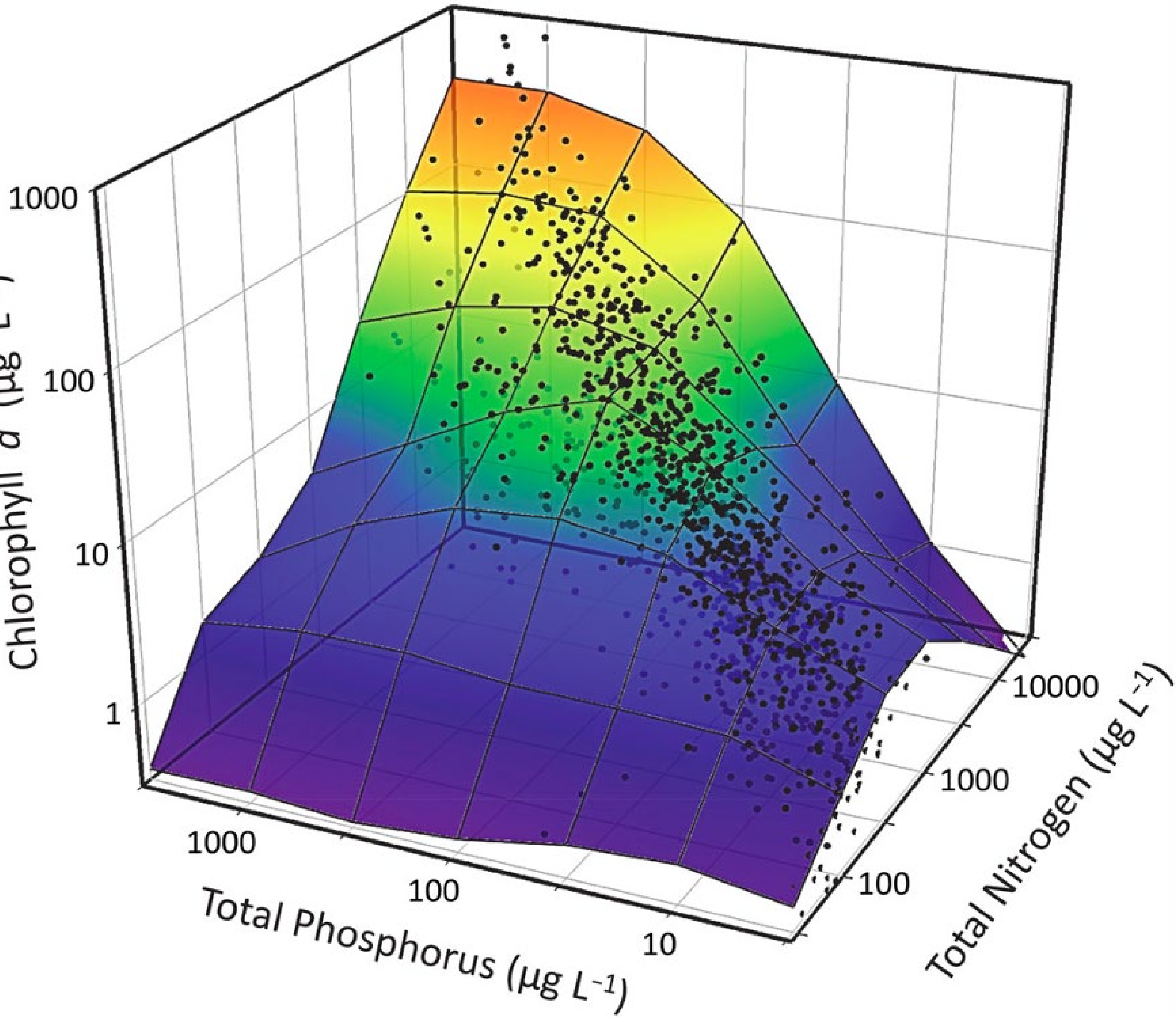


Figure 4.1.1. Relationship between phytoplankton chlorophyll a levels (z-axis) and total phosphorus and total nitrogen levels in 1264 lakes studied in the U.S. (EPA 2017). Only lakes where total phosphorus was ≥3 µg L^{-1} were used in the analysis. The surface curve was fit with local regression smoothing (LOESS modeling) and a sampling proportion of 0.4. A similar relationship has been shown between N, P and chlorophyll *a* in benthic periphyton of streams are frequently N-limited (Dodds & Smith 2016). In oceanic zones with limited upwelling of deep nutrients, both iron and N may be limiting (Downing et al. 1999; Moore et al. 2013). Copyright © 2006 Canadian Science Publishing; National Research Council of Canada.

There are many freshwaters where both P and N control algal growth (Moss et al. 2013). The most pristine oligotrophic aquatic environments are largely in balance relative to the N and P needed for low levels of algal growth. Pollution with N or P increases primary productivity and alters algal assemblages to favour species that thrive with the availability of one or the other nutrient. Runoff from fertilised cropland and atmospheric N deposition may have lessened N limitation over parts of North America, Europe and Asia, causing P to become the limiting nutrient for those environments (Elser et al. 2009; Stoddard 1994). Non-point source pollution from agriculture is high in NO_3^- relative to P (Wurtsbaugh et al. 2019). Conversely, sewage from urban areas is typically high in P, which may cause high P loads in waters, leading to more N-limited conditions (Moss et al. 2013). Floating plants cannot access sediment P, while aquatic plants with roots can uptake P from the sediment. Therefore, increased P from point sources due to urbanisation and industrialisation without enough wastewater treatment can increase blooms of the floating plants.

The effect of climate change on N transport and processing in fresh and coastal waters will be felt most strongly through changes to the hydrologic cycle. Alterations in precipitation amount and dynamics will alter runoff, thereby influencing both rates of N inputs to aquatic ecosystems and the water residence times that affect N removal within them (Baron et al. 2013). Increasing temperature will also increase metabolic rates of algae and heterotrophs, enhancing production and decomposition (Griffith & Gobler 2019; Paerl 2016). In wetlands, drying periods can lead to internal eutrophication where stored nutrients are mobilised through faster decomposition of organic matter.

4.1.2 Coastal and estuarine eutrophication

Reactive N pollution in the marine environment (i.e., eutrophication) creates multiple undesirable disturbances resulting from both increased phytoplankton growth and shifts in nutrient ratios (e.g., N:P, N:Si) leading to altered phytoplankton and macrophyte community composition. Intense phytoplankton blooms enhance organic matter production that ultimately decomposes, driving up rates of heterotrophic respiration and re-oxidation that lead to hypoxia (i.e., low oxygen environments; Zhang et al., 2010). Hypoxic environments diminish fish, zooplankton, crustaceans and shellfish populations. Elevated organic matter production and high phytoplankton biomass can also have a detrimental effect on benthic habitats and communities like corals by limiting light penetration to the benthos. Phytoplankton communities in high N environments shift toward domination by cyanobacteria that may produce harmful toxins (Glibert et al. 2014). Nuisance species hurt tourism and recreation by generating unsightly foams and red tides, while toxic species can impact human and animal health through direct contact (e.g., while swimming) or via fish or shellfish consumption.

Atmospheric N deposition and flushing of N from river basins can acidify coastal waters, potentially harming a range of benthic and pelagic organisms that form calcareous shells, including corals, coralline algae, foraminifera, pteropods and coccolithophores (Doney et al. 2007, 2020; Rheuban et al. 2019). Changes in these organisms, many of which form the basis for coastal food webs, can alter coastal marine communities and ecosystems, with economic consequences for human populations (Doney et al. 2020).

Processes of eutrophication

Whether a sea region experiences an undesirable disturbance from a particular nutrient input is highly dependent on the environmental context. Shelf and coastal seas receive a substantial fraction of their nutrients from the open ocean. Where denitrification and burial of reactive N exceed N fixation (as is often the case), these regions are a net sink of N (Voss et al. 2013). Hence, the impact of anthropogenic nutrient inputs is highly dependent on the degree to which a sea body is mixed and flushed with wider oceanic water. Generally, isolated regions (e.g., bays, estuaries) are more susceptible to eutrophication than strongly flushed coasts, although away from equatorial regions riverine inputs tend to remain close to the coast and coastal currents (Izett & Fennel 2017). Similarly, whether hypoxic conditions arise depends on the degree of ventilation: Strong density stratification is usually required to isolate bottom waters from re-oxygenation by air-sea exchange. Both flushing and stratification are dependent on highly variable physical processes operating on daily, seasonal and inter-annual time scales, modulated by the geography of the region in question and sensitive to climate change (Holt et al. 2016). Therefore, the assessment of the trophic state of sea regions is highly complex and multiple regionally specific approaches have been proposed (Ferreira et al. 2011).

Methods used to assess impacts

Countries around the world legislatively require assessments of the trophic state of coastal and estuarine waters (Borja et al. 2008). The following examples illustrate the types of data and analyses necessary to produce such assessments. Current eutrophication assessments fall into two broad categories: Those which apply biogeochemical, biological and biophysical indicators to attempt an integrated assessment of the ecosystem, and those that rely on threshold nutrient concentrations alone (Xiao et al. 2007). One prominent integrated assessment approach is the 'Assessment of Estuarine Trophic Status' model (ASSETS; http://www.eutro.org/), which combines expert opinions of six symptoms of eutrophication, such as decreased light and changes in algal dominance (Bricker et al. 1999). These opinions are weighted based on factors like the robustness of the underlying data and elaborated into a pressure-state-response model that generates indicator ratings for present estuary trophic condition, susceptibility to eutrophication and likelihood of future eutrophication (Bricker et al. 2003). The framework determines the pressure by considering nutrient input against flushing and dilution potentials; the state by considering the overall estuary condition using six eutrophication symptoms; and the response by using estimates of future nutrient loading. The framework has been used successfully in the United States, Europe and China (Whitall et al., 2007; Borja et al. 2008). In comparison with nutrient-based methods, integrated methods have advantages in identifying incremental changes in trophic state and can be applied more feasibly over wider areas (Xiao et al. 2007). Some researchers have proposed that integrated models should mix nutrient-concentration indicators with direct and indirect biological and biophysical indicators (Ferreira et al. 2011).

In Europe, the Oslo-Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) has worked to identify threats to the marine environment since 1972. The OSPAR commission and 16 contracting parties work together in the North-East Atlantic to assess pressures from human activities, with a major focus on eutrophication. In their common assessment procedure, sea regions are initially screened with a "broad brush" approach to remove obvious non-problem areas, followed by an iterative comprehensive assessment for the remaining areas (OSPAR 2013). Data sources include riverine, atmospheric and direct inputs, alongside *in situ* biogeochemical and biological observations (see www.ices.dk). The European Monitoring and Evaluation Programme (EMEP) provides atmospheric deposition data from numerical model analyses of N emissions. Individual countries develop national models to assess the direct, indirect, and transboundary transports and deposition of N (OSPAR Commission 2017). All the assessments are based on the European Union's Water Framework Directive (WFD), the Marine Strategy Framework Directive (MSFD), and the methods of assessment under the Convention on the Protection of the Marine Environment of the Baltic Sea Area (HELCOM convention) in the regions of the Baltic Sea (Liu et al., 2015). These directives and conventions could be tailored to fit N monitoring needs in nations outside of Europe.

Imbalances of N:P:Si in freshwaters flowing to the coast can trigger harmful algal blooms. A measure of the concentration of those three nutrients and a subsequent application of the Indicator of Coastal Eutrophication Potential (ICEP) can help assess coastal eutrophication risks (Billen & Garnier 2007). This index is based on nutrient imbalances. A recent study proposes an improved index (B_ICEP), which includes the effect of the physical features of the receiving bay (Garnier et al. 2021). Modeling platforms linking models from the farm to the coast in the land-river-sea continuum are useful tools to develop scenarios looking for the best strategies to reduce coastal eutrophication (Desmit et al. 2018).

4.1.3 Freshwater eutrophication

In freshwater lakes and reservoirs, both N and P may contribute to eutrophication and toxic algal blooms, indicating the need for an integrated approach to risk assessment. Both the United States and Europe focus management strategies on limiting freshwater eutrophication by controlling nutrient inputs to lakes, reservoirs, rivers and streams. The United States quantifies total maximum daily loads (TMDLs, EPA 2011), whereas Europe tends to use critical loads (Section 3.6). However, management strategies that consider concentrations of N or P separately are challenged by interactions of N and P, as well as climate change, implying that nutrient concentrations should be considered together, and models should incorporate future warming (Paerl 2014). Additional complexity comes from intra-annual variability in lake inputs and processing capacity: In temperate regions where air temperatures vary throughout the year, season-specific standards for nutrients, chlorophyll *a*, or other related analytes may provide the best assessment of trophic condition (Guo et al. 2018).

Methods to assess the impacts

The N cycle is complex in natural ecosystems, characterised by feedbacks and a wide range of retention times for N species within the system. Freshwaters integrate the effects of upstream processes, so a systems or watershed approach is often used to understand drivers of changes to freshwater quality. There is extensive N-saturation literature on the consequences of excess N to terrestrial ecosystems that provides conceptual models of how N moves through natural ecosystems to freshwaters (Aber et al. 1989; Stoddard 1994; Ohte et al. 2001; Lovett & Goodale 2011; Chiwa et al. 2018). Mechanistic models are used to both portray understanding of processes by which N influences freshwaters and also to identify where gaps in data or understanding exist (Canham et al. 2003). Monitoring of water chemistry, combined with other tools such as remote sensing, have been employed to identify changes in system properties that can be further studied to determine the role of impacts from N.

Remote sensing technology contributes to eutrophication assessment, as it allows for prediction of algal blooms using spectral indicators of chlorophyll *a* concentration in lakes and reservoirs, which has been done in the eastern United States using Landsat-8 images (Keith et al. 2018). An integrated monitoring system in China's Lake Taihu combined three elements: Remote sensing using MODIS satellite imagery; automated monitoring sensors on buoys for water temperature, turbidity, chlorophyll *a* concentration; and twice-weekly shipborne sampling of N, P, and chlorophyll *a*. Modeling approaches facilitate the integration of data for multiple water quality parameters across space and time. Using the Environmental Fluid Dynamic Code (EFDC) model, investigators at the Danjiangkou Reservoir in China determined the risk of eutrophication and algal blooms from available water quality data (e.g., dissolved oxygen, total nitrogen, chlorophyll *a*, water temperature), differentiating between high- and low-risk areas within the reservoir (Hamrick 1992; Chen et al. 2016). A further application of the EFDC model carried out in Lake Chaohu, China, produced an index of integrated spatio-temporal "hot spots" and "hot moments" of highest eutrophication risk (Huang et al. 2018).

As N is a limiting element for primary productivity in most terrestrial ecosystems (particularly in temperate regions), atmospheric N deposition may substantially increase productivity and C sequestration in those systems. However, atmospheric N inputs in excess of terrestrial ecosystem requirements (i.e., where critical loads are exceeded) can leach from soils or foliage in the form of N gases, causing elevated stream NO_3^- and acidification. Ecosystem monitoring over time is useful to assess how atmospheric N deposition may alter the N cycle (Aber et al. 2003; Baron et al. 2000; Groffman et al. 2018). Spatial comparisons between atmospheric N deposition (as a pressure indicator) and ecosystem N output, such as soil N leaching or stream water N concentration (as state or impact indicators), help identify N impacts on freshwater quality (Figures 1.4, 4.1.2 & 4.1.3). *In situ* manipulation of N addition to ecosystems has also been used to assess the N-deposition impact to the N cycle and to surface water quality. Long-term N-addition experiments at the watershed scale have yielded significant research findings on the complex responses of N dynamics and stream chemistry (e.g., Fernandez et al. 2010). In agricultural ecosystems, N fertilizer is the dominant N input to the cropland ecosystem (except for the N-deficient regions and for soybean that are dominated by biological N fixation). Plant harvest is a significant N output, so surplus N (fertilizer input minus harvest) can be a useful indicator to assess the fertilizer impact on the N export to groundwater and stream water (Figure 4.1.4).

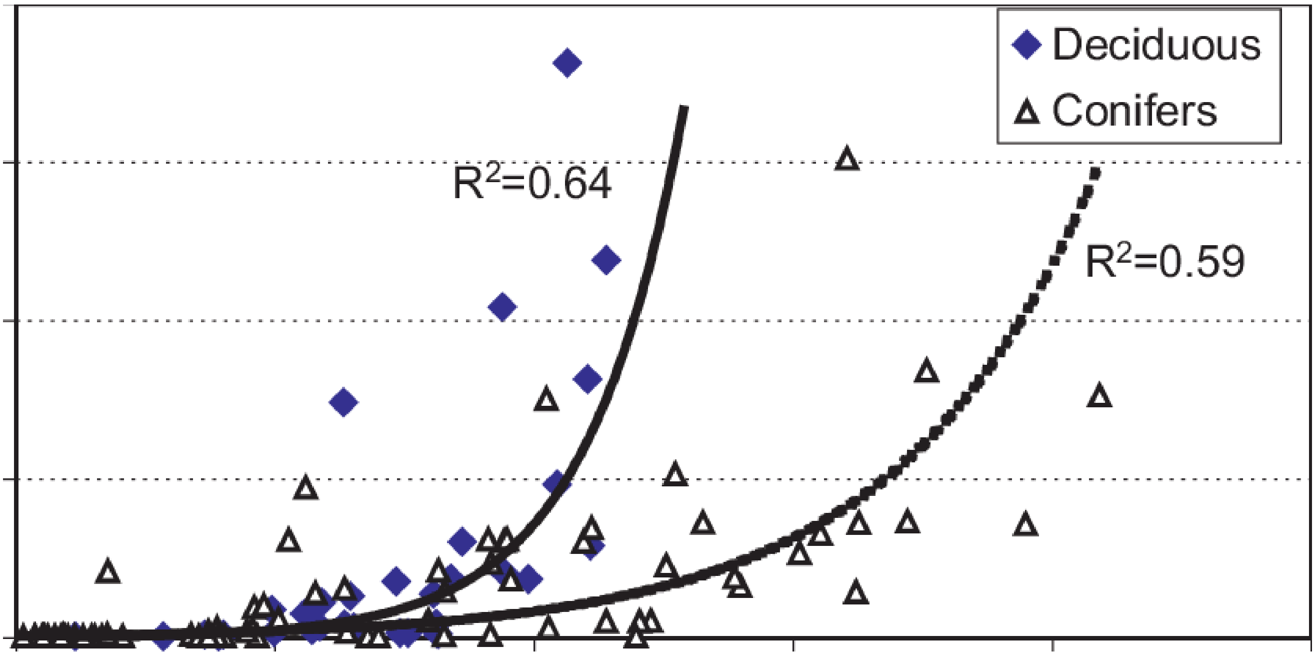


Figure 4.1.2. Relationship between reactive nitrogen (N) inputs by throughfall (as a pressure indicator) and the soil solution nitrate-N concentration below rooting depth (as a state indicator) in European forest ecosystems (Gundersen et al. 2006). The N leaching from the forest soil becomes a source of water quality impairment in groundwater and stream water, and deciduous and coniferous forests have different responses to atmospheric N deposition. Copyright © 2006 Canadian Science Publishing; National Research Council of Canada.

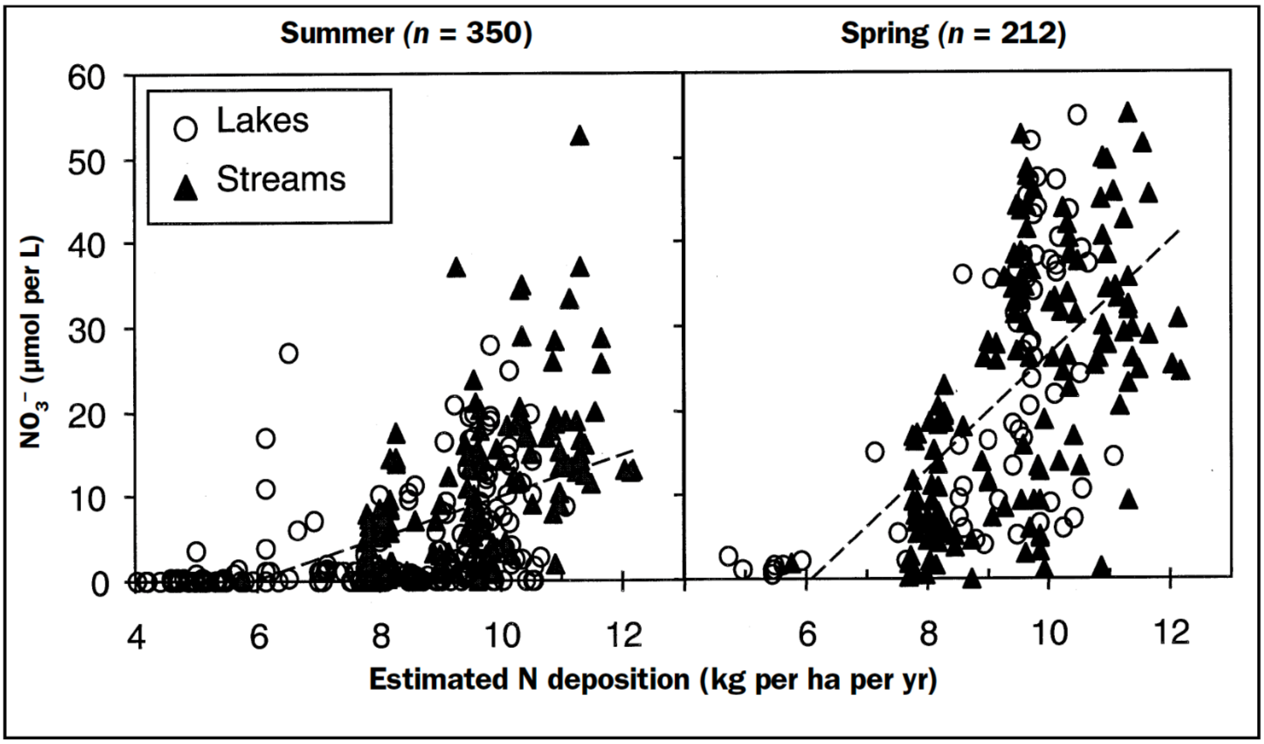


Figure 4.1.3. Response of the nitrate concentration in lakes and streams to atmospheric nitrogen (N) deposition in the northeastern United States (Aber et al, 2003). Response functions are different between growing (summer) and dormant (spring) seasons. Copyright © 2003, Oxford University Press.

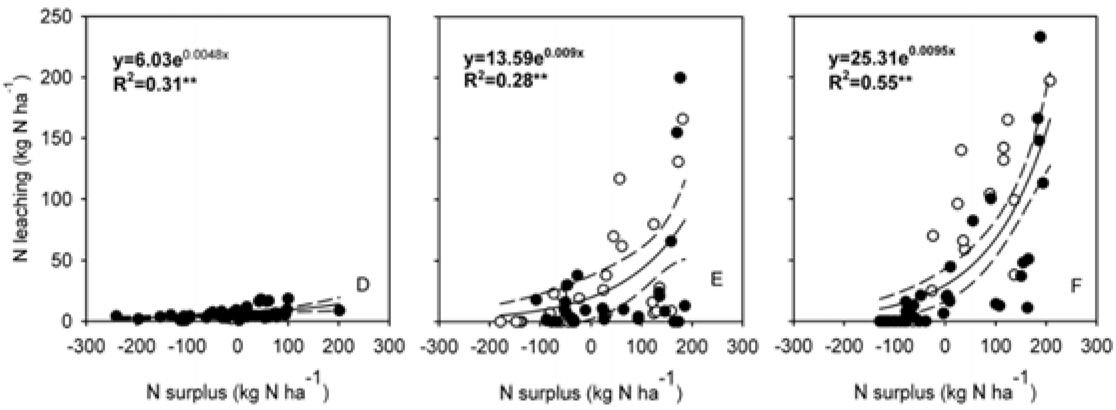


Figure 4.1.4. Exponential response curve of nitrogen (N) leaching to the N surplus in rice (left), wheat (centre) and maize (right) production in China (Chen et al. 2014). Nitrogen surplus was calculated as N application rate minus above-ground N uptake. Solid and hollow circles represent data from Chinese journals (or theses) and ISI journals, respectively. Different crop types have different responses to N surplus. Copyright © 2014 Springer Nature.

4.1.4 Nitrogen impacts on freshwater, coastal and marine fisheries

Nitrogen has multiple, contrasting impacts on fish population and fisheries. Reactive N can affect the quantity of prey (e.g., plankton) available for consumption by fish and influence many aspects of their abiotic habitat including the availability of oxygen, water transparency and the presence of toxins. A positive relationship has been shown between N loads and fisheries landings (harvests) across a wide range of estuaries and semi-enclosed seas worldwide (Breitburg et al. 2009b), though it should be noted that a reduction in landings near the very high end of N inputs (i.e., due to the presence of dead zones) could not be ruled out. This finding indicates that negative effects of eutrophication on fisheries, including through anoxia, are not (yet) apparent. However, ever-increasing areas of the coasts and open ocean are affected by anoxia (Breitburg et al. 2018), which is certain to compress fish habitats. Negative impacts of high N inputs on fish landings may be masked by an increase in fisheries' efforts in response to decreases in fish stocks. This could temporarily counter a decrease in landings but would ultimately have severe consequences for stocks if they are fished above a maximum sustainable yield.

The ultimate effect of eutrophication may be a completely different type of food web in which the role of fish, and their abundance, is severely reduced. For instance, eutrophic or hypereutrophic conditions may result in largely anoxic systems in which most of the elemental cycling is performed by microbes. In such a system, any positive 'fertilisation effects' of eutrophication can no longer propagate to fish, leaving fisheries landings to be unavoidably reduced. Such drastic shifts are typically associated with small-scale environments with very high anoxia or very specific sites, such as the Thames, Mersey, and Elbe estuaries when they were subject to raw sewage discharge (Breitburg et al. 2009b).

Processes by which nitrogen alters fish populations and ecosystems

The main impact of increased N loads from river inputs, sewage treatment outflows and direct fertilisation has been termed the 'fertilisation effect' or 'agricultural model': Increased concentrations of N lead to increased primary productivity. This sustains greater abundances of species predated upon by fish, resulting in greater levels of fish production (Nixon & Buckley 2002). This relationship has been found in both the marine environment, where regions with higher N inputs show greater catches (Breitburg et al. 2009b), and in freshwater systems where the reduction in nutrient loads over past decades ('re-oligotrophication') has decreased fish catches, with sometimes severe economic consequences (Eckmann et al. 2007; Stockner et al. 2000).

Aquaculture may be similarly affected if dependent on food supplied by the ambient environment (e.g., bivalve aquaculture that relies on natural plankton), but in many cases feed is separately added to the water. When feeds are added for aquaculture, aquaculture can itself become a significant source of N. At aquaculture sites, individuals are farmed at a much higher density compared to natural environments to maximise the economic benefit. This leads to concentrated waste products from the animals farmed, including NH_3.

Nitrogen loads not only affect food available to fish, but also their habitat. Through its stimulation of biological productivity, increased N input enhances the production of organic matter. Subsequent consumption or degradation of this organic matter draws down oxygen. When this happens in deeper waters with infrequent exposure to the atmosphere, the oxygen concentration can be reduced to values below the threshold suitable for animal life (Brennan et al. 2016; Vaquer-Sunyer & Duarte 2008). Such a drop in oxygen, leading to hypoxia or anoxia, can result in fish kills if the fish cannot escape the affected area.

Excess nutrients and the imbalance between N and P can stimulate the proliferation of algae and cyanobacteria that cause Harmful Algal Blooms (HABs; Anderson et al. 2002; Heisler et al. 2008). HABs can produce toxins that can cause massive fish and shellfish kills with subsequent economic losses (Brown et al. 2020). Toxins from HABs can also lead to illness in people who directly come in contact (e.g., through inhalation or skin contact) or by ingesting contaminated fish and shellfish. The range of illness from HABs is quite broad, ranging from relatively minor (e.g., nausea, headache, diarrhoea) to severe (e.g., liver and kidney damage, bradycardia, amnesia, hallucinations; Berdalet et al. 2016). For this reason, aquaculture production is screened for those toxins, and when the concentration exceeds thresholds that could cause illness in consumers, harvesting the product is prohibited until the level of toxins falls below the recommended values, resulting in economic loss.

Nitrogen compounds in water, particularly NH_3, can be directly toxic to fish and shellfish. High environmental levels of aqueous ammonia compromise an animal's ability to excrete the ammonia produced by metabolic processes. High internal NH_3 concentrations affect the central nervous system and can cause convulsions and death (Randall & Tsui, 2002). The critical concentration is species specific and depends on the physiological condition of the individuals, as well as environmental factors like pH, temperature and salinity, which influence the equilibrium between ammonium and ammonia. For example, European Union directive 2006/44/EC, which is aimed at protecting freshwater environments to support fish life, has different limits depending on the circumstance. The directive imposes a maximum concentration of 1 mg l^{-1} of total NH_3 with much stricter limits of 0.04 mg l^{-1} for waters with salmonid species and 0.2 mg l^{-1} for cyprinid (minnows and carps) waters (EP-EUCO 2006). Meanwhile, the U.S. Environmental Protection Agency (EPA) recommends a limit on the 30 days' average of 1.9 mg l^{-1} at pH 7 and temperature of 20 °C (EPA 2013).

Methods by which effects on fisheries are determined

Effects of N on fisheries are not easily quantifiable as they are always indirect (mediated by the ecosystem); potentially contrasting as outlined above; and influenced by additional human factors such as harvest pressure. Most studies use a correlative approach, looking at how fishery indicators (e.g., fish landings) vary with changes in N loadings or concentration. A positive correlation is often observed, particularly in areas that are sensitive to changes in N loadings like estuaries and semi-enclosed seas (Breitburg et al. 2009b) or in areas that experience drastic changes like the Egyptian coast off the Nile (Nixon 2003). The positive correlation does not hold when N loads are very high. Under these conditions, fisheries output either stabilises (potentially indicating that the ecosystem reached a carrying capacity dictated by factors other than N availability) or collapses due to the negative effects of N described above. Some studies did not report any correlation between fish biomass and N availability (Micheli 1999).

Indirect impacts of nitrogen through habitat modification

The dominant response of fish to reduction in oxygen concentration is avoidance, which can be triggered at oxygen concentrations far above those considered to be lethal, for instance, at 50% saturation (Breitburg 2002). This leads to changes in the spatial distribution of fish and, possibly, habitat compression (Bertrand et al. 2011; Prince & Goodyear 2006). The concentration of individuals in smaller volumes of water can enhance predator-prey interactions (Rose et al. 2009). Indeed, the same can affect fish-fisher interactions; there is evidence for enhanced catches at the edge of suboxic zones. A secondary effect of habitat compression is a change in the structure of the fish community, and, accordingly, in the type of species that are caught. In particular, with low oxygen concentrations typically being confined to deeper water, one might expect an increase in the ratio of pelagic to demersal fish as near-bottom habitats become unviable due to anoxia (Hondorp et al. 2011).

Nitrogen loading can further compress fish habitats by shifting the balance between phytoplankton and submerged vegetation. The increase in pelagic productivity that results from enhanced N availability typically increases turbidity, which negatively affects bottom-attached macrophytes and seagrasses (Rabalais 2002). These form key habitats and nursing grounds for a wide range of fish species. However, the impact of reduced water transparency on fish does not need to be exclusively negative. It has been suggested that more opaque waters may act as refugia for fish, as their predators mainly hunt by sight (Breitburg et al. 2009b).

4.2 Terrestrial ecosystems

Ecosystems and biodiversity are affected by a globally perturbed N cycle through a variety of means. These can have a cascade of effects that generally follow four potential pathways in terrestrial ecosystems:

i. Increases in the availability of N that then alters biogeochemical and population dynamics (i.e., eutrophication)

ii. Soil acidification that can lead to leaf and soil cation imbalances (i.e., acidification)

iii. Direct toxicity from exposure to concentrations in the air

iv. Modification of disturbance by other stress factors (e.g., colder temperatures, fire)

The two dominant mechanisms are eutrophication and acidification. Eutrophication (often termed 'enrichment' in terrestrial contexts) is the process by which N is available at unnaturally high levels due to human activity. There can also be impacts from direct toxicity of ammonia or NO_x gases, particularly on plants, such as lichens and mosses (Krupa 2003). In the case of ammonia, its alkalinity can also contribute to adverse effects (Sutton et al. 2020), where the pH of leaf surfaces increased. Subsequently, NH_3 may be nitrified in the soil to nitrate (NO_3^-), which is an acidifying process. In this way, ammonia can contribute to both surface alkalinisation and subsequent soil acidification. The 'alkaline air' effect of ammonia has become more significant in Europe over recent decades as SO_2 and NO_x emissions have been abated, while NH_3 emissions have largely continued unchecked, therefore increasing the gaseous alkaline fraction. Such situations of NH_3 excess may be more widespread in other parts of the world, and also be influenced by dusts released from naturally alkaline soils.

Ecosystems receive excess N as atmospheric deposition of N emissions that are byproducts of transportation, industrial and agricultural activity, and as runoff from agricultural areas. Here, we omit natural sources of excess N (e.g., natural forest fires, hotspots of rock N weathering), as those are generally outside of human control. Based on many years of study from investigators worldwide, we have high confidence in the effects of N on biodiversity and ecosystem function (Bobbink et al. 2010; Simkin et al. 2016).

Processes by which nitrogen affects ecosystems and biodiversity

Nitrogen is a commonly limiting resource (or co-limiting with P; Figure 4.2.1) to autotrophic growth (i.e., by photosynthesis) in both terrestrial and aquatic ecosystems (Vitousek & Howarth 1991). Increases in its availability can increase autotrophic production, which can then lead to a cascade of secondary effects — some good and some bad (Figure 1.2). Acidification is the process by which atmospheric deposition of N and S leads to soil acidification, which can then have a myriad of impacts on terrestrial and downstream aquatic ecosystems, including:

- Leaching of base cations from the soil (e.g., Ca^{2+}, Mg^{2+})
- Reduced concentrations of base cations in plant tissue
- Reduced plant health and vitality,
- Increased aluminum mobility in the soil, which is phytotoxic
- Reduced acid-neutralizing capacity in surface waters
- Impacts to aquatic biota (Chapter 4.1)

Considering varying chemistry of nitrogen compounds, from nitric acid (HNO3) to NH_3 as an alkali, it must be emphasized that it is possible to have both surface alkalinisation (if NH_3 is the dominant dry deposited N gas) and subsequent soil acidification, especially where nitrification of NH_x to NO_3^- occurs.

The following N impacts are presented in this section. See the associated N-MIP matrix (supplementary file "N-MIP matrix.xlsm") for more information about these N impacts.

- Eutrophication effects on biodiversity in terrestrial ecosystems (Section 4.2.1)

- Eutrophication effects on below-ground biota and biodiversity in terrestrial ecosystems (Section 4.2.2)
- Acidification, nutrient imbalances, N_2O emission and NO_3^- leaching (Section 4.2.3)
- N saturation (Section 4.2.4)
- Ozone and aerosol damage to trees and natural vegetation (Sections 4.2.5, 4.2.6)
- Enhanced plant and soil carbon sequestration in natural and managed forest ecosystems from atmospheric N deposition (Section 4.3.4.)
- Biodiversity protection from land conversion to cropping system by advances in crop productivity and increased cropping intensity (Section 4.4.3)

Methods to assess nitrogen impacts of terrestrial ecosystems and biodiversity

There are three major types of evidence available to relate atmospheric N deposition to biodiversity for terrestrial ecosystems (See also Section 3.6.2):

- Manipulation experiments in which N deposition is increased, usually by application of NH_4^+ and/or NO_3^- in artificial rainwater, or occasionally by enhancement of gaseous NH_3
- Spatial field surveys along a gradient of N deposition or N concentrations, generally with a regional, national, or continental focus
- Re-surveys over time of previous vegetation studies

Each of the above approaches has strengths and weaknesses, as summarised in Table 4.2.1, and these studies can be complementary. Multiple strands of evidence from a variety of approaches thus provide the most convincing support for N-driven changes in biodiversity (Dise et al. 2011).

Table 4.2.1. Advantages and disadvantages of nitrogen (N) manipulation experiments, surveys along N-deposition gradients and re-surveys over time for terrestrial ecosystems and biodiversity.

Approach	Advantages	Disadvantages
Manipulation experiments	Evaluate the discrete impact of N Can provide cause-effect (pressure-impact) relationships Can identify thresholds Opportunity to compare the effects of different N forms explicitly	Assess relatively short-term responses (few exceed 20 years) Potential for artefacts (e.g. high N concentrations) Systems may already be impacted by N (hence experiments need to be located in clean locations)
Spatial field surveys	Can provide insights into longer-term responses Can cover a wide range of N-deposition Avoid experimental artefacts	Cannot prove causality Other drivers on diversity need to be accounted for
Re-surveys over time	Type of evidence that can directly identify changes occurring over long periods of time, without experimental manipulation	Confounding influence of other factors (e.g., land use, climate), locating sites, methodology changes, incomplete records, data accessibility, etc.

Manipulation experiments

These evaluate the impact of N by comparing results of an N addition plot with a control plot where no N is added, while all other conditions are (assumed) identical. The disadvantage is that manipulation experiments typically assess relatively short-term responses (seldom exceeding 20 years) and often use unrealistically high concentrations of the applied pollutant, which may influence the response of the vegetation. Experiments conducted in areas with a relatively long history of elevated N deposition that may already have impacted biodiversity, or a legacy of past land use may confound the ability to identify thresholds of response from experiments. Such disadvantages can be addressed by using a range of N doses, by establishing funding to allow long-term experiments and by conducting experiments at clean locations. Despite the potential disadvantages, the fact that manipulation experiments evaluate the impact of N on plant species diversity under otherwise similar environmental circumstances means empirical critical loads of N, as used in European environmental policy, are mainly based on manipulation experiments. The other advantage is that this approach allows the explicit comparison of different forms of nitrogen deposition (wet *versus* dry deposition, oxidized *versus* reduced nitrogen deposition (e.g. Levy et al. 2019; Sutton et al. 2020).

Results of manipulation experiments have been used to derive pressure-impact (i.e., N load-plant species diversity) relationships (Figure 1.4), such as empirical relationships between plant species richness ratio and total N inputs for European grasslands habitats (Hettelingh et al. 2015). The species richness ratio was calculated per N treatment as S_n/S_c, where S_n is the number of species in the N-treated situation and S_c their number in the control situation. This ratio thus expresses whether species richness in the addition experiment is equal to ($S_n/S_c = 1$), higher than ($S_n/S_c > 1$), or lower than ($S_n/S_c < 1$) the control. A significant negative relationship (negative exponential fit; $p<0.001$) was found between the species richness ratio and the total N load (Figure 4.2.2). However, the scatter is relatively large, probably caused by data from studies in very different grassland types and differences in experimental set-up, especially duration (Hettelingh et al. 2015).

Spatial field surveys

These can provide insight into longer-term responses, cover a realistic range of N deposition and avoid potential experimental artefacts. Since gradients of N deposition may be correlated with those of other potential drivers (e.g., sulphur deposition, climate or management intensity), these other drivers need to be measured and considered in statistical analyses and interpretation. Because they are correlative, surveys cannot prove causality, but can often determine the statistical significance of N deposition as a potential driver of changes in biodiversity. Ecological surveillance networks can be analyzed for spatial relationships between diversity and N deposition. Field surveys typically record the presence or absence of species in larger areas (e.g., 10×10 km squares). As they are usually not designed to specifically identify N deposition (or even pollution) impacts, such studies reflect the influence of land use and a range of climatic, edaphic and management factors. Attribution of change to N deposition, therefore, can be difficult. However, field surveys can have wide spatial coverage, and so can potentially detect signals of change in biodiversity at the national level, including effects on rare species.

Using regression analysis Stevens *et al.* (2004) found that the floristic diversity of acid grasslands in the UK declined as a linear function of the rate of inorganic N deposition, with a reduction of one species for every 2.5 kg N ha^{-1} yr^{-1} of N deposition. At the mean N deposition of central Europe (17 kg N $ha^{-1}yr^{-1}$), this implies a 23% species reduction compared with grasslands receiving the lowest levels of N deposition (5 kg N $ha^{-1}yr^{-1}$). The same pattern of species richness decline with increasing N deposition was found across acid grassland habitats across Europe (Figure 4.2.3; Stevens et al. 2010) and similar relationships have been found in other habitats such as heathlands, bogs, and sand dune grasslands (Maskell et al. 2010; Field et al. 2014).

Re-surveys over time

These are often limited by the confounding influence of other factors that also influence vegetation change, such as land use and, increasingly, climate. It also may be challenging to identify the exact sites studied many years ago. However, given the limited duration of most experiments, re-surveys are the only type of evidence that can directly identify changes occurring over long periods of time, and so are an essential component of the strategy to characterise N deposition impacts on vegetation community composition and diversity. Long-term ecological surveillance studies across Europe have shown a significant relationship between the elevated level of N deposition and the decline in species characteristic of low-nutrient conditions, as well as an increase in nitrophilic plant species over recent decades (Figure 4.2.4; Duprè et al. 2010).

4.2.1 Above-ground biodiversity

Elevated N inputs to terrestrial natural and semi-natural ecosystems are generally caused by atmospheric N deposition. As described above, these can have a cascade of effects that generally follow four potential pathways in terrestrial ecosystems: Eutrophication, acidification, direct toxicity (including alkalinity) and secondary impacts induced by other stress factors (Figure 4.2.1). Most terrestrial systems are primarily affected by eutrophication and/or acidification, but all four pathways are possible in all terrestrial ecosystems, and each may play a larger or smaller role based on local factors. For example, in a community that is strongly N-limited, release from N-limitation via increased deposition may alter community composition by giving some plants a competitive advantage over others. In a community where the soil is already acidic, increased acidification from N (and S) deposition could have a larger effect than in a community with a well-buffered soil (Simkin et al. 2016; Schmitz et al. 2019).

Other components of an ecosystem can be directly impacted by excess N – independently from or only partly dependent on plant community response. These include the lichen, invertebrate communities, and soil microbial communities, each of which are sensitive to excess N. Changes in these communities can have cascading effects on the ecosystem (Clark et al. 2017; Irvine et al. 2017), influencing nesting sites for avian and non-avian animals, as well as soil biogeochemical processes. Once the plant and/or soil community is affected, most other taxonomic groups in the food web can be affected as well, including insects, animals and other soil micro- and macro-organisms (Boot et al. 2016; Shaw et al. 2019). Much less, however, is known about these potentially sensitive groups compared with the plant community. Lichens and bryophytes appear to be especially sensitive to gaseous NH_3 (Wolseley et al. 2006; Pinho et al. 2012), as reflected in a low critical level of 1 $\mu g\ m^{-3}$ compared with 3 $\mu g\ m^{-3}$ for higher plants (CLRTAP 2017), which appears to reflect the sensitivity of naturally 'acidiphyte' species to the alkaline effects of ammonia (Sutton et al. 2020).

Changes in the biodiversity of a community can have a variety of effects on the ecosystem as a whole. These include effects on *ecosystem structure* such as the biomass and relative abundance of plants, herbivores, carnivores, soil microbes and other trophic or functional groups; on *ecosystem function*, or processes such as carbon sequestration and net primary production (NPP), decomposition rates, nutrient capture and leaching; and on ecosystem services such as water purification, erosion control and human health. In addition, biodiversity can also affect the stability of various ecosystem processes, structures and services through time. More diverse assemblages are often more stable through time, while less diverse assemblages may have the same average condition as more diverse assemblages, but may be susceptible to wider swings in variation as monocultures may be severely affected by one particular perturbation (e.g., population explosion of pine beetles on pine trees due to increased foliar N contents of pine needles).

Methods by which effects of N on vegetation and ecosystems are evaluated

As described above in Table 4.2.1, there are four general approaches used to explore the observed or predicted relationship between N inputs and biodiversity. First, **manipulation experiments** are where researchers add specific amounts of additional N and record the ecological response. There have been hundreds of manipulation experiments globally, especially in Europe, the United States, and Asia. These have the benefit of isolating the source of the effect to elevated N but have limitations in that they are difficult and costly to implement over large scales and for many locations. Also, some manipulations may cause possible artefacts due to experimental disturbances by the manipulation. Furthermore, there are uncertainties whether the form, timing and manner of manipulation accurately simulates atmospheric N deposition (Zhang et al. 2018). Such concerns can be addressed by careful experimental design and site selection which takes account of such effects (e.g. Sheppard et al. 2014). Second, are **spatial field studies across a gradient** of N deposition (termed "gradient studies"). Gradient studies are much more limited in number compared with manipulation studies (Simkin et al., 2016). They have the advantage of assessing the effect of actual N deposition, but they have the disadvantage that other factors may covary with the deposition gradient, muddling the response of interest. For example, ozone and sulphur deposition may spatially correlate with N, thus researchers need to transparently test for and communicate this covariance before interpreting results. Third, **re-surveys of plots** that have been sampled at some time in the past (presumably with lower deposition) are resampled currently (presumably with higher deposition) and can be used to infer changes due to deposition. These have the same advantages and disadvantages as gradient studies but with the covariation problem across time rather than space. Lastly, **modeling studies**, where ecosystem responses are simulated across a range of conditions that are difficult or impossible to replicate in the field provide a powerful tool for evaluating N effects. The advantages of modeling studies are their flexibility in exploring current understanding and in pointing researchers towards new avenues that may be underappreciated. The limitations of modeling studies are that the models themselves are built from current understanding of how ecosystems work, and so while they may shed light on sensitivities that are difficult to empirically test in the field, they are not, strictly speaking, new evidence of an effect.

Each of these methods may be used to estimate critical loads (CLs; Chapter 3.3), i.e., deposition thresholds below which there are no measurable effects, as well as critical levels, i.e. the concentration thresholds above which there are measurable effects. These are useful scientific approaches for assessing the risk from atmospheric deposition and concentrations of N and S.

Values of empirical critical loads for nitrogen are summarized according to ecosystem type in Table 4.2.2. Values of critical levels for NO_x and NH_3 are summarized according to averaging period in Table 4.2.3.

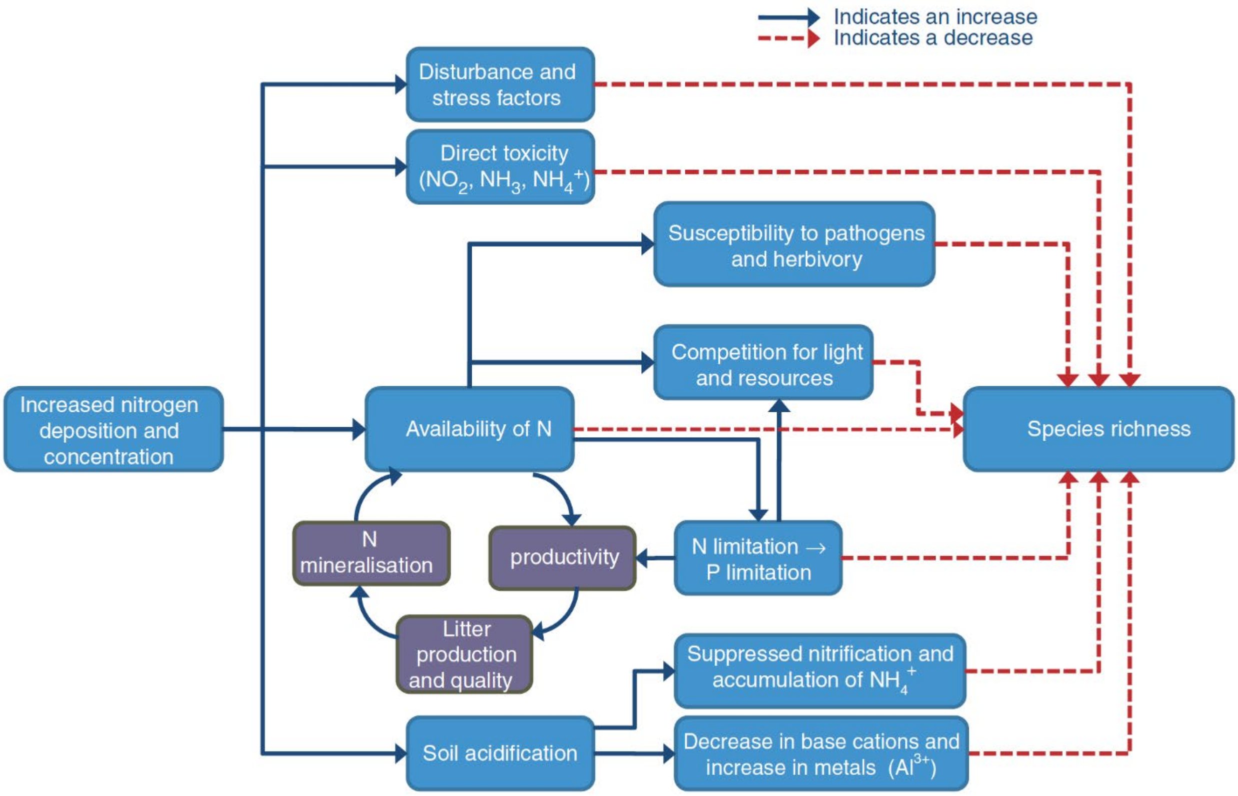


Figure 4.2.1. Conceptual diagram of the main impacts of enhanced nitrogen (N) deposition on terrestrial ecosystem processes and species richness (Sutton et al. 2011; Dise et al. 2011). Dark blue arrows show that elevated N increases indicators and ecosystem processes (purple boxes). Red arrows show the negative impact on species richness (right box). © Cambridge University Press 2011. Direct toxicity of NH_3 includes its alkaline effect on vegetation (Sutton et al. 2020), while soil acidification from NH_x is partly mediated by nitrification to NO_3^-, which tends to be suppressed under acid conditions.

Table 4.2.2. Empirical critical loads (CL) for nitrogen (N) deposition (kg N ha^{-1} yr^{-1}) to natural and semi natural ecosystems (modified from Bobbink et al. 2010). See Chapter 3 for details on critical loads. Reliability of the critical load as an indicator is shown as least reliable and requires expert judgement (#), somewhat reliable (##), and most reliable (###).

Ecosystem type	CL for N deposition	Reliability	Indication of exceedance
Forest habitats			
Temperate forests	10-15	##	Changed species composition, increased nitrophilous species, increased susceptibility to parasites, changes in mycorrhiza
Boreal forests	5-10	##	Changes in ground vegetation, mycorrhiza, increased risk of nutrient imbalances and susceptibility to parasites
Heathland, scrub, and tundra habitats			
Tundra	5-10	##	Changes in biomass, physiological effects, changes in species composition in moss layer, lichen decline
Arctic, alpine, and subalpine scrub habitats	5-15	#	Lichen, moss, and evergreen shrub decline
Northern wet heath	10-25	#	Decreased heather dominance, lichen and moss decline, transition heather to grass
Dry heath	10-20	###	Transition heather to grass, lichen decline
Grasslands and tall forb habitats			
Sub-Atlantic semidry calcareous grassland	15-25	###	Increased tall grasses, decreased diversity, increased mineralisation, N leaching
Non-Mediterranean dry acid and neutral closed grassland	10-20	##	Increase in graminoids, decline in typical species
Inland dune grasslands	10-20	#	Decreased lichens, biomass increase, increased succession
Low and medium altitude hay meadows	20-30	#	Increased tall grasses, decreased diversity
Mountain hay meadows	10-20	#	Increased tall graminoids, decreased diversity, decreased bryophytes
Molinea caerulea meadows, heath (*Juncus*) meadows, and humid (*Nardus stricta*) swards	10-15	##	Increased nitrophilous graminoids, changes to diversity
Alpine and subalpine grasslands	5-10	#	Increased nitrophilic graminoids, changes to diversity

Ecosystem type	CL for N deposition	Reliability	Indication of exceedance
Moss and lichen dominated mountain summits	5-10	##	Effects on mosses and lichens
Mire, bog, and fen habitats			
Raised and blanket bogs	5-10	###	Changes to species composition, N saturation of Sphagnum
Poor fens	10-20	##	Increased sedges and vascular plants, peat moss decline
Rich fens	15-35	#	Increased tall graminoids, decreased diversity and mosses
Mountain rich fens	15-25	#	Increased vascular plants, decreased mosses
Coastal & marine habitats			
Shifting coastal dunes	10-20	#	Increased biomass, increased N leaching
Stable dune grasslands	10-20	##	Increased tall grasses, decreased prostrate plants, increased N leaching
Dune heaths	10-20	#	Increased plant productivity, increased N leaching, accelerated succession
Moist to wet dune slacks	10-25	#	Increased tall graminoids
Pioneer and low to mid salt marshes	30-40	#	Increased plant productivity, increased late-successional species

Table 4.2.3. Critical levels for NO_x and NH_3 concentrations as used by the UNECE Convention on Long-range Transboundary Air Convention (CLRTAP 2017). These values mainly draw on experimental and observational studies under temperate European conditions, and further study is needed to refine critical levels for other climates (e.g., Ellis et al., 2022). Concerning the possibility of additive effects between NH_3 and NO_x, which could lower these values when the gases are present in combination, see Sutton et al. (2022). For wider recent review, see Franzaring et al. (2022).

Vegetation type	Critical level	Time period	Comment
Gaseous NO_x			**(NO and NO_2 added, expressed as NO_2, µg m^{-3})**
All vegetation (according to average sensitivity)	30 µg m^{-3}	Annual mean	The primary reference (WHO 2000) indicates that the most sensitive vegetation may have a value of around half this.
All vegetation (according to average sensitivity)	75 µg m^{-3}	24-hour mean	Considered less reliable than the annual mean value
Gaseous NH_3			**(Expressed as NH_3, µg m^{-3})**
Lichens and bryophytes (including ecosystems where lichens and bryophytes are a key part of ecosystem integrity)	1 µg m^{-3}	Annual mean	This refers especially to naturally 'acidophytic' species. Some lichen species are 'nitrophytic' and will increase in abundance above the critical level.
Higher plants (including heathland, grassland and forest ground flora)	3 [2-4] µg m^{-3}	Annual mean	The uncertainty range for higher plants is intended to be used when applying the critical level in different assessment contexts (e.g. precautionary approach or balance of evidence.)
Higher plants (provisional value only)	23 µg m^{-3}	Monthly mean	This value is considered more uncertain. A monthly value for lower plants was not determined, but would be expected to be lower.

4.2.2 Below-ground biodiversity

Soil biodiversity is extremely high (Giller et al. 1997). Soil organisms carry out essential ecosystem functions, including decomposition of organic matter, nutrient recycling, and maintenance of plant biodiversity (Tedersoo et al. 2014). The soil biota are composed of bacteria, fungi, protozoa and invertebrates living in complex relationships with each other in food webs. Soil biota are responsive to changes in above-ground ecosystems, to inputs of materials to soils from deposition, and to inputs of both above- and below-ground plant materials (Eisenhauer et al. 2012). Soil organisms and their interactions influence ecosystem processes, including decomposition, net primary production and trace gas production. These ecosystem processes in turn influence critical global biogeochemical processes (Guerra et al. 2021), including agricultural productivity and food security, water quality and climate (Wall & Moore 1999). Elevated soil N availability due to increased atmospheric N deposition can depress both bacterial and fungal biomass (Zhang et al. 2018), as well as rates of decomposition and soil respiration (Janssens et al. 2010). Nitrogen fertilisation of all soils, whether in grasslands, forests or agriculture, is expected to reduce soil biodiversity through a complex combination of chemical changes (e.g., lowered pH), changes to above-ground vegetation, which alter soil biogeochemical properties, and changes to soil food webs.

Processes by which N alters soil biota and biodiversity

The response of soil biota to additions of N is tightly coupled to their above-ground vegetation communities. In grassland soils, the microbial community response is related to the magnitude of plant-related responses (Leff et al. 2015). Forest soils have responded similarly, where N additions increase above-ground plant productivity, which in turn increases inputs of below-ground root materials and root exudates and above-ground plant products such as leaves. Nitrogen addition alters soil chemistry, including by decreasing soil pH and influencing soil organic matter chemistry (Reuss & Johnson 1986; Frey et al. 2014).

Methods for determining effect on soil biota and biodiversity

The effects of N on soil diversity are studied with field and laboratory experiments. Changes to inputs, their ratios of N to C, and responses, including microbial biomass, enzyme activity, shifts in bacterial to fungal ratios and gene sequencing are quantified (Bardgett et al. 1999; Boot et al. 2016; Leff et al. 2015; Ramirez et al. 2012; Eisenhauer et al. 2012). Long-term experiments with chronic N additions are used to compare control with treated plots (Frey et al. 2014; Eisenhauer et al. 2012; Boot et al. 2016). Nitrogen critical loads for below-ground taxonomic richness and diversity are typically empirically based (e.g., Pardo et al. 2011b). Like other methods, statistical methods include regression approaches for univariate measures such as species richness or diversity (Pardo et al. 2011b), multivariate analyses of community structure (Lilleskov et al. 2001) and threshold indicator taxon analysis (van der Linde et al. 2018; Figure 4.2.2.).

Experimental approaches for assessing N impacts on soil communities include longitudinal studies, atmospheric N deposition gradient studies, and N addition and removal experiments (Zhang et al. 2018; Lilleskov et al. 2019). Several factors should be considered in these approaches. First, background rates of atmospheric N deposition and cumulative amounts of N addition should be measured or estimated (In the USA, background estimates can be found through the Oak Ridge National Laboratory Distributed Active Archive Center's data set (Dentener 2006) or interactive map – (webmap.ornl.gov/ogcdown/dataset.jsp?ds_id=830 Dentener 2006). The soil N status of controls or reference sites in N addition or removal experiments should be also measured, because elevated background atmospheric N deposition may have already altered the below-ground communities in such sites prior to any nitrogen addition. Insufficient information about background levels of soil N availability can disrupt attempts to set critical N loads; for example, if background atmospheric N deposition already exceeds these threshold levels. Also, there can be lags in community microbial responses to atmospheric N deposition, so short-term, high-dose fertilisation studies are poor substitutes for long-term, low-dose fertilisation studies or gradients. In addition to quantifying background N inputs and pools, other soil nutrients (e.g., phosphorus), plant nutrients and pH status should be assessed, as these are the proximal drivers of below-ground community response (Lilleskov & Parrent 2007).

Many studies address the effects of N addition on soil microbial activities that control decomposition and trace-gas fluxes to the atmosphere (Frey et al. 2014; Chapin et al. 2002), and many researchers have explored the responses of soil bacteria and fungi to additions of reactive N (Leff et al. 2015; Edgerton-Warburton et al. 2000; Ramirez et al. 2010). A meta-analysis of N addition studies found that total microbial community and fungal abundance decreased as N load and duration of the N addition treatment increased (Treseder 2008).

Work on atmospheric N deposition impacts on fungi has focused primarily on mycorrhizal and saprotrophic fungi in soil. Mycorrhizal fungi are symbiotic on plant roots, exchanging soil resources, such as nutrients and water for host sugars. Saprotrophic fungi consume dead organic matter and convert reduced carbon into fungal biomass, carbon dioxide and other compounds. Sampling of soil fungi focuses on reproductive structures called sporocarps (mushrooms and related fruiting bodies), fine roots (for mycorrhizal fungi) and soils. Sporocarps provide longitudinal information unavailable from other sources (Arnolds 1991), but are biased because some species do not fruit regularly or are cryptic, and so are under-sampled. Many saprotrophs and one class of mycorrhizal fungi, called ectomycorrhizal fungi, produce above-ground sporocarps that can be sampled readily, whereas another class of mycorrhizal fungi, called arbuscular mycorrhizal fungi, are primitive fungi with only asexual below-ground microspores, and so are sampled by spore separation from soils. Sampling methods for sporocarps ideally include collections throughout the growing season and over multiple growing seasons to account for year-to-year climatically driven variation. See Mueller et al. (2004) for detailed methods.

Links between fungal taxonomy and function can be examined by several methods. Simplest is functional assignment to taxonomy when functions are known. For example, Funguild (Nguyen et al. 2015) distinguishes between the major guilds (e.g., endophytes, pathogens, mycorrhizal fungi, saprotrophs) at the genus level. Studies on isolates of known nitrophilic and nitrophobic taxa can identify suites of anatomical, physiological, and biochemical traits that are associated with taxa that respond differently to atmospheric nitrogen deposition (e.g., Lilleskov et al. 2011, 2019). At the aggregate level, changes in fungal functions — such as hyphal production, nutrient supply to hosts and substrate turnover — can be assessed with various methods, including:

- Hyphal ingrowth mesh bags with and without specific nutrients added (Lilleskov 2011)
- Substrate decomposition assays, nutrient content in fungi, natural abundances of N and C isotopes in component pools such as soils and fungi (Lilleskov et al. 2002)
- Nitrogen and C isotope tracer studies (Hobbie et al. 2014)
- Carbon tracers linked to phospholipid fatty acids (PLFAs; Hogberg et al. 2010)
- Stable isotope probing, enzyme assays and transcriptomics (Zak et al. 2006, Lilleskov et al. 2019).

Fungi are identified in fine roots and soils by fungal-specific PCR-based amplification of DNA barcode markers, usually the internal transcribed spacer (ITS) of ribosomal DNA. This is followed by either Sanger sequencing (with or without cloning depending on tissues), and, increasingly, high-throughput sequencing on various platforms (Nilsson et al. 2018). Although Sanger sequencing is low throughput and declining in use, with appropriate sampling methods it can provide quantitative information on taxon abundance for some groups and target tissues, such as root tips of ectomycorrhizal fungi (e.g., van der Linde et al. 2018). High-throughput sequencing provides large amounts of information but is semi-quantitative because of inherent amplification biases that vary among sequencing platforms (Nilsson et al. 2018). Quantitative PCR (qPCR) of specific taxa can be used to assess abundance with less bias (Fierer et al. 2005).

To measure bacterial abundance, researchers can directly count bacterial cells or quantify bacterial abundance following cell culturing in the lab. Because most bacteria cannot be cultured in the laboratory, researchers often employ various other techniques to characterise soil bacteria. Methods (described in detail in Agrawal et al. 2015) include analysis of fatty acids (e.g., FAME: Fatty acid methyl ester analysis) and molecular techniques, such as:

- Sequencing of 16S rRNA genes that include denaturing gradient gel electrophoresis (DGGE)
- Temperature gradient gel electrophoresis (TGGE), single-strand conformation polymorphisms (sscps)
- Amplified ribosomal DNA restriction analysis (ARDRA)
- Terminal restriction fragment length polymorphisms (T-RFLPS)
- Ribosomal intergenic spacer analysis (RISA)

More recently there has been an increase in high throughput sequencing for 16S rDNA targeted metagenomics for community analysis whereas shotgun metagenomics, metatranscriptomics, and metabolomics are increasing for microbial community and functional analysis (Aguiar-Pulido et al. 2016). Similar to fungi, qPCR can be used to assess bacterial abundance broadly or for specific taxa (Fierer et al. 2005).

To quantify fungal and bacterial biomass, researchers can use the fumigation extraction technique that kills microbes and lyses their cells and then measure the amount of carbon, nitrogen or phosphorus released. Substrate-induced respiration can measure potential microbial activity in the soil; sugar is added to soils to stimulate microbial activity and the difference between amended and non-amended soils indicates microbial biomass. Enzyme activity or ATP assays are also used as measures of microbial abundance. Numerous other techniques can provide information about genetic diversity and structure of soil microbial communities.

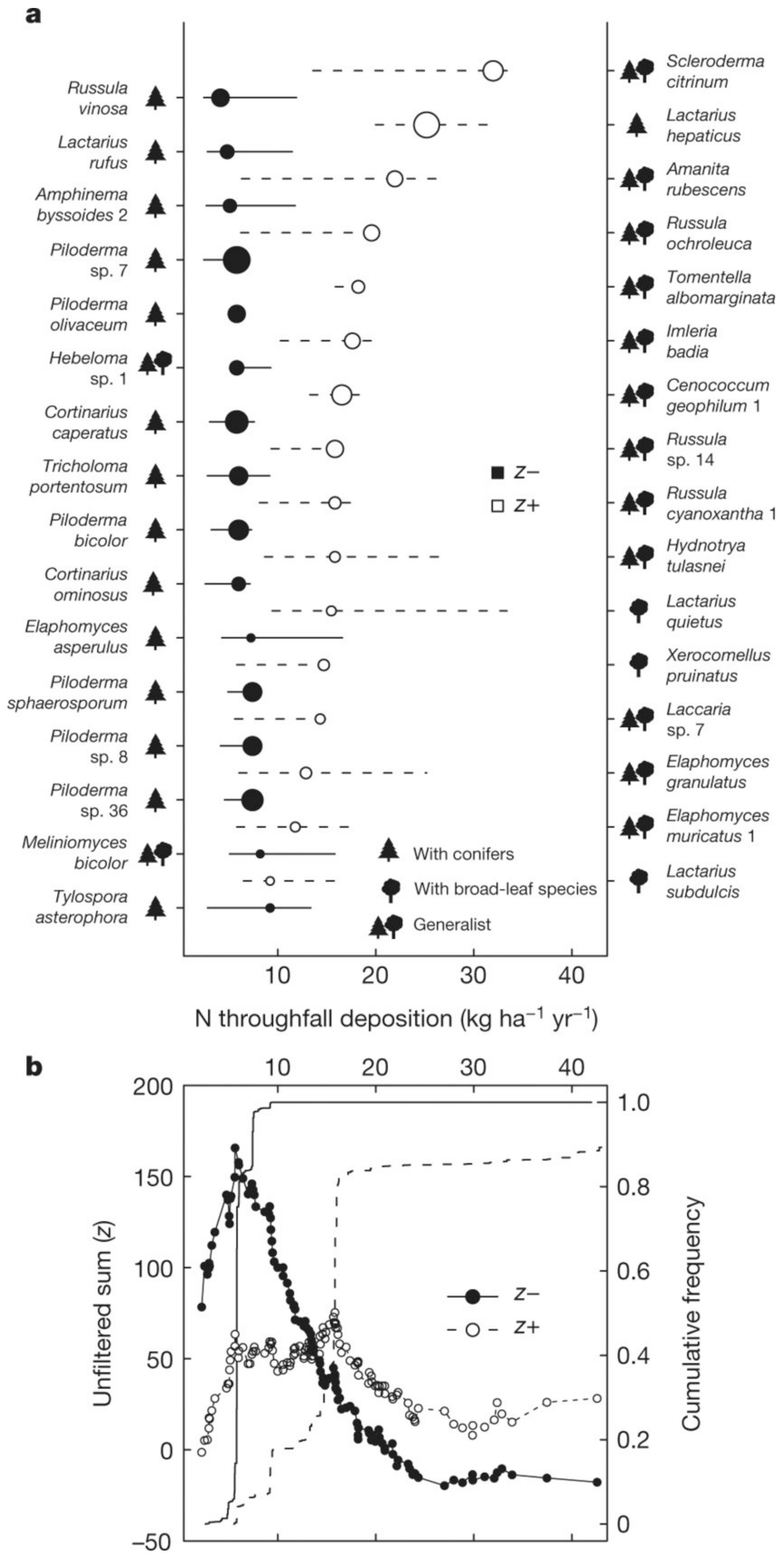


Figure 4.2.2. a) The below-ground abundances of individual ectomycorrhizal (EcM) species in relation to atmospheric nitrogen (N) deposition across 137 intensively monitored ICP Forests plots in Europe. Black symbols show species declining with increasing N deposition (z−) and open symbols depict species increasing with increasing N deposition (z+). The symbol size is proportional to the magnitude of the response (z-score). The horizontal lines represent 5th and 95th percent quantiles of values resulting in the largest change in species z-scores among 1,000 bootstrap replicates. Tree shapes next to species names indicate host generalist, conifer-specific, or broadleaf-specific species. b) In response to atmospheric N deposition, mycorrhizal communities shift dramatically at 5.8 kg ha^{-1} yr^{-1}, with a secondary shift for positively affected fungi at 15.5 kg ha^{-1} yr^{-1}, based on the community-level output of accumulated z-scores per plot. Source: van der Linde et al. 2018. Copyright © 2018, Macmillan Publishers Ltd., part of Springer Nature.

Fewer studies have explored the response of soil food webs (Groffman & Bohlen 1999; Lokupitaya et al. 2000). Several studies found nematodes to be more abundant, but less diverse and successionally less mature in long-term fertilised plots (Shaw et al. 2019; Eisenhauer et al. 2012). Other studies, however, found increased diversity of nematodes, collembolans, mites and enchytraeids (van der Wal et al. 2009; Cusack et al. 2011). In general, N enrichment changes nematode community structure and simplifies soil food webs, raising the possibility that soil fauna control microbial activities (Shaw et al. 2019; Eisenhauer et al. 2012; Moore & de Ruiter 2000).

Interaction of N impacts on above and below-ground ecosystems with other factors

Long-term N addition experiments were shown to reduce soil respiration and soil microbial biomass (Treseder 2008), increase mortality under drought conditions and slightly increased above-ground biomass production and thus carbon storage (Magill et al. 2004). In extensive inventory data for the eastern United States, N enhanced the growth rates of some species, decreased the growth rates of others, and stimulated above-ground carbon stored in biomass (Thomas et al. 2010; Horn et al. 2019). In a modeling study, net ecosystem production increased linearly with N deposition increases in six out of seven forested watersheds in the United States, while nitrous oxide (N_2O) flux was related to N deposition only under warming scenarios (Hartman et al. 2014).

There are independent or partially independent factors that affect the strength of the nitrogen impacts (e.g., climate change, ozone, sulphur deposition). The potential effect from soil acidification is influenced by soil mineralogy, base cation weathering rates, climate, soil pH and presence of acid-sensitive versus acid-tolerant species. Atmospheric deposition of NO, NO_2, NH_3 and NH_4^+ can be directly phytotoxic (See Section 4.21). Deposition as HNO_3 can acidify soils once the molecule is dissociated to H^+ and NO_3^- in soil solution. Both oxidised and reduced N can acidify soils: Oxidised N as a mobile anion associating with and then leaching with base cations (Reuss and Johnson 2012), and reduced N through the acidifying effects of both nitrification and root exchange of NH_4^+ for H^+ to maintain charge balance (Bolan et al. 1991). This effect is reduced where atmospheric inputs occur as gaseous NH_3, which is alkaline. NO_3^- is more bioavailable for many plants than NH_4^+ because it is more loosely held on soil exchange sites. Nitrogen can also be more easily lost via leaching for the same reasons, especially in regions that are limited by phosphorus (Figure 4.2.3). Deposition as reduced dry N (e.g., NH_3) is susceptible to re-volatilisation, whereas wet deposition is readily adsorbed to physical surfaces (Pleim et al. 2013). Wet N deposition may also be readily incorporated into the soil solution, where it can have a variety of ecological effects.

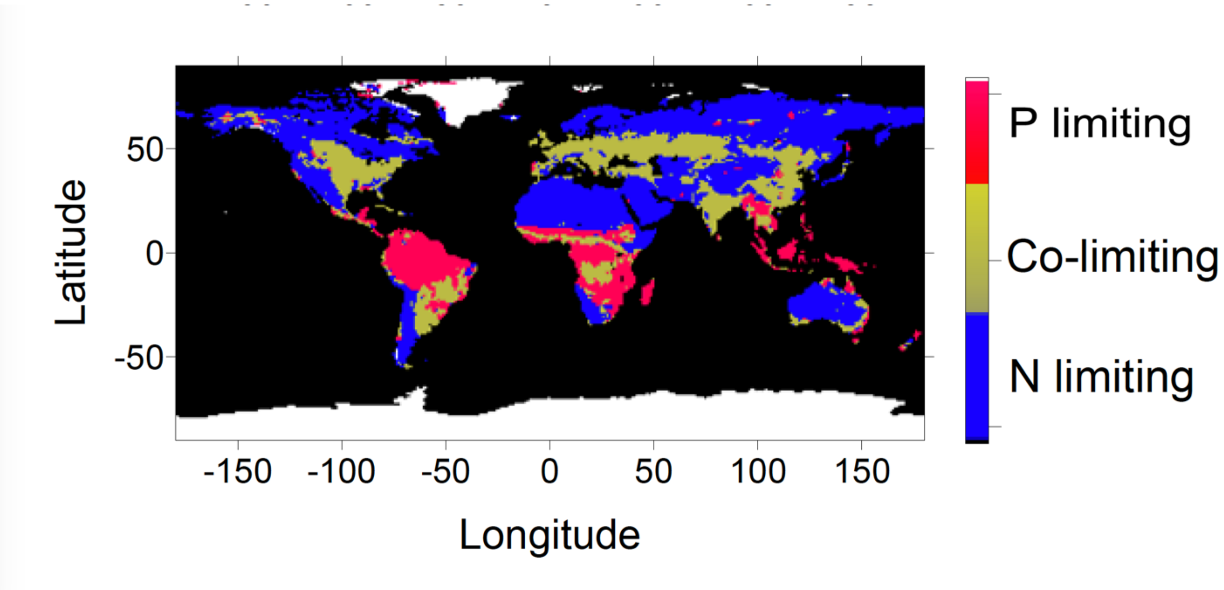


Figure 4.2.3. Estimates of where ecosystems are naturally limited by nitrogen (N), phosphorus (P) or both, from dynamic global models (Wang et al. 2010). This work is distributed under the Creative Commons Attribution 3.0 License.

4.2.3 Soil acidification

Soil acidification, defined as a decrease in soil acid neutralizing capacity (ANC; van Breemen et al. 1984; De Vries and Breeuwsma 1987), is a major consequence of elevated rates of atmospheric deposition in natural and seminatural soils. Similarly, soil acidification occurs in agricultural soils in response to N fertilisation (Zhu et al. 2018), thus requiring the need of liming (Xu et al. 2020). The loss of ANC ultimately leads to a decrease in soil pH in both natural and agricultural soils (e.g., Guo et al. 2010). Changes in soil pH are dependent on the buffering capacity of the soil (Ulrich 1986). Soil acidification can lead to aluminum toxicity in plants and animals at low pH levels (typically below pH 4.5); nutrient imbalances in vegetation; and acidification of aquatic ecosystems. One study provides an overview of the abiotic impacts of acid deposition on soils and waters with their potential impacts on terrestrial and aquatic ecosystems and the related geochemical indicators — pH, base saturation, aluminum concentration, aluminum/base cation ratios — with derived critical limits for those indicators (De Vries et al. 2015). These results have been observed and validated over many decades and our confidence in the pressure-impact relationship is high.

Soil acidification resulting from N inputs has potentially severe and long-term impacts on soil fertility and food production. These impacts arise from two interrelated factors: Decreased availability of essential plant nutrients and increased availability and uptake of harmful metals by plants. These factors can significantly suppress crop and forage yields and decrease the productivity of natural ecosystems while leading to long-term soil impairment that may be difficult or very expensive to reverse.

Data syntheses of soil pH measurements under agriculture provide insights into the rates and extent of soil acidification driven by N inputs in the context of natural variation in soil properties. Globally, at least 30% of potentially arable land has pH < 5.5, indicating potential acid stress (von Uexküll & Mutert 1995), and a much larger fraction is vulnerable to N-induced acidification. For example, across Chinese agricultural systems, soil pH declined by 0.3 – 0.8 units over three decades across five of the six major agricultural soil types, including all major crop types (Guo et al. 2010). The very high N fertilisation rates in this region corresponded to inputs of 20 — 220 kmol H^+ ha^{-1} y^{-1}, and only soils with very high carbonate content resisted significant declines in pH during this period. If these fertilisation rates continue, as much as 13% of Chinese agricultural land may suffer from aluminium ion (Al^{3+}) toxicity by 2050 (Zhu et al. 2018). In southeastern Australia, approximately 1/3 of agricultural soils had pH < 5 in 1990, sufficient to suppress yields of sensitive crops, and most soils were predicted to experience major pH decline (~1 unit) from even moderate acid inputs (3 – 6 kmol H^+ ha^{-1} y^{-1}) over the following decades (Helyar et al. 1990). In Oklahoma, USA, a substantial fraction of soils under wheat cultivation has been impacted by acidification over the last several decades, where 28 to 39% of statewide soil samples had pH < 5.5, indicating likely yield suppression of wheat and forage production without additions of lime to counter the acidification effect (Juo et al. 1995; Lollato et al. 2019; Zhang et al. 1998).

Processes by which N affects a decline in acid neutralizing capacity (ANC)

Reactive N additions can generate acidity via multiple biogeochemical pathways. The magnitude of acidification depends on the chemical forms of N added, the forms of N taken up by the vegetation, and the extent to which excess N is lost to leaching as NO_3^- (Matson et al. 1999). When plants assimilate NH_4^+, they acidify the soil by releasing a proton (H^+). This effect does not apply where the atmospheric input is as dry deposited NH_3. When NH_4^+ is converted to NO_3^- during the microbial process of nitrification, two protons are produced (or one proton in the case of nitrification originating from NH_3).

If a plant takes up this NO_3^-, it consumes H^+ to balance the negative charge of the NO_3^-, and there is no net change in soil pH for NH_3, and a smaller acidifying effect of NH_4^+. If NO_3^- is present in soil in excess of plant demand, it is vulnerable to leaching, which typically occurs along with a base cation (e.g., Ca^{2+}, Mg^{2+}, K^+). Thus, the loss of NO_3- and associated base cations during leaching is a critical process in soil acidification. In this scenario, the protons generated during nitrification remain in the soil, contributing to acidification. Because the fate of N is closely linked to plant N demand, soil acidification often increases *exponentially* with increasing rates of N inputs, as plant N demand becomes saturated (Tian & Niu 2015). While synthetic

N fertilizer is essential to the productivity of most modern agricultural systems, it presents long-term risks to soil fertility, such as soil acidification, that remain costly and difficult to mediate in many cases.

In calcareous soils, protons (H^+) are exchanged on the soil complex for ions such as bicarbonate (HCO_3^-) and calcium (Ca^{2+}), thereby buffering the soil against acidifying processes. HCO_3^- and Ca^{2+} ions leach from the system, but the pH remains the same until almost all the calcium carbonate has been depleted. In soils dominated by silicate minerals, buffering is taken over by cation exchange processes of the soil adsorption complexes. In these soils, protons are exchanged for Ca^{2+}, magnesium (Mg^{2+}) and potassium (K^+), and these cations are leached from the soil together with anions (mostly nitrate or sulphate). Buffering by cations and leaching will continue until all base cations are exchanged. Continuing acidification will lead to a shift in the buffer range of the soil from base cation buffering to aluminum (Al^{3+}) buffering (pH < 4.5). In mineral soils with a large cation exchange capacity and high base saturation, the cation buffering may continue for several decades, even at relatively high acid inputs. At low pH (< 4.5), hydrous oxides of several metals start to dissolve. This causes a strong increase in dissolved concentrations of Al^{3+} (Ulrich 1981) and other metals, such as manganese.

Methods for determining soil acidification rates and pH changes from N

Nitrogen fertilisation or N deposition causes proton production (and related soil acidification due to buffering this proton production with related ANC decline) when N enters the soil as NH_4^+ (by atmospheric deposition or when N is added as an NH_4^+ fertilizer) and typically leaves the soil as NO_3^-. The N induced acid (proton) production rate can be quantified as $H_{pro,N} = NH_4^+{}_{in} - NH_4^+{}_{le} + NO_3^-{}_{le} - NO_3^-{}_{in}$. (De Vries & Breeuwsma 1987; Zhu et al. 2018). These points illustrate the situation where:

- N is applied as NH_4^+ and this is followed by transformation of NH_4^+ to NO_3^- by nitrification and then by NO_3^- leaching. This produces two protons (H^+), whereas one proton is produced when N is applied as NH_4^+ and removed by NH_4^+ uptake, NH_3 emission or NH_4^+ retention.
- N is added as gaseous NH_3 by atmospheric dry deposition. Initially, this consumes one proton in naturally acidic systems, where NH_3 is converted to NH_4^+ tending to reduce acidity of leaf surfaces. If this is followed by nitrification to NO_3^- in the soil accompanied by NO_3^- leaching, then one proton (H^+) is release overall, contributing to acidification. Where deposited NH_3 is taken up by plants or immobilized in soils (as $R\text{-}NH_2$ forms), there is little contribution to soil acidification.
- N is applied as NO_3^- and leached as NO_3^- there is no acidification, while uptake of NO_3^- (no leaching) implies the consumption of one proton.
- N is applied as organic N (manure) or urea-N input (no input of NH_4^+ nor NO_3^-) and this is followed by mineralisation (organic) or hydrolysis (urea) to NH_4^+ (implies consumption of one proton) followed to transformation to NO_3^- by nitrification (implies production of two protons) and then by NO_3^- leaching, this causes the net production of one proton. Note that organic N and urea can be treated as if it was NH_4NO_3.

This approach can be used to quantify soil acidification based on a quantification of input-output budgets in natural (forest) ecosystems (De Vries et al. 2007) and in agricultural soils (Hao et al. 2018; Zhu et al. 2018), but also in modeling approaches (see Reinds et al. 2009 for forest ecosystems and Xu et al. 2020 for agricultural systems).

Considering the effects of added N forms on soils acidification it is important to distinguish between consequences of the 'actual deposited species' and the 'equivalent emitted pollutants' (Sutton and Fowler 1993). The first approach considers the acidifying consequences of the chemical species as they are added to ecosystems. The second approach considers the acidifying consequences according to the chemical form in which they were originally emitted. In this way it can be seen that NH_4^+ is more acidifying for soils than NH_3 (when expressed as the 'actual deposited species'), while NO_2 and HNO_3 are more acidifying than NO_3^-. This explains what may appear surprising, that NH_4^+ is acidifying but NO_3^- is not. It should be recognized however, that NH_x is generally emitted as NH_3, while NO_y (i.e., all oxidized N) is generally emitted NO and NO_2 (i.e. NO_x). This means that according to the 'equivalent emitted pollutant', the additional acidifying proton of NH_4^+ actually originated (prior to air chemistry transformations) as SO_2, NO_x

emission etc, which oxidize to form sulphuric acid and nitric acid. Following atmospheric reaction of NH_3 with these acids, the NH_4^+ thus carries part of the acidifying effect of SO_2 and NO_x into ecosystems.

There are a variety of approaches used to examine how N addition affects soil pH changes in response to soil acidification (ANC decrease), including experiments that use N fertilizers applied at different rates and different forms, studies that take advantage of spatial gradients in atmospheric deposition, and long-term studies that examine the changes in pH and soil acidification over time.

Scientists often apply N fertilizers to mimic elevated inputs from atmospheric deposition or agricultural fertilisation and then examine resulting changes in soil acidification (Tian and Niu 2015). Within N fertilisation experiments, researchers often include control (or reference) plots and plots with added N. To isolate ecological effects of N-linked acidification from the impacts of N as a plant and microbial nutrient, some studies have directly manipulated soil pH separately from N. Such studies have revealed that N-linked acidification often dominates observed changes in plant and soil processes over timescales of years to decades (Ye et al. 2018). Study duration varies from days to more than 20 years and application rates vary from <5 g to more than 15 g N m^{-2} yr^{-1}. Forms of N inputs include ammonium, nitrate and urea, and these have occurred over several biomes, including grasslands and forests. Many long-term experiments have indeed shown that soil pH has significantly decreased due to long-term application of ammonium- or urea-based N fertilizers in croplands, especially in soils with a low H^+-buffer capacity (Miao et al. 2011; Zhang et al. 2015). This difference is also illustrated in Figure 4.2.4. It should be noted that the choice of compound used to add N in experiments can also affect soil pH. Thus, ammonium sulphate leads to stronger acidification due to the addition of the sulphate ion. Similarly, adding nitrate as sodium nitrate can lead to an increase in pH partly due to the addition of sodium (noting from above that addition of NO_3^- itself is not acidifying).

Atmospheric deposition gradients allow researchers to examine the response of soil acidification to changes in N availability across relatively large spatial and long temporal scales. Along these gradients, researchers often set up plots to measure soil pH and other ecosystem responses (Chapters 4.2.1, 4.2.2 and 4.2.4).

Within each site along N deposition gradients or across experimental plots, multiple soil samples are collected, all to the same depth and on similar dates when possible. To collect soils, researchers often gather soil cores with a hollow tube or auger to a relatively shallow depth (often to 10-30 cm) or dig large soil pits from which soils are sampled at different horizons. Soil pH can be measured in the field with *in situ* electrodes or soils can be sampled and brought back to the laboratory for analysis. Additionally, pH and acid neutralizing capacity of soil solution can be analyzed using lysimeters or ion exchange resin bags or membranes.

Researchers often characterise soils further to support measurements of soil acidification, including bulk density, base saturation, and cation exchange capacity, which can strongly affect the buffer capacity of the soil (Zhang et al. 2015). In studies that aim to isolate the effects of N deposition on soil acidification, it is important to keep as many variables as constant as possible across plots because soil pH is influenced by many factors, including climate, vegetation, soil microbes, topography, geology, hydrology and time since disturbance.

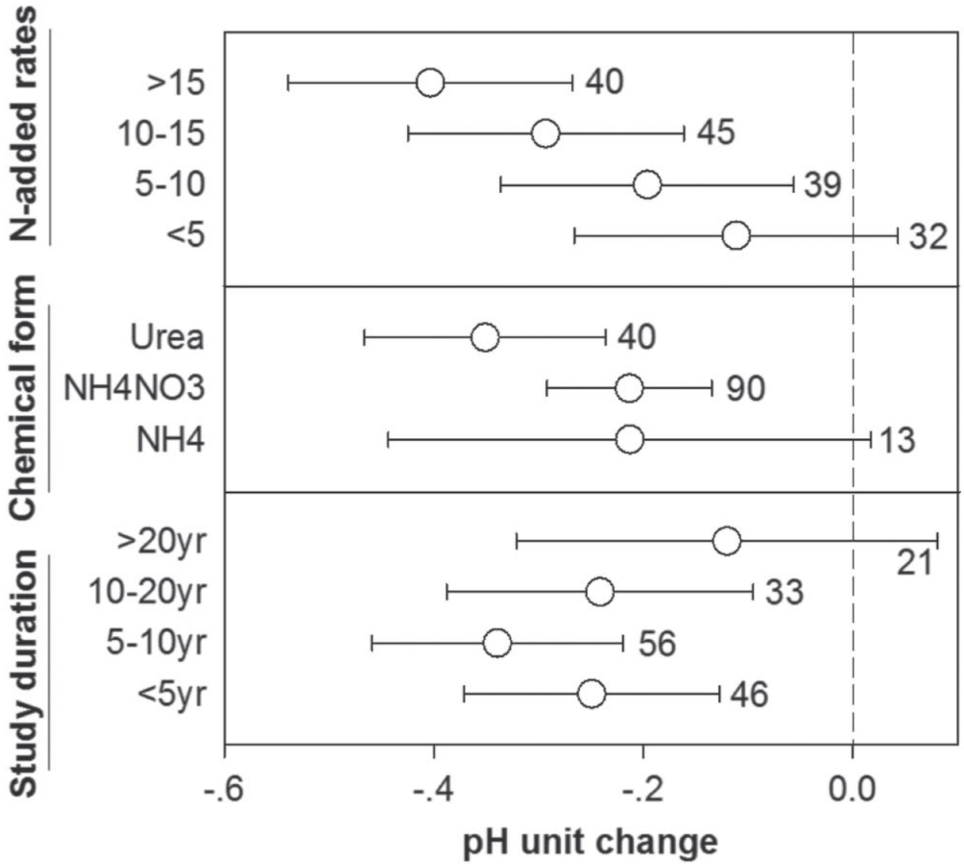


Figure 4.2.4. The effects of study duration, form of nitrogen (N) addition and rates of N application on soil pH change. The number next to the dots is the sample size of each variable. The unit of the N-added rates is g m^{-2} $year^{-1}$. Dots represent soil pH unit changes with 95% confidence intervals (CI). Source: Tian & Niu 2015. This work is distributed under the Creative Commons Attribution 3.0 License.

4.2.4 Soil nitrogen saturation and leaching losses

Soil quality can be affected by either too much or too little N. Too little N can inhibit above-ground productivity in agriculture or native vegetation and is a major cause of food insecurity in some parts of the world. It should also be noted that limited N availability in natural ecosystems is a primary mechanism for their rich biodiversity (Chapter 4.2.1). Excess N in soils can lead to soil acidification, produce changes to soil C:N ratios, influence ecosystem processes such as C sequestration in soil organic matter or in above-ground vegetation, alter N and C gas flux, and shift soil biota communities. There are methods available to determine indicators and develop Pressure-Impact relations (Figure 1.4). 'Nitrogen saturation' is the term used to describe the availability of inorganic N in excess of plant demand (Aber et al. 1989).

Soil N enrichment leads to increased terrestrial ecosystem N via increased above- and below-ground N storage (Lu et al. 2010). However, the increase in stored N is usually less than the full application due to losses from the system through leaching or nitrification and denitrification (Aber et al. 1998; Dise and Wright 1995; Peterjohn et al. 1996). The rate of loss in agricultural systems accelerates with higher fertilizer application rates and N losses are greatest where soil N is high to begin with (Wang et al. 2019). Nitrate leaching from terrestrial systems results in pollution of groundwater (Spalding & Exner 1993) and surface water ecosystems (Fleischer et al. 1987). This section considers losses of nitrate through leaching, while denitrification losses are addressed in Chapter 4.3.2.

Nitrogen enrichment studies have led to the elaboration of the N saturation concept, whereby N inputs exceed the uptake or storage capacity of an ecosystem and lead to downstream N export (Ågren & Bosatta 1988). Figure 4.2.5 illustrates the trade-off between the positive impact of nitrogen enrichment on forest growth and related carbon sequestration on the one hand, and the negative impact of nitrogen enrichment on biodiversity loss and on nitrogen losses with related soil acidification and water quality degradation on the other hand. More information on the various ranges for the N deposition level at which plant species diversity, N retention and forest growth are optimal has been given elsewhere (De Vries & Schulte-Uebbing 2020).

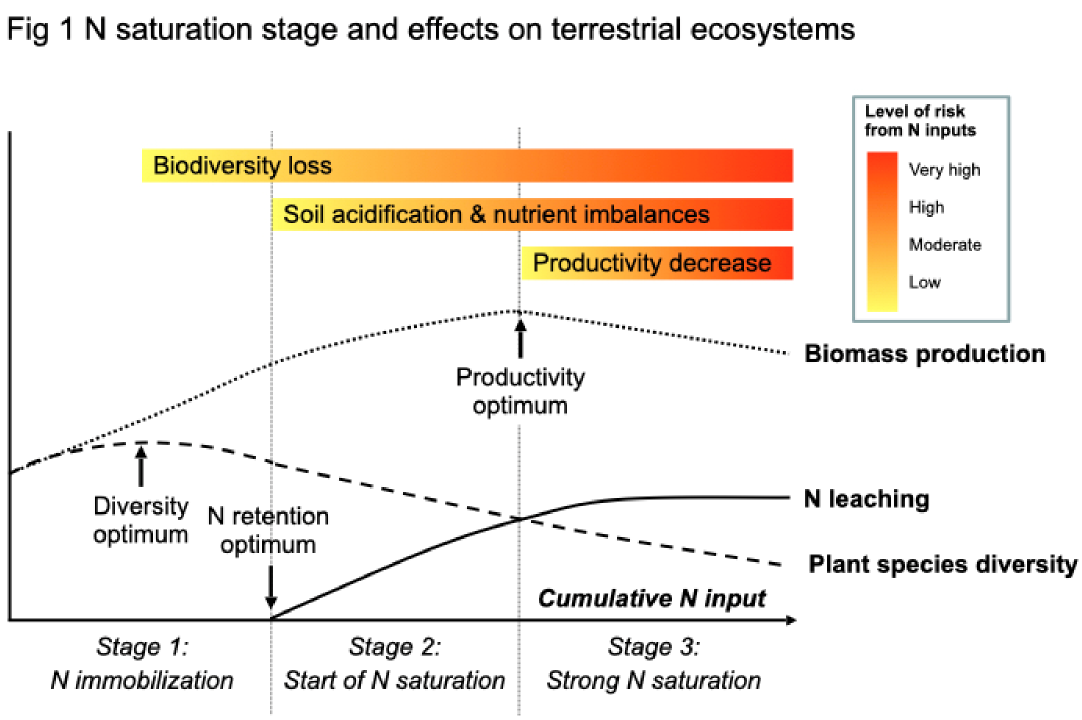


Figure 4.2.5. Conceptual diagrams of the relationship between the stage of nitrogen (N) saturation and the effects on terrestrial ecosystems in terms of soil processes, vegetation changes and growth. This figure is an update of the figure by Aber et al. (2003) by De Vries and Schulte-Uebbing (2020). 'Cumulative N input' on the x-axis refers to cumulative input at a constant annual rate.

Processes by which N enrichment influences soils and leaching

Nitrogen enrichment of soils leads to losses of highly soluble nitrate (NO_3^-), regardless of the N source. Nitrate forms by the microbially-mediated oxidation of ammonium (NH_4^+), which itself derives from the mineralisation of organic N. Because the progression from organic N to ammonium to nitrate is thermodynamically favourable, nitrate is the dominant form of soluble N in most aerated soils, produced whether N enrichment occurs by N deposition or fertilizer application (Erisman et al. 2008). When soils are enriched with N, nitrate leaching rates become controlled by rates of mineralisation and nitrification, the processes that may ultimately produce nitrate (Kirchmann et al. 2002).

The amount of NO_3^- leaching that occurs under N enrichment ultimately depends on the factors that control nitrification and mineralisation. Soil depth regulates these processes because moisture and soil organic matter (SOM) content are positively correlated with, and vary with, depth, often being relatively high just below the soil surface (Schimel & Parton 1986). Soils with higher C and N content tend to have higher mineralisation and nitrification rates, while at a given C content the C:N ratio of SOM negatively correlates with N mineralisation (Booth et al. 2005). At lower C:N ratios (generally 23-30), NO_3^- concentrations will tend to be higher because the supply of N to microbial decomposers outstrips the N input required to mineralize the available C (Gundersen et al. 1998; MacDonald et al. 2002; De Vries et al. 2007; Van der Salm et al. 2007; Dise, 2009). The microbial community itself exerts a powerful control on gross rates of nitrification and limits NO_3^- leaching through assimilation uptake (Stark & Hart 1997; Bengtsson et al. 2003).

In addition to NO_3^-, dissolved organic N (DON) leaches from N enriched soils, contributing to downstream N burdens (Baron et al. 2013). DON losses have not been as historically well-studied as NO_3^- losses, but some researchers have found they may contribute up to 28% of total dissolved N in agricultural soils (van Kessel et al. 2009), or from 14-38% of total N addition (Hussain et al. 2020). As with NO_3^-, some forms of DON may be taken up by plants, while others flush more readily, and all forms are susceptible to loss during high runoff events (Neff et al. 2003). Studies of N losses via soil water pathways should consider both NO_3^- and DON concentrations to fully constrain N budgets.

Methods for determining soil N enrichment and loss

As with many other Pressure-Impact relationships (Figure 1.4), methods for determining the effects of soil N enrichment and loss have been studied using experimental methods, input-output budgets and modeling. Long-term N fertilisation experiments like NITREX (https://cordis.europa.eu/project/id/STEP0056) used varied amounts and types of N input in varied geographic locations to identify ecosystem responses, including soil responses to N as well as N input-output budgets across Europe (Bredemeier et al. 1998). Field-scale agricultural experiments are used to test the impact on N loss of a wide range of agricultural practices, including crop conversion (Smith et al. 2013), altered tillage (Stenberg et al. 1999) and amendment applications (Svoboda et al. 2013). Some of these experiments have found N enrichment increases the flux of N_2O to the atmosphere in croplands and drylands (Lafuente et al. 2020; Liu & Greaver 2009). Long-term monitoring programmes such as the called ICP (International Cooperative Programme) Forests level-II dataset with more than 800 forested sites involved has also led to many insights in soil nitrogen enrichment and N losses (De Vries et al. 2007; Van der Salm et al. 2007; Dise et al. 2009; Schmitz et al. 2019).

Nitrogen budgets may be used to develop regional or national models of N balance and test scenarios for altered N inputs, input methods, or mitigation techniques (Velthof et al., 2009; Howarth et al. 2012). Simple models are deployed at the farm scale to leverage empirical data for management decision making (Cameron et al. 2013). In a comparison of seven of the available models for this scale, one study found that choice of model had a significant influence on predicted N losses (Henryson et al. 2020).

4.2.5 Ozone effects on vegetation

Nitrogen oxides (NO_x) are a major precursor to the formation of tropospheric ozone. Tropospheric ozone (O_3) is considered the most important and pervasive air pollutant affecting vegetation at a global scale

(Ashmore 2005; The Royal Society 2008; Fuhrer et al. 2016). This section describes mechanisms by which O_3 alters plant processes in natural and agricultural vegetation. Ozone effects are observed across spatial scales, from cells to whole plants finally affecting ecosystem-level processes. Visible foliar injury induced by O_3 has been well characterised and documented over several decades on many tree, shrub, herbaceous and crop species (Hayes et al. 2007; http://www.ozoneinjury.org). Ambient O_3 concentrations decrease photosynthetic rates at leaf level and accelerate plant senescence. Different meta-analyses on crops and tree species have reported reductions of maximum rates of photosynthesis and stomatal conductance, and increased rates of respiration (Wittig et al. 2007; Ainsworth et al. 2012; Li et al. 2017). Besides impacts on plant productivity, O_3 affects reproductive development (Black et al. 2000). Ozone can diminish flower and seed production, pollen germination and seed quality reducing reproductive fitness of sensitive species (EPA 2013; Mills et al. 2013). Ozone induces changes in the timing of flowering, which may affect reproductive success of plants by causing asynchrony between plants and their insect pollinators. Because of decades of research into the many consequences of ozone to vegetation, our confidence in the effects of ozone is high.

Ozone can reduce agricultural yield significantly through different mechanisms (Ainsworth 2017; Emberson et al. 2018). Visible injury for those species with a market value based on their visible appearance can cause an immediate loss of economic value. This is the case of some leafy crops affected by O_3 such as spinach (González-Fernández et al. 2016) or lettuce (Calatayud et al. 2004; Goumenaki et al. 2007). But O_3 can reduce the marketable value of many crop species in the absence of visible injury. Staple crops like wheat are among the most O_3 sensitive crops (Mills & Harmens 2011; CLRTAP 2017), being currently used as the representative crop species for O_3 risk assessments at regional, continental, and global scales (Mills et al. 2007; Ainsworth et al. 2012). The wide range of the intraspecific O_3 sensitivity provides mitigation options through the selection of tolerant cultivars.

The effects of O_3 on the growth of forest tree species have been assessed mainly with short-term studies (maximum a few years in organisms that live several decades or centuries) using individual seedlings growing under non-competitive conditions. Some meta-analyses have reported reductions from 7% in northern temperate tree species of Europe and North America (Witting et al. 2009) to 14% in China considering both temperate and subtropical species (Li et al. 2017). The few studies investigating the effects of O_3 on mature trees in open-air experiments generally confirm the results obtained in studies using chamber conditions, but highlight the relevance of interactive factors such as environmental conditions and competition (Karnosky et al. 2003; Pretzsch et al. 2010). The responses obtained in experimental studies of the effects of O_3 on photosynthesis and growth have been combined with mechanistic models to evaluate the impacts of O_3 on C sequestration in the living biomass of trees and in the ecosystems. Different models have estimated that O_3 reduced the productivity of various ecosystems, but the severity of these impacts varies among plant communities and is influenced by multiple interactions with biological and environmental factors (Harmens & Mills, 2012; EPA 2013). Most studies about ozone effects on semi-natural vegetation are based on single or artificial mixtures of early established species and only a few studies have been performed with intact communities (Weigel et al. 2015). Importantly, no information is available on the O_3 sensitivity of species or communities, including biodiversity, for many major biomes of the Earth.

Relative to the direct impact of O_3 on short-term plant growth, other O_3 impacts on plant phytochemistry and nutrient cycling at the ecosystem level are less well-studied. While N availability modulates O_3 effects on biomass growth (Wyness et al. 2011; Calvete-Sogo et al. 2017), O_3 can alter N cycling through reducing biological N fixation, inorganic N assimilation and changing ecosystem N pools (Davison & Barnes 1998; Hewitt et al. 2014; Calvete-Sogo et al. 2014; Bassin et al. 2015). These changes can in turn alter population density and structure of associated herbivorous insect communities (Valkama et al. 2007). Additionally, O_3 altered the quality and quantity of litter input further changing below-ground biogeochemical cycles, soil nutrient pools, and soil emissions of greenhouse gases (Bassin et al. 2015; Fuhrer et al. 2016; Sánchez-Martín et al. 2017; Toet et al. 2017).

Ozone impacts at the ecosystem level can also be mediated through O_3-induced changes in emissions of biogenic volatile organic compounds (Peñuelas et al. 1999; Calfapietra et al. 2009; Pinto et al. 2010) that have been associated with a degradation of floral scent and reduction of pollinator attraction to flowers (McFrederick et al. 2008; Farré-Armengol et al. 2016).

Processes by which O_3 affects plant health

The rate of O_3 penetration into the leaf and the capacity of the leaf to tolerate reactive oxygen species generated from O_3 are major control points for how O_3 influences plant primary productivity (Ainsworth et al. 2012). Through O_3-induced reduction of C metabolism and C allocation to roots and root growth (Wittig et al. 2009; Grantz et al. 2006), plants become more susceptible to other stresses such as drought, pests or other diseases. Ozone affects water use in plants and ecosystems through several mechanisms including damage to stomatal functioning, loss of leaf area and reduction of water use efficiency and root growth.

Interactions with other drivers

In recent years, research has been focused on understanding O_3 effects at community and ecosystem levels where many other environmental factors can confound or interact with ozone. To disentangle those mechanisms, the combination of experimental results with epidemiological studies and process-based ecosystem models is required. Interactive effects of O_3 exposure on insect pests and fungal diseases could be relevant for both crop, fodder and forest productivity but need further investigation. The assessment of O_3 effects on ecosystem functioning and ecosystem services should include the interactions with other important environmental and changing factors such as climate, rising CO_2 concentrations and N deposition.

4.2.6 Nitrogen aerosols effects on forest health

Agricultural, traffic and industrial NH_3 and NO_x emissions increase the secondary formation of NH_4^+ and NO_3^- aerosols (Paulot & Jacob 2014; Pozzer et al. 2017; Wu et al. 2016), which are major components of $PM_{2.5}$ (fine inhalable particles, with diameters that 2.5 micrometers and smaller) and PM_{10} (inhalable particles, with diameters that are 10 micrometers and smaller) (Putaud et al. 2010). The indirect effects of aerosols on plants and ecosystems come from aerosol interaction with radiation and nutrient input (Grantz et al. 2003). Direct aerosol effects on the drought tolerance of plants, based on aerosol hygroscopicity, have recently been reported (Burkhardt 2010; Burkhardt et al. 2018). It may contribute to regional tree mortality and forest decline (Burkhardt et al. 2018). This impact is newly discovered, and methods have not yet evolved beyond laboratory experiments. There is potential for this to be an important consequence of N aerosols, but the uncertainty in effects at this time is high.

Processes by which N aerosols impact plant health

Aerosols act on plant leaves to wick water from inside plant cells via their stomata, contributing to desiccation. They also degrade the waxy cuticle of leaves, enhancing desiccation. Deposited hygroscopic aerosols on leaf surfaces (such as nitrate and ammonium salts) attract water vapour, lead to condensation and aerosol deliquescence, and stabilize the presence of minute amounts of liquid water under unsaturated conditions. Thin films of these highly concentrated salt solutions can overcome the hydrophobicity of leaf surfaces and penetrate the stomata of leaves, where they can join with liquid water coming from roots (Figure 4.2.6). While water normally evaporates within the substomatal cavity and exits the stomata exclusively as water vapour, liquid water may now follow the thin saline films along the stomatal walls and evaporate from the leaf surface (Figure 3.2.1). Deposited hygroscopic aerosols thus attract water vapour, deliquesce, form a wick-like structure, and drive additional water loss, thereby decreasing stomatal control, water use efficiency and plant drought tolerance (Grantz et al., 2018).

Synergistic effects with other drivers

This direct impact of air pollution on plants is likely to increase with increasing vapour pressure deficit (VPD) from climate change.

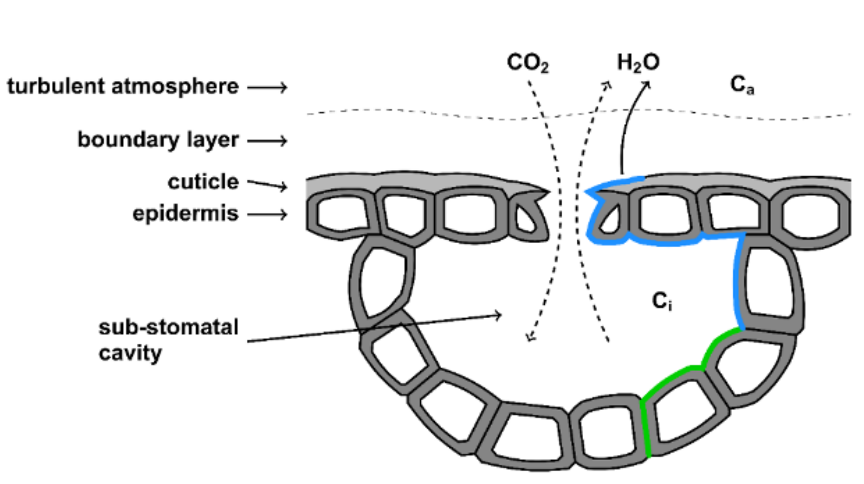


Figure 4.2.6. Hydraulic activation of stomata (HAS): Liquid water forms by condensation to hygroscopic particles on the leaf surface, from where it creeps into the substomatal cavity (blue line; thickness probably < 100 nm) and connects with apoplastic liquid water (green line). In this way, the plant's hydraulic system is extended to the leaf surface. Liquid water reaches the leaf surface from where it evaporates (solid arrow), mostly independent of stomatal aperture. Dashed arrows indicate gas exchange. C_a and C_i are external and internal CO_2 concentrations. Reprinted with permission from Burkhardt et al. (2018).

Methods used to evaluate aerosol impacts

Particulate mass concentrations and composition are routinely measured by sorption onto filters (Chow 1995; McMurry 2000; Alfarra et al. 2004), and by condensation particle counters (Burkhardt et al. 2018). Satellite infrared observations are used to detect aerosol concentrations and composition from space (Ackerman 1997). Effects on stomatal conductance and water loss are measured directly by measuring dehydration kinetics of leaves over time and gas exchange rates (Grantz et al. 2018). Waxy layer thickness is measured using scanning electron microscopy (Burkhardt et al. 2018).

4.3 Greenhouse gas balance

Increased N inputs alter the fluxes of greenhouse gases that contribute to global warming, while also impacting stocks of C in the biosphere. Increased nitrous oxide (N_2O) emissions not only contribute to warming, but also are the leading driver of stratospheric O_3 loss, which contributes to human health impacts. Elevated N deposition from the atmosphere to the land and ocean can alter ecosystem C budgets, sometimes stimulating C sequestration. The following N impacts are presented in Chapter 4.3. See the N-MIP for more information about these N impacts.

- Soil N enrichment (Section 4.3.2. Also see Section 4.5.2)
- Global warming by N_2O (Section 4.3.2)
- Climate cooling by aerosol (Section 4.3.3)
- Global warming by O_3 (Section 4.3.3)
- Ocean CO_2 emission (Section 4.3.3)
- Enhanced C sink (Section 4.3.4. Also see Section 4.5.4)
- Climate cooling by CH_4 uptake (Section 4.3.4)

4.3.1 Nitrous oxide: a potent greenhouse gas

Nitrous oxide is a trace gas that is dominantly produced by microbial processes in soil and water, with additional contributions from fossil fuel combustion, biomass burning and industry. Human activities—primarily related to agriculture—are directly or indirectly responsible for almost half of current annual N_2O emissions to the atmosphere, with natural ecosystems supplying the remainder. Recent growth in anthropogenic emissions has made N_2O the third most important greenhouse gas contributing to climate change, behind carbon dioxide (CO_2) and methane (Myhre et al. 2013). Although atmospheric mixing ratios of N_2O are small (~330 parts per billion) relative to CO_2 (~410 parts per million), a given mass of N_2O has a global warming potential currently estimated at 265 times greater than CO_2 over a 100-year timescale (Forster et al. 2021), and a long (~120 year) residence time in the atmosphere. Separate from its importance as a greenhouse gas, N_2O is also now the single greatest driver of stratospheric ozone loss (Ravishankara et al. 2009; UNEP 2013). Consequently, minimizing current and future anthropogenic N_2O emissions, largely through improved efficiencies in agricultural N application and management while reducing high fertilisation rates (Philibert et al. 2012; Thompson et al. 2019), remains a pressing goal. Given the large spatial heterogeneity in N_2O emissions that creates challenges when scaling emissions from plot to region to globe, we assign medium confidence to our ability to quantify N_2O emissions.

Processes affecting nitrous oxide emissions

Emissions of N_2O from soil and water are primarily byproducts of the microbial processes of nitrification (conversion of ammonium to nitrite and nitrate) and denitrification (conversion of nitrate and nitrite to nitric oxide, nitrous oxide, and dinitrogen). Several other biogeochemical processes can also produce significant N_2O under some circumstances. Net production of N_2O typically increases as greater amounts of inorganic nitrogen cycle through biogeochemical pathways, with absolute rates depending on a wide range of soil and ecosystem properties, especially moisture and temperature (Firestone and Davidson 1989). Cumulative emissions of N_2O from ecosystems tend to be dominated by "hot spots" and "hot moments" of disproportionately high production (Groffman et al. 2000). These result from interactions among environmental, biological and management factors, with greatest N_2O emissions typically occurring after events such as fertilisation or soil thawing (Butterbach-Bahl et al. 2013). The extraordinary spatiotemporal heterogeneity of N_2O emissions has in many cases challenged our capacity to effectively measure and model N_2O emissions (Barton et al. 2015; Gaillard et al. 2018), such that describing extant emissions and assessing the effectiveness of proposed management interventions remains a pressing area of research.

Relationships between N_2O emissions and N fertilizer input show threshold/non-linear behaviour that is strongly driven by crop demand for N, i.e., once inputs exceed demand, emissions increase non-linearly

(Figure 4.3.1; Kim et al. 2013). This has been shown convincingly in the long-term studies at the Kellogg Biological Station in Michigan USA for both maize and wheat (Millar and Robertson 2015), and in a global meta-analysis of 78 published studies (233 site years) (Shcherbak et al. 2014). These dynamics are consistent with basic principles of mass balance that provide an overarching conceptual framework for understanding how emissions are linked to the overall N cycle. Similar to N_2O, observations in many locations show that hydrologic losses of N also increase non-linearly once thresholds of input/crop demand are exceeded.

Methods for predicting N_2O emissions

An emission factor (EF) is the proportion of a given anthropogenic N input that is ultimately emitted as N_2O. Mean emissions factors derived from literature syntheses of field flux measurements (typically conducted by measuring the accumulation of gas in a chamber placed on the soil, or more recently by micrometeorological techniques) are often combined with data on fertilizer and other N inputs and management factors (e.g., land use changes) to estimate N_2O emissions over broad spatial scales (Beltran et al. 2021; Dorich et al. 2020)). This approach has been adopted by the Intergovernmental Panel on Climate Change (IPCC) (De Klein et al. 2006), although the calculation has high uncertainty, as a given mass of N can be emitted as N_2O during many steps along the "N cascade," each requiring a unique (and potentially variable) emission factor. Note that uncertainties in EFs can vary by an order of magnitude. IPCC proposes a 3-Tier methodology to estimate N_2O emissions in agricultural systems with increasing levels of complexity and data demand. Even if more expensive, the application of Tier-2 or even better Tier-3 procedures is strongly recommended (https://unfccc.int/process-and-meetings/the-paris-agreement/the-katowice-climate-package/katowice-climate-package; Thompson et al. 2019). Adequately accounting for "indirect emissions" of N_2O from N that is applied to a farm field and transported elsewhere (either leached or volatilised) is particularly challenging. At the global scale, atmospheric data imply that approximately 4% of global N generated through human activities has been emitted as N_2O (Smith et al., 2012). At national to global scales, this top-down estimate approximately matches N_2O emissions calculated using a bottom-up emissions inventory approach, but these estimates diverge markedly at smaller scales (Del Grosso et al., 2008). At the regional to local level, impacts of climate, soil properties and management factors—particularly fertilisation rates—are paramount for predicting and mitigating N_2O emissions.

Interactions with other drivers

The non-linear dynamics of N_2O emissions related to reactive N inputs are amplified by climate variability and change, whereby warmer and wetter conditions tend to increase N_2O emissions, other factors being equal. In a regional-scale analysis in the U.S. 'corn belt', Griffis et al. (2017) found large interannual variability in N_2O emissions despite relatively constant input of fertilizer. Variability was particularly high for indirect emissions of N_2O from surface waters and was strongly driven by precipitation. These dynamics are also consistent with basic principles of mass balance and with understanding of the controls of hydrologic losses of agriculturally derived N; high precipitation drives N flux across the landscape, increasing hydrologic losses and the indirect emissions of N_2O along hydrologic flow paths. Climate and management variability complicate the evaluation of mitigation practices but do not negate the basic mass balance principle that inputs of N must be keyed to plant demand to reduce the N surpluses that lead to non-linear dynamics and extraordinarily high N_2O emissions and hydrologic losses of N.

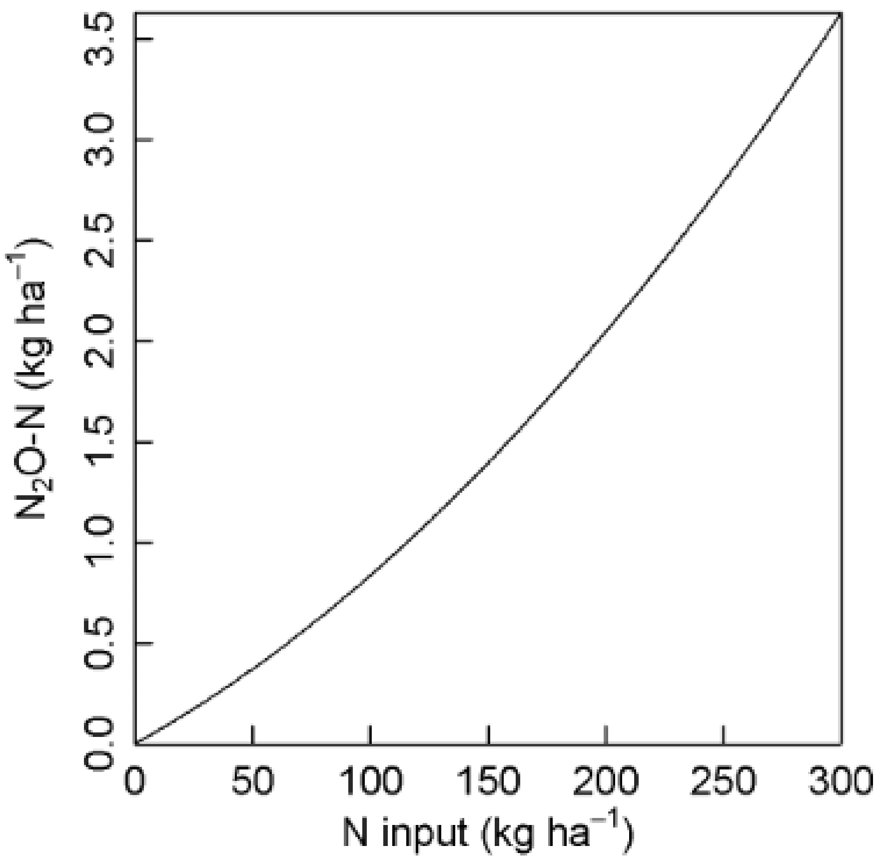


Figure 4.3.1. Non-linear relationship between nitrogen (N) inputs and N_2O emissions from agricultural systems, redrawn from the meta-analysis of Shcherbak et al. (2014). This relationship will be altered by numerous biophysical and management factors at the site scale, with potentially steeper increases under warm, wet conditions, and shallower increases in semi-arid regions. $N_2O\text{-}N = 0.001\ N\ (6.49 + 0.0187\ N)$, where N represents reactive nitrogen input. Basic principles of mass balance provide an overarching conceptual framework for understanding how N_2O emissions are linked to the overall N cycle.

4.3.2 Impact of reactive N on non-N_2O greenhouse gas balance

Impacts and Processes by which N contributes to climate forcing

While N_2O has a strong and direct effect on Earth's climate through its heat-trapping potential, oxides of N and NH_3 are short-lived influences on the chemistry of the atmosphere that contribute warming as well as cooling effects (Butterbach-Bahl et al. 2011; Pinder et al. 2013). Ammonia (NH_3) and nitrogen oxides (NO_x) contribute to climate change indirectly. They alter the production and loss of climate forcers, atmospheric constituents that perturb the Earth's energy balance by trapping heat (greenhouse gasses) or scattering incoming solar energy (aerosols). NO_x impacts greenhouse gasses by increasing the formation of ozone (O_3), contributing to warming and increasing the removal of methane (CH_4), contributing to cooling. Both NO_x and NH_3 can enhance light-scattering aerosols for a cooling effect. When deposited out of the atmosphere into ecosystems, reactive N can stimulate plant growth, enhancing carbon sequestration (Butterbach-Bahl et al. 2011; Pinder et al. 2013). The acidification of the ocean from N deposition can also lead to small, but measurable, emissions of CO_2 to the atmosphere (Doney et al. 2007).

Nitrogen oxides (NO_x) emissions are largely from combustion sources, including industry and power generation, transportation including construction, shipping and rail as well as vehicles on road. There are also emissions from agricultural and non-agricultural soils, which become more significant where mitigation efforts have been successful in controlling non-agricultural NO_x emissions (Skiba et al. 2020). Ammonia largely has an agricultural source, coming from animal manure and fertilised fields (Pinder et al. 2013). An additional source of both NO_x and NH_3 that is increasingly human influenced, is biomass burning, whether intentional for fuel or land management, or from wildfire (Galanter et al. 2000). Biomass burning is a major source of NO_x in the northern high latitudes during the summer and fall (Galanter et al. 2000), and also contributes significantly to global ammonia emissions (Whitburn et al. 2015; van Damme et al. 2018).

Methods by which N effects are estimated

Models are commonly used to estimate the radiative effects of N on aerosols, ozone and methane, and results differ substantially depending on the initial assumptions. Calculations based on instantaneous radiative forcing yield different values than those based on 20- or 100-year scales (Pinder et al. 2013; Butterbach-Bahl et al. 2011). The effects of N species on climate diminishes over longer time periods. Aerosols, ozone and methane consumption are most important for providing cooling effects in the near term, from years to a couple of decades. On a 100-year basis the effects are smaller, indicating that whereas control of short-lived components of the atmosphere may have immediate benefits for climate, considering the longer-term climate horizon, the contribution of nitrous oxide is more significant (see Figure 4.3.2). Models are also used to look at the release of CO_2 from the ocean in response to the acidifying effects of N deposition (Doney et al. 2007).

Because these models are poorly constrained by observations, researchers commonly compare model results with each other to develop a range of possible values and estimate uncertainty. There are detailed models for atmospheric radiative impacts, atmospheric N transport and deposition and ecosystem response (Pinder et al. 2013). There is uncertainty associated with each type of model, starting with uncertainties associated with the emissions estimates from each source. The time scale associated with radiative forcing, as mentioned above, adds to the range of uncertainties in interpretation of results.

Interactions with temperature and precipitation

Additional uncertainties come from the many interactions of N species with weather and climate factors, particularly short-term events and long-term trends in temperature and precipitation, and with ecosystem constraints related to water, temperature and other nutrient limitations on plant growth. Uncertainties can propagate through the calculations, lending to a range of possible effects found in Figure 4.3.2. Models that estimate the carbon sequestration effect of N deposition rely on fertilisation experiments and gradient studies (Pinder et al. 2013). A meta-analysis summarised the N fluxes and uncertainty factors for forests, grasslands, croplands and wetlands in the United States (Liu & Greaver 2010). In their critical review of 68 published studies, the authors found forests to be the most significant ecosystem for sequestering carbon in the United States, similar to findings for Europe (Butterbach-Bahl et al. 2011). Gradient studies, representative of the actual effects of N deposition on forest carbon sequestration, provide additional values for both above-ground and below-ground C storage from N deposition (Thomas et al. 2010; Butterbach-Bahl et al. 2011).

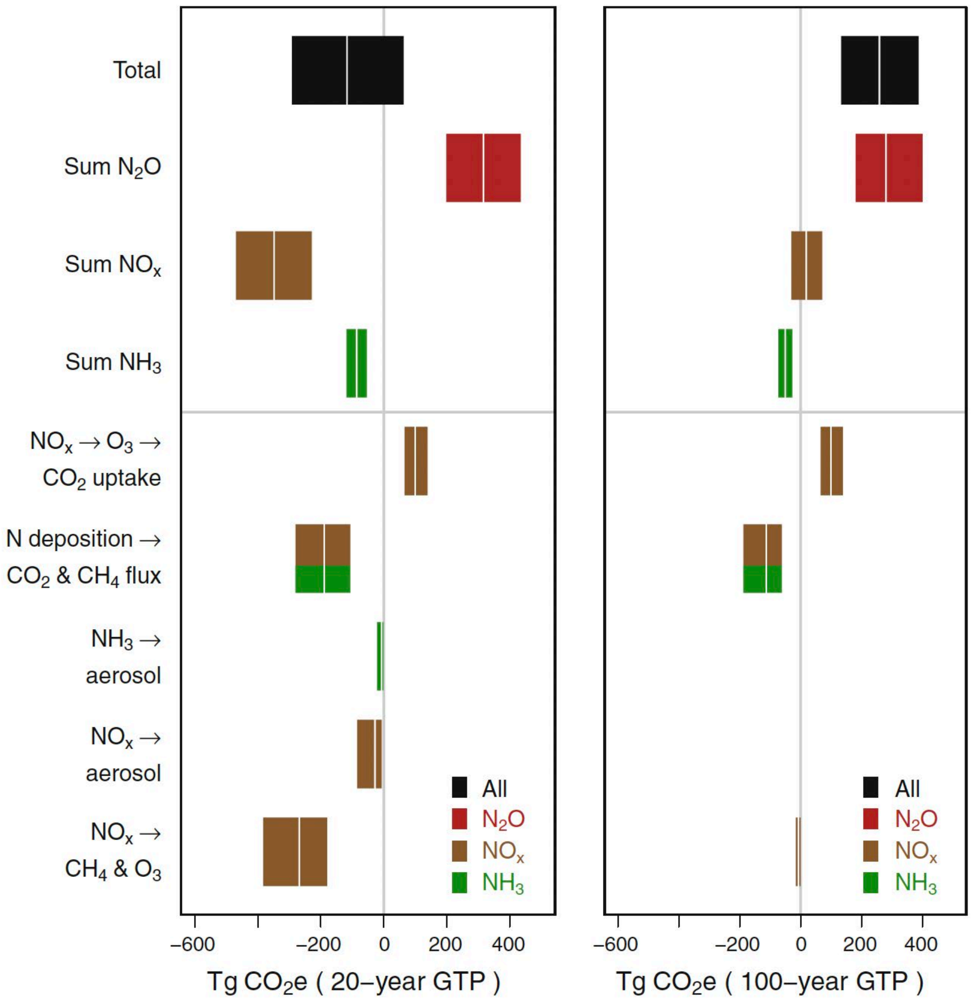


Figure 4.3.2. Climate change impact of U.S. reactive nitrogen (N) emissions, in Tg CO_2 equivalents, on a 20-y (Left) and 100-y (Right) global temperature potential basis (GTP = Global Temperature Change Potential). The length of the bar denotes the range of uncertainty, and the white line denotes the best estimate. Graphic derived from Pinder et al. 2012). Original graphic produced for this document © UKCEH 2025.

4.3.3 Impacts of reactive nitrogen on carbon accrual in plant biomass

If N is the limiting nutrient for plant growth, inputs of N can have strong positive impacts on plant productivity that presently contribute to a substantial net C sink in terrestrial ecosystems. However, the magnitude and sign of this response and its future trends hinge critically on specific characteristics of ecosystems, plant species, soil characteristics and the rate of N inputs. This section briefly summarizes relationships between N deposition (i.e., N inputs) and C accrual in plant biomass, its variation among ecosystems, and importance relative to climate forcing from other greenhouse gasses (impacts on soil C are discussed in section 4.5.3).

Processes by which N affects carbon sink in biomass

Plant growth in many forests, grasslands, wetlands and tundra ecosystems can be strongly limited by N (LeBauer & Treseder 2008; De Vries et al. 2009; Flechard et al. 2020). Accordingly, increased N deposition to wildland and managed ecosystems is responsible for increased net C uptake of approximately 112–243 Tg y^{-1} in woody biomass (Schulte-Uebbing & De Vries 2018). To appreciate the magnitude of this C sink for global climate, its impact on radiative forcing largely counteracts the warming effect of anthropogenic N_2O emissions on an annual basis (Tian et al. 2016; De Vries et al. 2017). However, it is important to recognize that increases in plant biomass C storage driven by N deposition are also vulnerable to loss during disturbances, such as drought- or pest- or fire-induced mortality, which are likely to become increasingly common in many biomes (Anderegg et al. 2015). In contrast, N_2O has climate impacts that are difficult to reverse given its atmospheric lifetime of more than 100 years (Myhre et al. 2013).

In contrast to wildlands and managed forests, agricultural systems and grasslands are less likely to exhibit consistent and significant increases in ecosystem C accrual under elevated reactive N inputs—and any C gains are likely to be offset by increased emissions of N_2O (Liu & Greaver 2009). Fertilizer N applications have increased or sustained the productivity of many annual row crop systems, such as the corn-soybean rotations common in the midwestern United States, have not commonly led to widespread net C accumulation (as crop residues or soil organic matter) over decadal timescales (David et al. 2009; Russell et al. 2009).

Impacts of N on coastal and marine C cycling and greenhouse gas production are complex and uncertain, requiring consideration of physical, biogeochemical and biological interactions. The case of coastal eutrophication illustrates how anthropogenic changes to biogeochemical cycling may be amplified by the physical environment and biological community interactions. Eutrophication in coastal waters and estuaries often results from anthropogenic N inputs, producing hypoxic conditions. When hypoxia occurs in upwelling zones, high denitrification rates produce large fluxes of N_2O (Voss et al. 2011). Coastal eutrophication also leads to algal blooms that reduce light penetration and drive community shift from seagrasses to macroalgae (Purvaja et al. 2008). Because seagrasses store much more biomass C than the macroalgae that replace them, this community shift creates a significant positive flux of C to the atmosphere (Banerjee et al. 2018).

In cases where marine ecosystems are N-limited, additional C may be deposited to sediments under enriched N conditions. Nitrogen fertilisation of plankton accounts for between one-sixth and two-thirds of the annual open-ocean C sequestration flux (Duce et al. 2008). However, the benefits of N fertilisation to climate change mitigation may be offset by increased N_2O production and may plateau as increasingly N-saturated systems become P limited (Krishnamurthy et al. 2007). Light, upwelling, chemical transformation, nutrient limitation and local ecology all mediate the net greenhouse gas impact of anthropogenic nitrogen inputs to marine systems.

Methods of assessing the impacts of reactive N on C accrual in plant biomass

The magnitude of the forest wood C sink attributed to N deposition has been estimated by combining plant biomass C and N measurements from plot-scale N fertilisation experiments with spatial models of N deposition (Schulte-Uebbing & De Vries 2018). These estimates are sensitive to many factors, including biome type, forest age and species composition, N addition rate and timing, and cumulative N loads. Importantly, not all ecosystems (or species within an ecosystem) exhibit increased growth in response to increased N deposition. Plant productivity is often limited or co-limited by other nutrients, such as P, K or micronutrients (Elser et al. 2007; Borer et al. 2017). This is especially likely in humid tropical ecosystems, which often show no increase in plant growth following N addition (Alvarez-Clare et al. 2013). In the future, drought and heat waves are increasingly likely to limit plant growth in some biomes, and interactions between these stressors and N availability are poorly understood (Gessler et al. 2017).

Plot- and landscape-scale studies have demonstrated that individual tree species often show broadly different responses to actual or simulated N deposition. For example, a fertilisation experiment at the Harvard Forest demonstrated increased productivity of hardwoods but growth suppression and mortality of pines after 15 years (Magill et al. 2004). Negative impacts of N on plant growth are often linked to soil acidification and related impacts on the availability of nutrients and toxic metals (see section 4.2.4). Similar divergent responses to N deposition have been observed at the regional scale. In a long-term comparison of forest inventory plots in the northeastern and north-central United States, N deposition was linked to enhanced growth of 11 species and decreased growth of 3 species (Thomas et al. 2010). Contrasting trends in productivity among species are predicted to continue in this region under future scenarios that incorporate effects of N deposition and climate change (Clark et al. 2023). (). Unraveling the context-specific responses of plant productivity to ongoing and future N inputs, and threshold effects linked to soil acidification, remains a key research challenge. For example, Flechard et al. (2020a) applied a multi-site

European network of long-term carbon and nitrogen flux measurements. They showed how net ecosystem productivity (ie. Increased carbon uptake from the atmosphere) of European forests increased with atmospheric N deposition, but also that high levels of nitrogen deposition reduced carbon update. The authors applied a Bayesian model approach to help untangle the different contributions to the carbon uptake (Flechard et al. 2020b).

4.3.4 Impact of reactive N on soil carbon sequestration

Human activities, such as the burning of fossil fuels and land-use change, have contributed significantly to the increasing concentrations of carbon dioxide in the atmosphere. Globally, however, soils store more C than the atmosphere and terrestrial vegetation combined (Post et al. 1990; Houghton 2007). Many researchers focus on organic carbon in soils as it originates from both living and decomposing plant, animal and microbial biomass, and it serves as a large C sink. Understanding the processes that affect below-ground cycling of C is important because any processes that alter the amount of C stored or lost from soils can influence atmospheric concentrations of carbon dioxide. In this section, we consider how inputs of reactive N interact with soil N availability and soil processes to help drive changes in soil C storage and sequestration and how these effects are commonly quantified by various available methods.

Processes by which N influences soil carbon

Increased availability of soil N from atmospheric N deposition has been shown to increase soil C storage (de Vries et al. 2009; Frey et al. 2014; Flechard et al. 2020) through reduced rates of soil decomposition and respiration (Janssens et al. 2010). These changes in soil processes are caused by reductions in lignin degradation (Berg et al. 1997) and changes in microbial composition (Fierer et al. 2007). However, the mechanisms by which N addition affects soil carbon stocks are complex. Agronomically optimum rates of N fertilisation have been shown to maximize soil carbon accumulation in maize and maize-soybean cropping systems (Poffenbarger et al. 2017) relative to sub-optimal rates. Increasing N rates from optimal to excessive in these systems had little effect on soil organic matter (Brown et al. 2014). To increase soil carbon storage, concomitant inputs of nitrogen are assumed to be necessary, with a C:N ratio of approximately 12 (van Groenigen et al. 2017). In the Broadbalk wheat experiment at Rothamsted, UK, the application of 144 kg N/ha for 135 years has increased total soil N content by 21%, or 570 kg/ha, in the top 23 cm of the soil (Glendining et al. 1996). Significant decreases in soil C accompanying long-term fertilisation may result from soil acidification (discussed in section 5.5.1) and its consequent effects on microbial metabolism and geochemical mechanisms of soil C stabilisation (Ye et al. 2017). Although some ecosystems can gain soil C following N inputs, a systematic and widespread increase in soil C stocks among ecosystems due to N may be unlikely (Greaver et al. 2016). Huang et al. (2020) indicated that N addition stimulates soil C storage both by increasing soil C input and (at high N rates) by decreasing decomposition of old soil C. Furthermore, they demonstrated that the widely reported saturating response of plant growth to N enrichment also applies to new soil C storage. A meta-analysis based on 107 datasets from 64 long-term trials from around the world revealed that mineral fertilizer application led to a 15.1% increase in the microbial biomass above levels in unfertilised control treatments (Geisseler & Scow 2014).

Methods to quantify N impacts on soil carbon

There are many techniques used by researchers to understand the interactions between N availability and soil C pools and fluxes. Two common approaches include observations along atmospheric deposition gradients and N fertilisation experiments. Atmospheric deposition gradients permit researchers to examine the response of soil C to changes in N availability across relatively large spatial and long temporal scales. Along these gradients, researchers often set up plots to measure both pools and fluxes of soil C. Within N fertilisation experiments, researchers often include control (or reference) plots and treatment plots that have varying amounts of N added, typically ranging from 50-150 kg N ha^{-1} yr^{-1}. Plot sizes range based on experimental design. For example, studies that only examine below-ground responses to added nitrogen have smaller plots, typically 5 X 5 m^2 or smaller, whereas studies that examine both above and below-ground responses often have plots sizes up to 30 X 30 m^2. Nitrogen fertilizer can be added as solid pellets or solution and either applied on top of the ground or injected into soils. The most common forms of N in fertilizer include ammonium, nitrate and urea.

The sampling approach for collecting soil samples for carbon analysis is important to consider when comparing results across studies or over time as soil cores can provide lower estimates of total soil carbon compared to soil pits (Cole & Harrison 2018). Soil C pools are often measured by these methods:

i. Use of the loss on ignition approach, which includes heating soils to more than 400 degrees Celsius to convert all organic matter to carbon dioxide and measuring the change in remaining mass with the assumption that 50% of organic matter is carbon;

ii. analysis of dried soil samples on an elemental analyzer; and

iii. use of reflectance spectroscopy to measure soil C either *in situ* or remotely.

Because soil C stocks change very slowly, examining fluxes into and out of soils can be used more readily than soil C pools alone for detecting changes in soil C over time. For example, decomposition is measured using the litter-bag technique. Soil losses of carbon dioxide to the atmosphere from soil respiration are measured using soil collars.

Researchers often characterize soils further to support measurements of soil carbon pools and fluxes and to compare results between studies (Figure 4.3.3). For example, bulk density of soils is an essential measurement to scale soil carbon content per mass soil to larger spatial scales. Other soil properties that researchers often measure in conjunction with soil carbon are soil pH, moisture and N content.

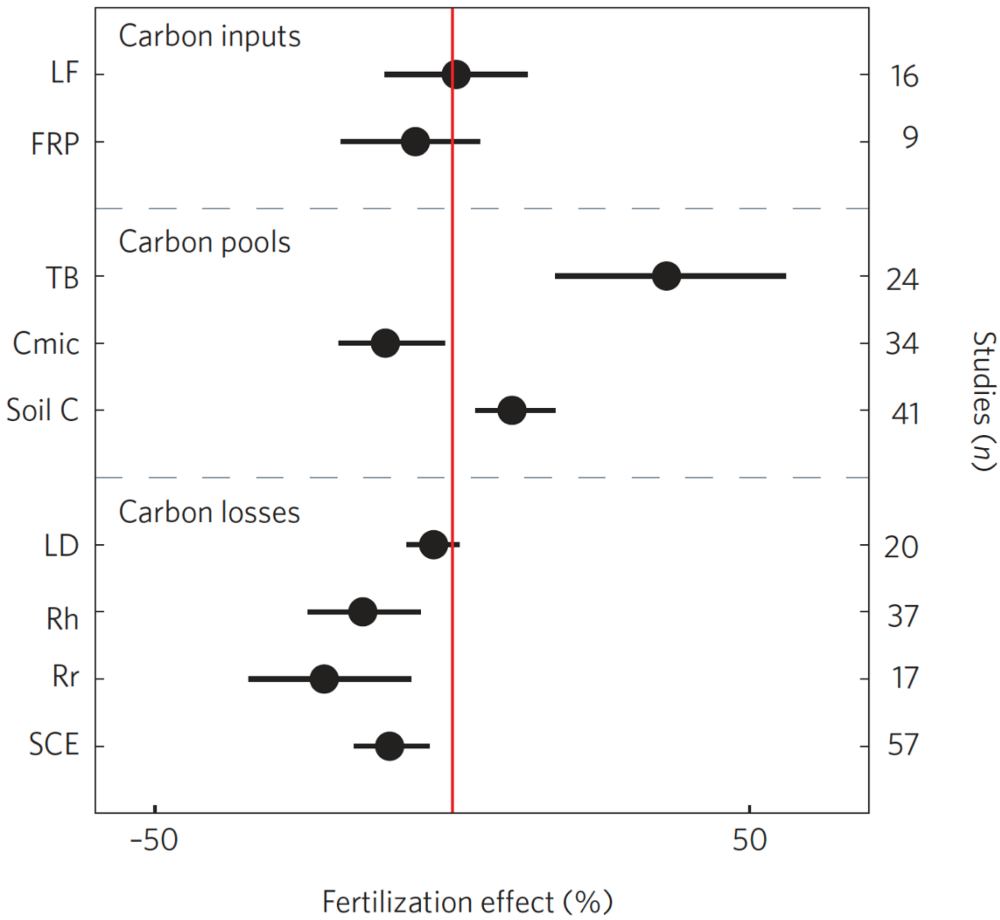


Figure 4.3.3. Effect of experimental nitrogen (N) addition on various forest carbon pools and fluxes as calculated by meta-analysis. Positive values indicate that nitrogen addition had a positive effect, negative values indicate a decrease. Error bars indicate the 95% confidence interval. Data are the weighted means for n data points (right axis). Parameters listed are carbon inputs (left axis): litter fall (LF) and fine-root production (FRP); carbon pools: total tree

biomass (TB), microbial biomass (Cmic) and soil carbon content (Soil C); and carbon losses: litter decomposition (LD), heterotrophic respiration (Rh), root respiration (Rr) and soil carbon dioxide efflux (SCE). Figure and figure caption from Janssens et al. (2010). Copyright © 2010, Nature Publishing Group.

4.4 Agricultural production

Nitrogen is an essential nutrient for agricultural products. However, excess nitrogen can lead to environmental and human health impacts, and it can also decrease the quality of food products. The following N impacts are presented in this section. See the N-MIP matrix (Supplementary materials) for more information about these N impacts.

- Increase of crop yield from N fertilizer (Section 4.4.1, Also see Section 4.5.4)
- Increase of crop yield from N deposition (Section 4.4.1, Also see Section 4.5.4)
- Increased animal protein (Section 4.4.1)
- Reduced agricultural productivity (Section 4.4.1)
- Trade-offs between intensification and biodiversity protection (Section 4.4.3)

4.4.1 Nitrogen use for agricultural production

Historically, N has been a limiting nutrient for agriculture and necessitated rotations with legumes to restore N depletion at the cost of caloric output. The creation of the Haber-Bosch process in the early 1900s greatly increased the supply of N for agriculture (Erisman et al. 2008; Galloway et al. 2013). Another strong motivation for the Haber-Bosch process was not related to agriculture — it was for production of ammonia for munitions. The Haber-Bosch process was developed by German scientists Fritz Haber and Carl Bosch in the years prior to World War I, and the discovery in Germany had significant implications for both World Wars (Smil 2002). The Haber-Bosch process was increasingly applied from the mid 1950s, at which point industrial ammonia production rates began to increase dramatically. Nitrogen fertilizer was one of the key drivers of the Green Revolution, the global increase in yields of cereals that began in the 1960s. The introduction of cultivars of wheat and rice bred for shorter straw enabled these crops to take up more N without lodging, allowing a greater fraction of photosynthetic energy capture to be translated into harvestable carbohydrate and protein.

Haber-Bosch N has had both profound benefits and detrimental consequences. It is crucial to supporting life on Earth. It has been estimated that almost 50% of people are supported by Haber-Bosch N through the food they consume (Erisman et al. 2008). However, increased availability of synthetic fertilizer from the Haber-Bosch process has also doubled the N inputs and led to a significant alteration of the global N cycle.

Processes by which N is used in agriculture

Nitrogen is an essential nutrient for all agricultural production. Food, fiber and biofuel production all require a N source for crop growth. The three main types of N sources for agriculture are biological nitrogen fixation (BNF), organic fertilizers (e.g., manure, crop residue, compost) and synthetic fertilizers. The largest proportion of N in manure is recycled synthetic N from feed and forage cultivation. Atmospheric N deposition is also a significant N input in many regions.

Biological N fixation has been increased by human activity through the increased cultivation of legume crops such as soybeans, alfalfa, pulses and clover in grasslands. Many of these crops supply their own N needs efficiently and thereby avoid N limitation. Livestock manures are often applied to cropland and used as a N source, but considerable amounts of N are lost from such manures, starting from the moment of excretion, and continuing after land application. Crop residues and composts contain carbon as well as N and can potentially contribute to the maintenance or increase of soil organic carbon.

In the terrestrial environment, synthetic N produced industrially currently produces twice as much reactive N as natural terrestrial processes of biological N fixation and lightning (Fowler et al. 2013). Of the total anthropogenic reactive N creation, the Haber-Bosch process accounts for around 57%, followed by agricultural BNF (29%) and combustion (14%) (as estimated by Fowler et al. 2013). Much of this anthropogenic N has accumulated in the environment due to inefficiencies in agricultural production and emissions from fossil fuel production. Atmospheric N deposition is usually a lower source of N in agroecosystems, but it can still influence crop production. This excess reactive N has led to a cascade of negative impacts to human and ecosystem health (Galloway et al. 2003).

Methods for assessing N status and availability in agriculture

Nitrogen metrics for agriculture generally measure soil N status, N fertilizer demand and nutrient content in agricultural products. Those metrics also can measure how much product is produced (yield) or how efficiently N is converted into a product (N surplus, N use efficiency and feed conversion ratio). Nitrogen agricultural metrics can be applied to food, fiber, and biofuel production. These agricultural metrics can be related to each other in the context of a farm N budget, which assesses a farm's N inputs and outputs. A farm N budget assesses the balance between N inputs and outputs. While the farm-gate budget assesses N inputs and outputs to a farm (Figure 4.4.1; Oenema et al. 2003; Zhang et al. 2015), the soil system budget assesses N inputs and outputs to agricultural soil. Soil N budgets are meaningful at a wide range of scales, ranging from farm, county, catchment, regional, national to global levels. Nitrogen budgets can vary in detail and complexity. The simplest example of a N budget approach is the farm-gate budget which assesses N that flows into (e.g., fertilizer seed, BNF, N deposition) and out of (e.g., farm products) the farm; this space of budget does not consider N fluxes within the farm bounds. The most detailed and complete farm budget considers all N inputs, N flows within the property (e.g., crop residue, manure management), and N outputs (see Billen et al. 2025 for a detailed definition of boundaries and inputs/outputs for different agricultural systems). For further detail about the wider range approaches see the other chapters of the INMS guidance document on nitrogen use efficiency indicators across multiple scales (Lassaletta and Sanz-Cobena 2025), as well as the INMS Guidance Document on improving nitrogen management using nitrogen budgets for dairy farming (Gourley et al. 2025).

Yield is the amount of product per unit area, measured on a mass basis. This metric is helpful when tracking or comparing productivity. The farm N surplus is the difference between farm N inputs and outputs (Oenema et al. 2003; Zhang et al. 2015). A nitrogen surplus is either accumulated on the farm, in the soil organic matter or in perennial biomass, or lost to the environment, by processes including volatilisation of ammonia, denitrification to N_2 or N_2O, or leaching and runoff. Zhang et al. (2015) found that country-level N surplus rates can increase and then decrease with economic growth over time (Figure 4.4.2a). This pattern is an environmental Kuznets curve (EKC). The lessons learned about N efficiency from countries that have already followed the EKC can be applied to countries that are just beginning to develop.

Nitrogen use efficiency (NUE) is the dimensionless ratio of nitrogen in the harvested crop to the fertilizer N inputs applied to the field (Lassaletta et al. 2014; Erisman et al. 2018). The NUE metric has an optimum level. A value greater than 0.9 is often associated with agricultural practices that deplete soil organic matter (mining N from soil). A value in the range of 0.5 to 0.9 for many crops and conditions represents N efficient production. As NUE declines below 0.5, the potential for losses to the environment increase (Oenema 2015). This metric can assess how efficiently the N invested in a cropping system is converted into harvested crop products. NUE can be tracked over time for a given farm, region or country, and it can be used as a comparison metric across farms, regions, and countries. Country-level NUE rates also follow an EKC pattern over time (i.e., NUE decreases and then increases with economic growth over time; Figure 4.4.2b). Target values for NUE, N output and N surplus depend on the type of agricultural systems and the climate-soil-environmental conditions, as well as on the type of N inputs. This means that the targets are system and location specific. Reference values of 0.5 to 0.9 for NUE, N output > 80 kg ha^{-1} $year^{-1}$, and N surplus <80 kg ha^{-1} $year^{-1}$ serve as a first attempt to arrive at such reference values based on the EU Nitrogen Expert Panel's consideration of production and environmental objectives for the example of a cropping system in Europe (Oenema 2015).

Feed conversion ratio (FCR) is specific to meat and animal productions. It is defined as the kilograms of feed required to produce 1 kilogram of animal meat or product (Oenema et al. 2006). The FCR can also be reported in units of nitrogen to focus specifically on protein use and protein production. The FCR indicates how efficiently an animal converts feed into products, which can vary based on the animal type, the diet, and the farm management system.

Uncertainties of the N budget approach (Zhang et al. 2020, 2021) can arise from potential inputs and outputs that are not counted in the N budget and also from large background amounts (Herridge et al., 2008). Herridge et al. (2008) also found that errors in quantifying the N fluxes and inaccuracies in sampling and analyzing soil can lead to substantial uncertainties in the final estimates. On a country level, N fertilizer use for countries from 1961 to present by the FAOSTAT database is often used for calculating the N budget, but is lacking how they have been used for pastures and different crops (Zhang et al. 2015). The International Fertilization Association (IFA) provides some data on fertilizer use by crops, but the data are not for each crop nor for all countries and years. Atmospheric N deposition and N fixed at a country level are often calculated using yield and N content within countries and even within farms under the assumption of a closed N budget in the system.

These farm N metrics are useful indicators of farm productivity and resource use efficiency. For a more complete picture of a farm's performance, multiple N metrics can be considered together.

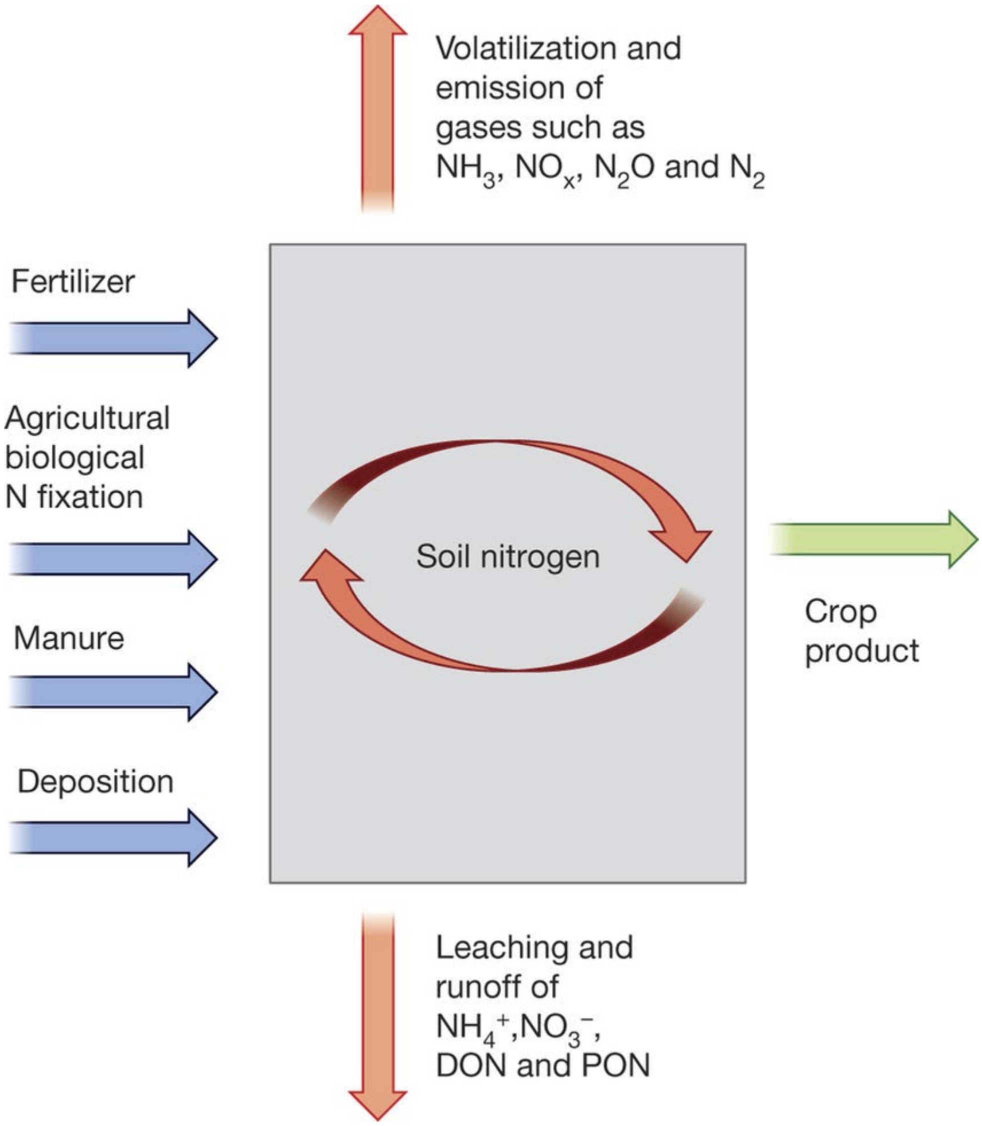


Figure 4.4.1. An illustration of the nitrogen (N) budget in crop production and resulting N species released to the environment. Source: Zhang et al. 2015.

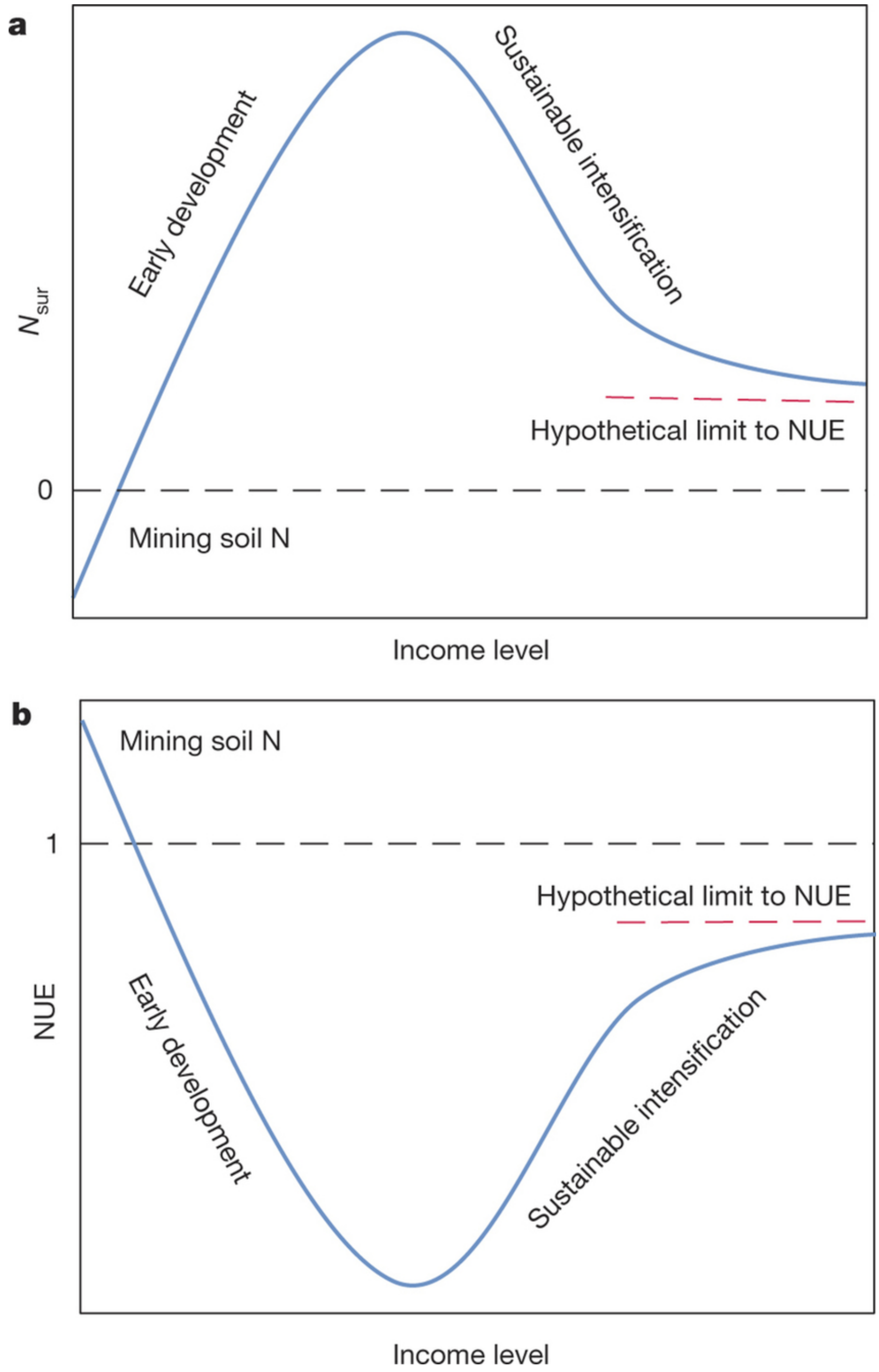


Figure 4.4.2. An idealised Environmental Kuznets Curve (EKC) for a) nitrogen surplus and b) nitrogen use efficiency (NUE). Source: Zhang et al. 2015. Copyright © 2015 Elsevier B.V. All rights reserved.

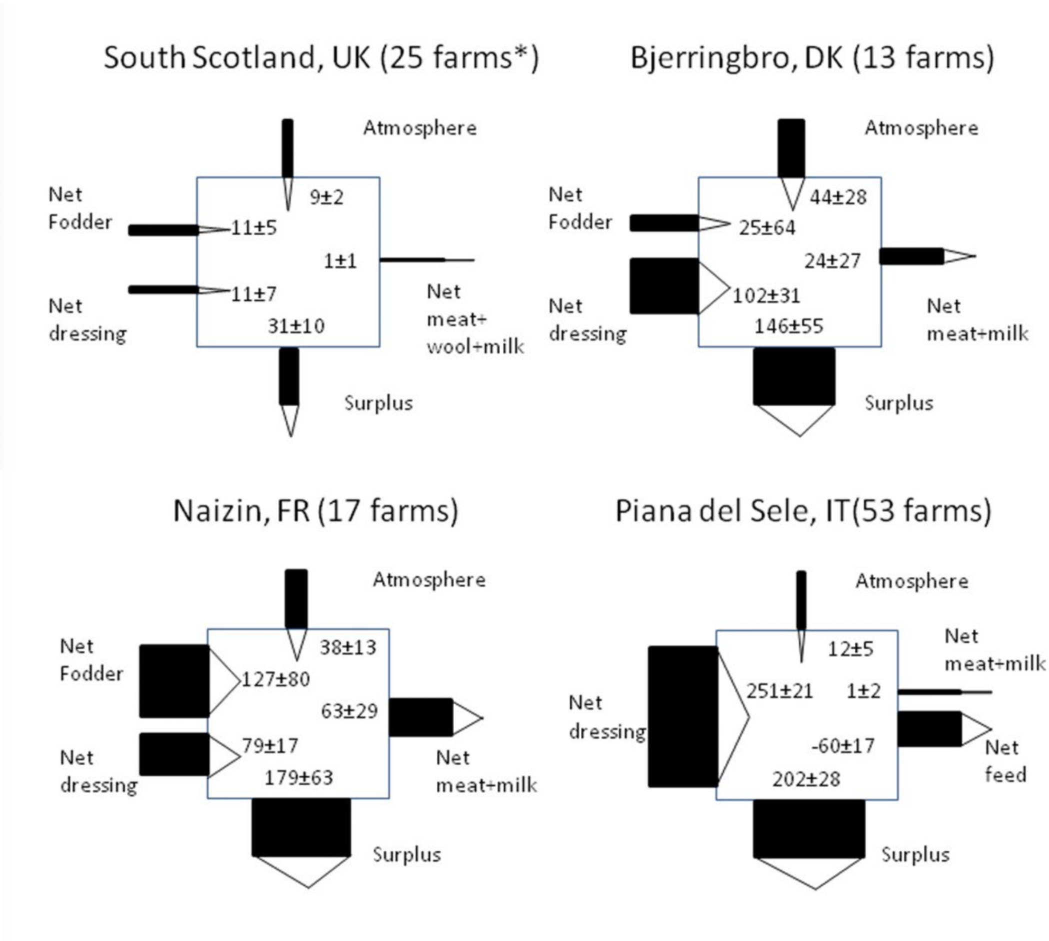


Figure 4.4.3. The intensity of agriculture, measured by nitrogen (N) input and output flows (kg N ha^{-1} yr^{-1}) for farming in four different European landscapes (redrawn from Dalgaard et al. 2012). The exemplified intensities range from low (left), to high (right) and include low livestock density, upland sheep grazing in Scotland (UK), mixed farming with a medium low- and medium high intensity in Poland (PL) and France (FR), respectively, and highly intensive farming, with specialised milking water buffaloes in Italy (IT). The N surpluses and the components of the N balance include N fixed or deposited from the atmosphere, net N fodder import (or feed export if negative), net import of dressing in the form of fertilizers or manure, and the net export of milk and meat, also including N in eggs and wool. All have 95 % confidence intervals, excluding landless poultry farming in Scotland (see text).

The intensity of agriculture can be characterised by the magnitudes of matter flows in- and out of the agricultural system, typically measured per land area or per product produced. In terms of N, this includes all the components of the N-balance, as exemplified in Figure 4.4.3, where the most extensive farming systems have the lowest N surplus while the most intensive systems have the highest N-surplus, defined as the difference between total annual N input and N output. Thereby, the N-surplus is an indicator for the intensity of the agricultural system.

In general, there is a positive correlation between the intensity of N flows through the farming system, the N-surplus and the losses of N from the agricultural system (Figure 4.4.3; 4.4.4). Thereby, extensive systems in general result in low N-surpluses and low N emissions, while intensive systems result in high emissions. However, there is a big variation between production systems and the natural conditions for production, and consequently there is a large potential to promote better use of N, and thereby a higher N efficiency, both for extensive and intensive systems, including improved soil-N balances, thereby preventing N losses to the environment.

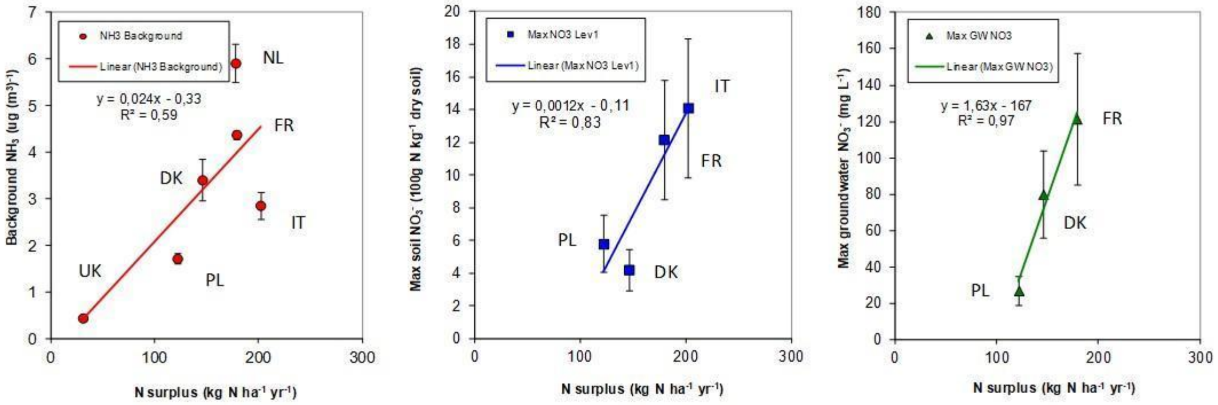


Figure 4.4.4. Comparison of nitrogen (N)-losses from agricultural systems with different intensity, indicated by average N surpluses from the Figure 4.4.3 farms studied in Scotland (UK), Poland (PL), France (FR) and Italy (IT), and additional data sets from The Netherlands (NL) and Denmark (DK) (Dalgaard et al. 2012). The results show measured ammonia concentrations (lowest site-mean, left) and soil nitrate levels (maximum nitrate concentration measured in the A-horizon of any site, centre) and groundwater nitrate levels (maximum nitrate concentration measured at any site, right).

An important point is the integration of extensive and intensive farming systems. Coupling of N-flows between systems can be a measure to improve N use (i.e., the law of diminishing returns from increased N input to plant and livestock production results in possibilities for optimization of production functions via a better distribution of N between systems). The potential for improved crop rotations, inclusion of N-fixing crops, and better distribution and use of livestock manures are often neglected at farm, landscape and regional levels. Therefore, when assessing N cycling in agricultural systems, it is not only a question of intensive versus extensive agriculture, but also about the level of integration between the types of intensity, and the potential effects from such integration (Dalgaard et al. 2012; Jin et al. 2020). In Figure 4.4.4, Poland and France represent examples on landscapes with mixed farming. In both cash crop farms and farms with cattle, pigs or poultry, there may be a high potential for a better distribution of N, and thereby a higher degree of integration. In contrast, the exemplified landscape of Piana del Sele in Italy includes very specialised and intensive water buffalo farms. With such high intensity it will not be able to distribute N optimally within the landscape. The options to reduce the N import and increase the N export, and/or reduce the intensity of farming would be a way to reduce N impacts and potentially increase N efficiency. Finally, note that extensive upland sheep grazing systems in the Scottish site, exemplifying extensive agriculture (Figure 4.4.4), also included a landless, highly intensive poultry facility. Such industrial production systems are common in many countries around the globe. Isolated, these extremely intensive production systems can be efficiently measured per product produced, but an assessment of the efficiency and sustainability in a larger picture is needed, including the natural effects of such very large point source emissions, and potentials for integration with other agricultural systems.

4.4.2 Crop quality with N use

There are two main negative consequences of N fertilizer use on crop production: excess N leading to decreases in crop quality and the acidification of soils leading to decreases in crop yield. Among the many plant fibers, the most studied fibre is cotton. Wang et al. (2012) demonstrated that excess N input may suppress carbohydrate/energy metabolism in early fiber development, which is very similar to the effect caused by N deficiency. Many crops show a narrow optimum range for N inputs, with excess leading to vegetative growth at the expense of the harvested product. A meta-analysis from Guo et al. (2010) describing N fertilizer reduced soil acidification showed that extensive N fertilizer application initially led to higher crop production but in the long-term threatens the sustainability of agriculture and cycling of nutrients and toxic chemicals in soils.

4.4.3 Tradeoffs between agricultural intensification and biodiversity protection

Agricultural intensification leading to higher yields has avoided greenhouse gas (GHG) emissions of up to 161 Gtonne of carbon (GtC) (590 $GtCO_2e$) between 1961 and 2005 (Burney et al. 2010). Future intensification should also result in avoided GHG emissions. In theory, agricultural intensification can protect native lands and ecosystems that are currently at risk from expansion of arable lands. However, some point out the trade-off between intensification and natural land preservation is by no means assured (Burney et al. 2010). For agricultural intensification to benefit forest and other land protections, intensification must be coupled with conservation and development efforts. As written, it sounds like agricultural intensification reduces GHG emissions, but it's well documented that adding N fertilizers leads to greater GHG emissions from soils such as nitrous oxide and sometimes methane.

A spillover effect of agricultural extensification is loss of native lands and biodiversity (Matson et al. 1997; Tilman et al. 2002). Land-use intensification itself is a major threat to the biodiversity of adjacent lands and waters, and is attributable to several factors, including especially inputs and unintentional losses of N fertilizer (Kleijn et al. 2009). There are unique and specific examples from Brazil, the Philippines and Costa Rica in which intensification has been successfully implemented to intentionally protect habitats and species (Phalan et al. 2011). In these examples an important mechanism for increasing crop yields along with conservation is increasing the access of farmers to experts who provide infrastructure and agronomic knowledge to increase nutrient use efficiency (Phalan et al. 2016).

While individual examples protect specific locations, a more coordinated response among countries could lead to greater overall protection of biodiversity through increased agricultural yields through intensification. One study ran scenarios of future intensification and compared locations of increased agricultural productivity with biodiversity records (Egli et al. 2018). They identified ten countries where the potential to reduce loss of biodiversity through agricultural intensification was highly significant (Table 4.4.1).

Table 4.4.1. Top 10 countries with the highest potential to reduce projected global biodiversity loss incurred by achieving projected production gains via agricultural intensification in the national scenario (Source: Egli et al. 2018).

Country	Projected production gain under the projections/national scenario (%)	Retained biodiversity value under the projections (%)	Retained biodiversity value under the national scenario (%)	Projected global biodiversity loss avoided under the national scenario (%)
India	32.6	71.9 (70.1–73.9)	84.4 (82.7–85.9)	5.8 (5.0–6.8)
China	13.5	91.3 (90.8–91.7)	98.5 (98.2–98.7)	4.7 (4.3–5.1)
Philippines	38.5	72.0 (69.3–74.9)	91.8 (89.7–93.5)	4.3 (3.2–5.5)
Brazil	13.8	94.2 (93.8–94.6)	98.3 (97.9–98.5)	4.0 (3.4–4.6)
Australia	22.5	85.7 (84.1–87.3)	98.7 (98.5–98.9)	3.2 (2.7–3.7)
Mexico	27.6	91.1 (90.5–91.7)	98.6 (98.3–98.9)	3.0 (2.5–3.5)
Indonesia	4.7	91.5 (90.0–93.3)	99.8 (99.7–99.8)	2.7 (2.1–3.3)
Congo, Democratic Republic of the	71.7	86.1 (84.5–87.7)	97.9 (97.5–98.2)	2.4 (1.9–2.8)
Ecuador	46.2	88.1 (86.4–89.8)	96.6 (95.1–97.9)	1.4 (0.9–1.8)
Vietnam	14.3	90.8 (89.5–91.9)	99.6 (99.4–99.7)	1.4 (1.1–1.7)

Agricultural intensification is likely to increase according to future projections (Foley et al. 2011; Egli et al. 2018). This is most likely to take place in regions of the world where the difference between current crop yields and potential crop yields is high, and these are often regions with high biodiversity. Addressing the trade-offs between agricultural intensification and biodiversity protection at all scales must consider real-world considerations of income generation, food and energy security, and where highest production gains can be achieved with the lowest potential costs to biodiversity (Egli et al. 2018). Yield gains alone do not necessarily preclude expansion of cropland into natural lands, indicating that intensification must be coupled with conservation and development efforts (Burney et al. 2010). Sustainable intensification is currently lagging relative to growing demand, resulting in a 15% expansion of the global area in staple crops from 2002 to 2014. Its continued advance is possible but depends on a well-prioritised and adequately funded research portfolio and appropriate policies and institutions to support it (Cassman & Grassini 2020). Further discussion is also needed in relation to alternative visions, including the concept of "sustainable extensification" (Garnett and Godfrey 2012; Sutton et al. 2024; van Grinsven et al. 2015).

4.5 Human Health

Nitrogen oxides, ammonia, various N particulates and other N compounds in the atmosphere cause a number of different impacts on human health. Quantification of those impacts is possible at different geographical scales, but quantification requires sufficient resolution to capture the correct levels of pollutant concentration. The assessment methods for air quality include air quality monitoring and modeling, manipulative experiments, and exposure/chamber experiments. The following N impacts are presented in Section 4.5. See the associated matrix for more information about these N impacts.

- Respiratory disease by aerosols (Section 4.5.1)
- Respiratory disease from NO_x (Section 4.5.1)
- Increased risk of cancer/cataracts (Section 4.5.1)
- Stratospheric ozone effects on human health (Section 4.5.2)
- Effects of water pollution on human health (Section 4.5.3)
- Dietary choice impact on human health (Section 4.5.4)

4.5.1 Air pollution and human health

A significant body of knowledge describes a strong relationship between air quality and human health on both short and long-term scales (Figure 4.5.1). According to the World Health Organization (WHO), urban air pollution exerts critical negative effects on public health. Premature deaths due to air pollution have increased from more than 2 million in 2006 to 6.5 million in 2012, and 91% of the human population lives in areas exceeding WHO guidelines for air pollution (WHO 2016; Figure 4.5.2). The consequences of air pollution on human health are not only confined to sickness and death, but also linked to lost work productivity and missed educational and other human developments.

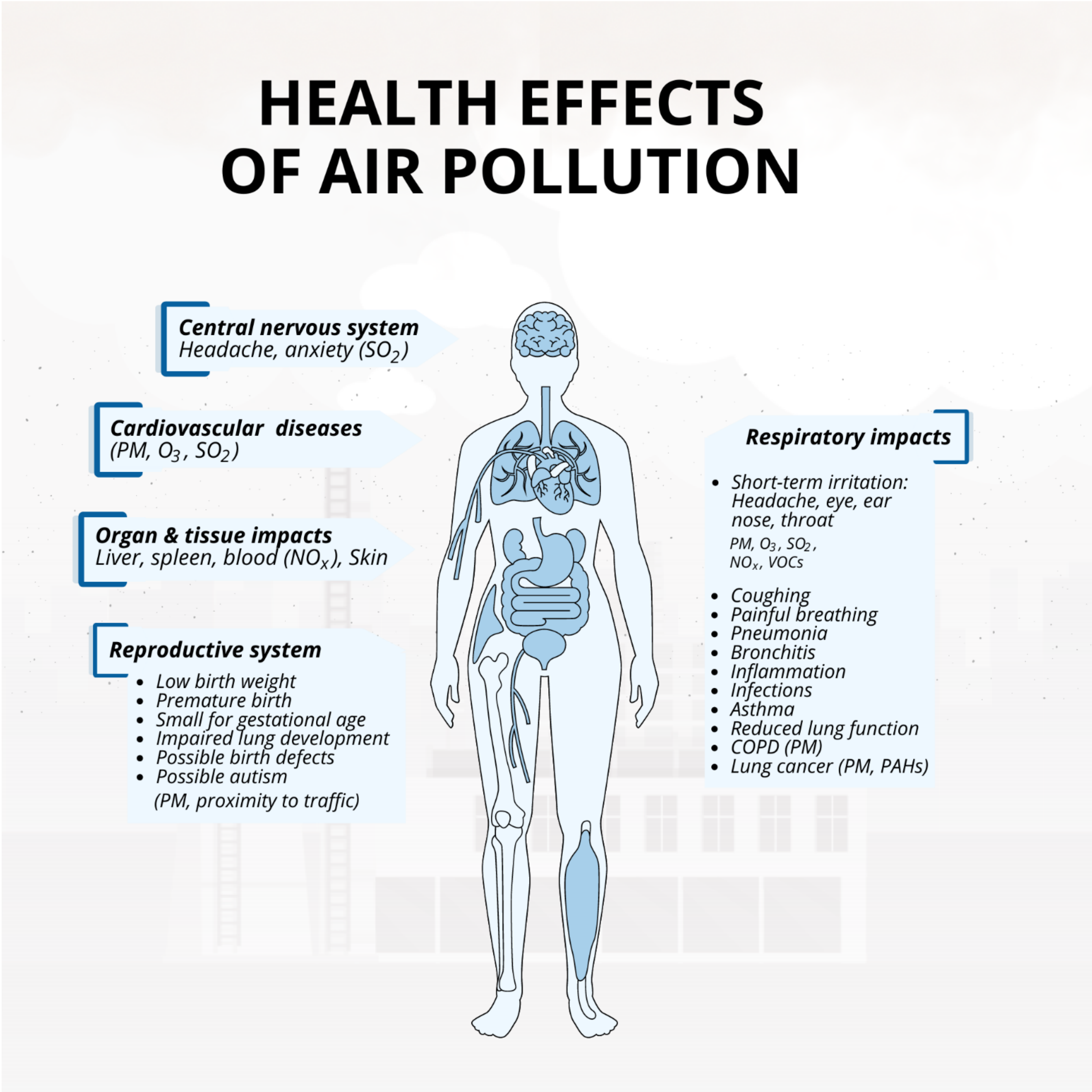


Figure 4.5.1. Summary of air pollution effects on human health. Source: Redrawn from Dabney 2013. Produced by Shel Evergreen. Particulate matter (PM), volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), Chronic Obstructive Pulmonary Disease (COPD). Original graphic produced for this document © UKCEH 2025.

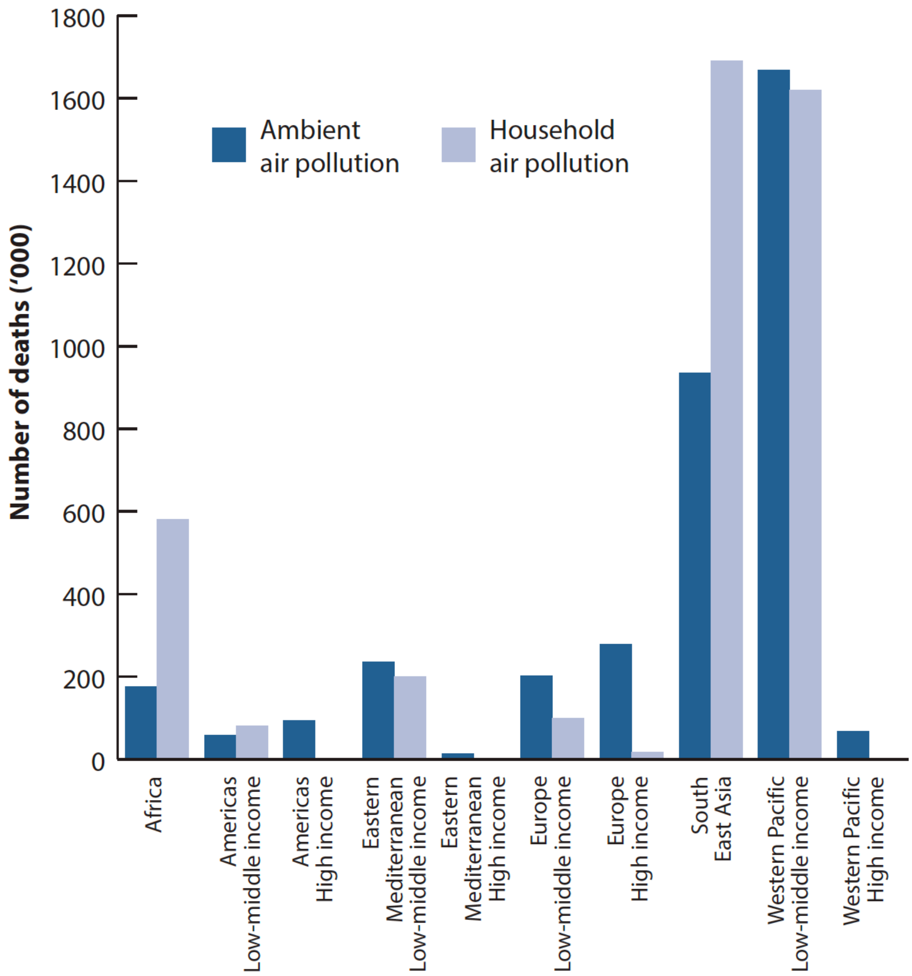


Figure 4.5.2. Total deaths attributable to the joint effects of indoor and outdoor air pollution in 2012, by region and economies (Source: Australian Government 2015).

Summary of the drivers, indicators, and mechanisms of impacts

The major drivers of environmental pressures on air quality are related to population and economic trends, transport, energy, agricultural demand and household consumption. These drivers are strongly connected to rapid global economic growth that has resulted in copious negative effects, including deterioration of air quality and associated impacts on public health (Figure 4.5.3). The fundamental link between global economic development is the rising demand for the production and delivery of goods and services, transportation and housing. Further, most of the energy used to meet this demand still comes from fossil fuels, which puts pressure on air quality and human health. Air pollution is intrinsically linked to various elements of Sustainable Development Goals (SDGs) relating to natural resources, food security, environment, and human health (Elder and Zusman 2016; Figure 4.5.3).

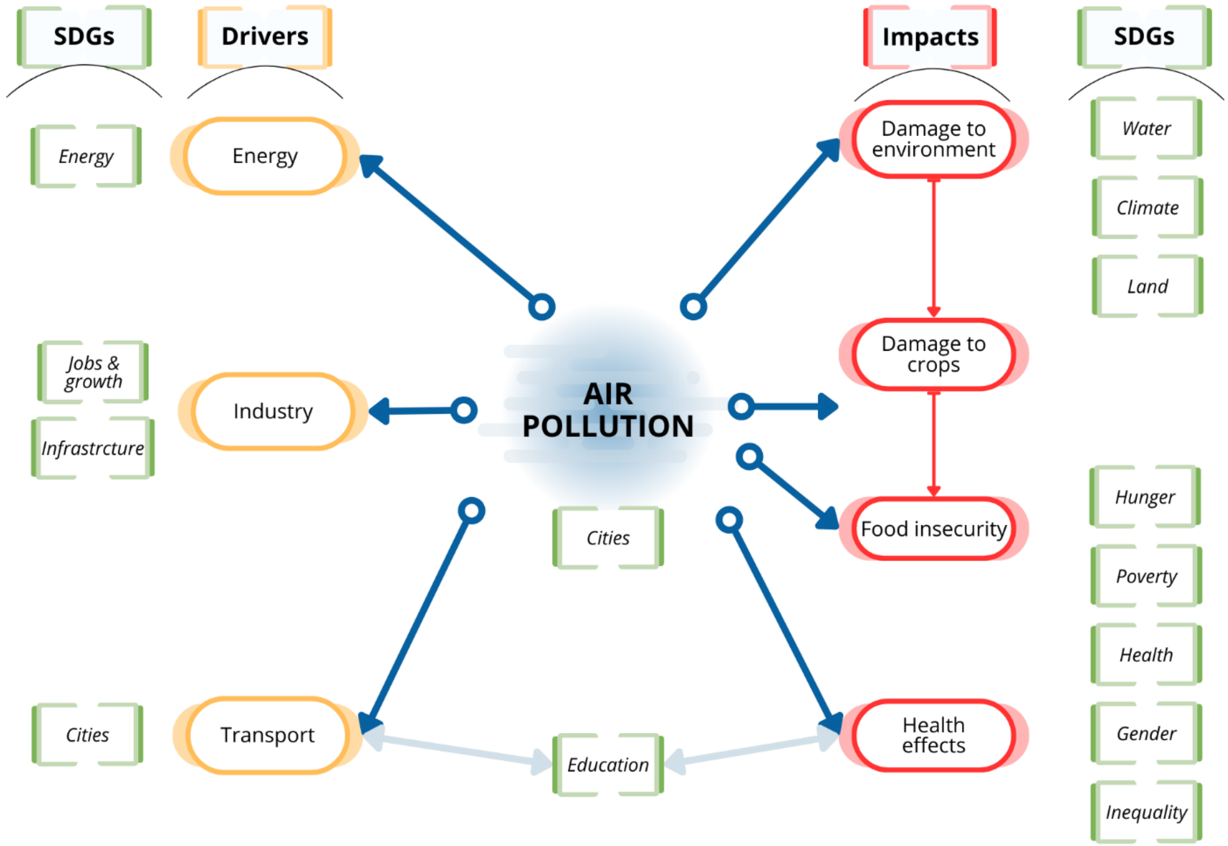


Figure 4.5.3. Summary of drivers and impacts of air pollution along associated sustainable development goals (SDGs) (Redrawn from: Elder and Zusman, 2016). Original graphic produced for this document © UKCEH 2025.

Outdoor air pollutant concentrations can be influenced by many factors including the quantity of air pollutants released by sources, distance from the sources, and meteorological conditions such as air temperature, the stability of the air, wind speed and direction. Depending upon the residence time of pollutants in the atmosphere, some air pollutants can be carried by the wind and affect air quality in locations at distances of hundreds to thousands of kilometres away from the sources.

Human exposure to air pollution is a major cause of death and disease on a global scale. For example, exposure to nitrogen oxides (NO_x) and sulphur oxides (SO_x) can irritate the lungs, reduce lung function, and increase susceptibility to allergies in people with asthma. Both NO_x and SO_x (as well as ammonium) are also precursors of fine particulate matter ($PM_{2.5}$) and contribute to the formation of smog and acid rain. Fine particulate matter and ground-level ozone (O_3) are the main components of smog and they have been associated with eye, nose and throat irritations, shortness of breath, exacerbation of respiratory conditions and allergies, chronic obstructive pulmonary disease and asthma, increased risk of cardiovascular disease and premature death (WHO 2016). Carbon monoxide (CO), sulphur dioxide (SO_2), nitrogen oxides (NO_x), volatile organic compounds (VOCs), ozone (O_3), heavy metals and respirable particulate matter ($PM_{2.5}$ and PM_{10}), differ in their chemical composition, reaction properties, emission, time of disintegration and ability to diffuse in long or short distances (Kampa and Castanas 2008). Air pollution has both acute and chronic effects on human health, affecting a number of different systems and organs. It ranges from minor upper respiratory irritation to chronic respiratory and heart disease, lung cancer, acute respiratory infections in children and chronic bronchitis in adults, aggravating pre-existing heart and lung disease, or asthmatic attacks. In addition, short- and long-term exposures have also been linked with premature mortality and reduced life expectancy. These adverse health effects of air pollution have created a public awareness of air pollution (Brunekreef & Holgate 2002; Bernstein et al. 2004).

In one study, authors investigated changes in patterns of air pollutants and drivers under rapid economic growth during the past three decades in heavily air-polluted areas of North China (Liu et al. 2018). They quantified the spatial patterns of air pollution and examined relationships of air pollution to several socioeconomic and climatic factors on decadal, annual and seasonal scales. Statistical analyses of this study indicate that energy consumption and gross domestic product (GDP) were the most important drivers for air pollution on the decadal scale, however, the effects of climatic factors were also significant. They showed that analyses on multiple temporal scales are necessary to determine key drivers of air pollution and to provide insightful understanding of the spatial patterns of air pollution.

Methods to assess the impacts

Scientific studies establishing possible effects of air pollution on human health have increased many-fold over the last few decades. However, the disparity in the effect estimated from different studies and recognizing important determinants of the disparity are challenging (Khafaie et al. 2016). Assessment of air pollution exposure has been a critical component in building a case for the relationship between air pollution and health effects in epidemiological studies since the early 1990s (Dockery et al. 1993; Zou et al. 2009). A blend of methods has been developed during the past 20 years for assessment of human exposure levels to air pollution, with the ultimate aim of estimating exposure at the individual level within an entire study population (Kousa et al. 2002; Ozkaynak et al. 2008; Ling et al. 2012). The quality of exposure measurement is a critical determinant in environmental epidemiological studies. However, the available exposure data and suitable methods for collection are often the main drivers behind the selection of measurement design. Despite significant developments in this field, epidemiological approaches to quantify the risks of exposure to air pollutants are still challenging (Table 4.5.1).

The assessment of personal exposure to air pollution is a critical component of epidemiological studies that link air pollution and human health. One review critically analysed 157 studies spanning over 29 years that used one of five categories of exposure methods listed in Table 4.5.1, such as proximity, air dispersion, hybrid, human inhalation and biomarkers (Zou et al. 2009). Proximity models were reported to be a questionable method because this technique assumes that closer proximity equates to greater exposure. Inhalation models and biomarker techniques were the most appropriate in measuring the personal exposure, however, these methods are generally not cost-effective for larger study populations. Their review recommends that:

i. Factors including data availability, validity, uncertainty, and transferability related to exposure assessment methods should be considered when selecting a model; and,

ii. Although an entirely discreet new class of method is not essential, significant progress could be made for the development of a 'hybrid' model utilizing the strengths of several existing methods.

Current knowledge emphasizes that future work should systematically evaluate the performance of hybrid models in comparison with other individual exposure assessment methods using geospatial information technologies (e.g., geographic information systems (GIS) and remote sensing (RS)) to obtain more robustly refined estimates of ambient exposure and find the linkages and differences between outdoor, indoor and personal exposure estimates.

Table 4.5.1. Summary and critical description of methods to assess air pollution effects on human health.

Method	Advantages	Uncertainties/limitations
Direct		
Personal Monitoring	Direct measurement of exposure during the monitoring period	Expensive and time consuming for studies having larger populations
Biological	Measure internal pollutant dose in human body High reliability to validate result generated from other exposure assessment methods	Differentiation between exposure pathway and chemicals is difficult Limited role of biomarker due to confounding factors High cost and time
Indirect		
Qualitative	Useful for larger population size Improved quality of epidemiological analysis	Designs can bias results and responses can be subjective Accuracy can bias design and subjective responses
Urban Monitoring Network	Useful for cross-sectional and cohort studies showing community exposure	Require appropriate volunteers for exposure Biased by availability of time-activity databases and other source of exposure such as indoor pollution
Proximity	Simple and appropriate for exposure research for unclear etiology of health outcome	Exposure misclassification and biased risk estimates
Interpolation (e.g. Kriging, Splines, Inverse Distance Weighting (IDW) and Theissen Triangulation)	Use of real pollution measurements in their computation of exposure estimates Generate estimates of the concentration of pollutant at sites other than the location of monitoring stations	Factors such as terrain or localised patterns are not considered Requires a reasonably dense network of sampling sites Proper application usually requires experience with geostatistical models

Land Use Regression (LUR)	Predict pollution concentrations at a given site based on surrounding land use and traffic characteristics Provide within city variability in pollution concentration	Independent variables and buffer radii for the pollutant require careful selection (e.g. wider for NO2 and narrower for estimates of diesel)
Dispersion	Use of data on emissions, meteorological conditions, and topography in estimating spatial exposure estimates of air pollution concentrations Provide more complete spatial and temporal variation of air pollutant concentration Provide high resolution analysis of patterns in health outcomes and environmental factors	Costly input data Need cross validation with monitoring data Provide environmental exposure concentration but not internal dose inhaled by individual
Hybrid	Personal or regional exposure plus one of models above Provide more accurate exposure Can use existing methods and do not have to struggle with new methods Measurement validation	Assessment cost and results are partly influenced by pollution under study (i.e. passive NO2 inexpensive vs. real time particle monitor expensive) and scale effects respectively
Human Inhalation Model	Produce potential dose inhaled by people at a local or even an individual scale	Restricted to areas with time–activity measurement databases High costs when population size is large
Biomarker	Measure internal dose of a pollutant in human body High reliability to validate results generated from other exposure assessment methods	Hard to differentiate between exposure pathways and chemicals The role of biomarkers is limited by confounding factors High cost and time consuming

Examples of the N impacts assessment with providing the Impacts-Pressures relationships

Excessive air- and water-borne N are linked to respiratory problems, cardiac diseases and several types of cancers. Ecological feedback to excess N can limit crop production, enhance allergies and potentially affect the dynamics of several vector-borne diseases, such as West Nile virus, malaria and cholera. Such examples show that our increasing production and use of fixed nitrogen poses a growing public health risk (Townsend et al. 2003). Health effects including clinical outcomes of increased morbidity and mortality, temporary or permanent loss of function and decreased health-related quality of life have been attributed to atmospheric N concentrations (Bascom et al. 1996; Table 4.5.2). Factors contributing to health effects may include a direct causal factor (e.g., NO_2), the exposure medium (e.g., air, water) and the susceptibility of a specific human population (e.g., those in developing countries, poverty) (Wolfe & Patz 2002).

Kim et al. (2013) critically reviewed the human health effects exposure to environmental (both outdoor and indoor) pollutants and discussed the dose-response relationships under a wide range of pollutant concentrations including ozone, NO_2, SO_2, VOCs and biophysical pollutants in relations to asthma-allergies. They reported various studies and found strong dose-response relationships for these pollutants. Kan et al. (2010) found a strong linear pressure-impact relationship between increases in PM_{10}, SO_2, NO_2 and O_3 and daily mortality rate in Shanghai, China. They also provided preliminary but inconclusive findings that women, older people and people with low education levels were more susceptible to air pollution effects than men. Qian et al. (2010) reported similar findings of stronger statistically significant correlations of mortality with NO, NO_2 and SO_2 than with PM_{10} in Wuhan China. Vichit-Vadakan et al. (2010) documented linear concentration-response functions for PM_{10}, O_3, NO, NO_2 and SO_2 pollution on mortality of various age groups in Bangkok. Their age-, disease- and age-restricted mortality analyses showed strong association with age. In another study, Peel et al. (2013) provided evidence of dose-response relationships of N-related ambient air pollutants and human health and mortality, and their study further added that climate change would aggravate the effects of reactive N on human health.

Hesterberg et al. (2009) established NO_2 no-effects levels from a critical review of the literature and reported that asthmatic persons were affected by NO_2 concentrations of up to about 0.6 ppm and, based on these values, health-protective and short-term-average NO_2 guidelines for vulnerable populations would reflect a policy choice between 0.2-0.6 ppm NO_2 concentrations. Garcia et al. (2019) used the G-computation model under hypothetical air pollution exposure scenarios targeting NO_2 and $PM_{2.5}$ in separate interventions to project childhood asthma incidence in southern California. Model results indicated that childhood asthma incidence rates would have been significantly higher if the observed reduction in ambient NO_2 in southern California had not occurred in the 1990s and early 2000s, and asthma incidence rates would have been significantly lower if NO_2 concentrations had been lower than what it was estimated to be.

Table 4.5.2. Health effects associated with air pollutants above ambient concentrations and their dose-concentration relationships. Adapted from Botkin & Keller (2007), Kim et al. (2013) and Kadam (2018). Parts per million (PPM), volatile organic compounds (VOCs).

Pollutant	Exposure time	Concentrations (ppm)	Health effects
Ozone	Short- to long-term	>5	Irritation of eyes and respiratory system
		>50	Death due to pulmonary edema
Particulate matter ($PM_{2.5}$, PM_{10})	Short- to long-term	>100	Respiratory hazard, cardiovascular alterations
Carbon monoxide	Short- to long-term	10	Impairment of judgment and vision
		100	Headache and dizziness
		250	Loss of consciousness
		>700	Death
Carbon dioxide	Short- to long-term	2000-5000	Headache, insomnia, nausea
		>5000	Death
Nitrogen oxides (NO_x)	Short- to long-term	50-100	Inflammation of lung tissue
		150-200	Bronchitis fibrosa
		>500	Death
Sulphur dioxide (SO_2)	Short- to long-term	>5	Tracheal irritation and cough
		5-10	Bronchitis fibrosa
		>200	Death
Formaldehyde	Short-term	<100	Skin irrigation, respiratory tract

Pollutant	Exposure time	Concentrations (ppm)	Health effects
		>100	Death

4.5.2 Stratospheric ozone depletion and human health

This section describes the role of nitrous oxide (N_2O) in the depletion of the stratospheric ozone layer. The layer of ozone (O_3) in the stratosphere surrounding Earth reduces harmful incoming solar ultraviolet radiation (UV) from reaching the surface, where UV-B radiation causes skin cancers, cataracts and ecosystem damage (Fang et al. 2019).

Discovery that the ozone layer was being depleted due to chlorofluorocarbons and other similar compounds led to the Montreal Protocol, signed in 1987. The Montreal Protocol is a global agreement to protect the stratospheric ozone layer by phasing out the production and consumption of ozone-depleting substances (ODS). The total global emissions of ODSs have been declining since the late 1980s, the observed atmospheric abundances of most ODSs are declining after peaking in the 1990s and 2000s and stratospheric ozone recovery is underway (Fang et al. 2019). Nitrous oxide now provides the largest contribution among all individual ODS emissions to stratospheric ozone loss (Ravishankara et al. 2009; UNEP 2013). Model forecasts estimate the recovery of the ozone layer to pre-1980 levels will be delayed by continued inputs of N_2O.

Depletion of the ozone layer results in greater potential exposure to UV-A and UV-B radiation, but actual exposure also depends on human behaviour, such as time spent outdoors, use of shade and wearing of sun protective clothing (Lucas et al. 2014). Although the incidence of melanoma continues to increase in many countries, in some locations, primarily those with strong sun protection programs, incidence has stabilised or decreased over the past five years, particularly in younger age groups (Lucas et al. 2014). The level of UV radiation is linked to the worldwide increase in cortical cataracts, with cataracts being the leading cause of blindness worldwide in 2010 (Bourne et al. 2014).

Methods of Assessing Impacts

There is an abundance of literature related to the relations between UV radiation exposure and melanoma skin cancers and cataracts (WHO 1994; Lucas et al. 2008). Methods used to determine the link between UV radiation include laboratory studies where tissues and specific proteins are exposed to levels of radiation and their responses recorded (Linetsky et al. 2014). Epidemiological studies are commonly conducted to quantify the associations between UV exposures and skin cancer risk (Savoye et al. 2018). In these types of studies, as large a cohort of people as possible — those who had been exposed to skin cancer and those who had not — are selected to respond to a questionnaire regarding lifetime UV exposure in nested case-control studies. Statistical analyses are performed to control for as many confounding factors as possible. Models are also used to develop population-level exposure-disease relationships (Lucas et al. 2008).

4.5.3 Water pollution and human health

Nitrate does not directly cause human health impacts. However, when consumed, about 5% of nitrate is converted to nitrite, which is more toxic (Santamaria 2006). Nitrate consumed in drinking water is linked to the blood disease methemoglobinemia, or "blue baby syndrome", and digestive-tract cancers, resulting in widespread statutory limits on nitrate in ground and drinking water (Grizzetti et al. 2011, but see discussion of controversy therein). The World Health Organization (WHO) recommends maximum standards for nitrate, citing reduction to nitrite in the bodies of infants as a cause of blue baby syndrome, and the further transformation of nitrite to carcinogenic N-nitroso compounds in mammalian bodies as a cause of cancer (WHO 2011).

Some have disputed the quality of evidence linking nitrate to disease (Bryan & van Grinsven 2013), suggesting that nitrate drinking water standards in the United States and Europe are overly cautious and should be raised (Addiscott & Benjamin 2004; Powelson et al. 2008). Currently, ground and drinking waters exceed regulatory standards around the world: In parts of Africa (Tredoux & Talma 2006), China (Gu et al. 2013), Europe (Bryan & van Grinsven 2013), India (Prakasa Rao et al. 2017) and USA (Pennino et al. 2017).

Methods of Assessing Impacts

The United States Environmental Protection Agency (EPA) produces N threshold guidelines for water impairment by waterbody type and ecoregion, which states then use to set TMDLs based on the usage of the waterbody (e.g., "swimmable", "fishable", "drinkable"); monitoring of N concentrations is carried out by EPA, the U.S. Geological Survey, and the National Oceanic and Atmospheric Administration, as well as state and local authorities (EPA 2011). In the EU, critical levels for N in drinking water emerge from a web of regulations, with member nations submitting individual administrative plans to the European Commission for approval (European Commission 2007). Citizens in these nations generally experience low probability of exposure to nitrate above standards in drinking water (Pennino et al. 2017; Grizzetti et al. 2011).

United Nations Sustainable Development Goals call for universal access to clean water by 2030, but data limitations inhibit verification in developing countries (Akale et al. 2018). The World Health Organisation and the United Nations Children's Fund produced a Rapid Assessment of Drinking Water Quality (RADWQ) protocol in 2010, and released microbial, chemical and physical water quality data for five pilot countries (WHO 2015). Researchers analyzing the bacterial data found that RADWQ likely overestimates the number of citizens in a nation with access to clean water, by assuming that piped or otherwise improved water sources are always safe (Onda et al. 2012). A study incorporating nitrate and other chemical indicators of water quality reached similar conclusions, demonstrating the presence of high nitrate (and other water quality indicators) in improved drinking water sources in all five developing nations (Bain et al. 2012).

In China, recent work has focused on developing methods of assessing nitrate presence and threat in rapidly developing agricultural and industrial regions. Research often takes a case-study approach, with work aimed at identifying sources of nitrate in drinking water (Zhang et al. 2015), or modeling areas of potentially serious contamination (He et al. 2018). Some have tested methods of reducing nitrate losses from agricultural systems (Huang et al. 2018). Much recent work takes the next step of connecting nitrate exposure to health risks in a given region (eg. Su et al. 2013; Chen et al. 2017; Zhai et al. 2017; Li et al. 2019).

4.5.4 Dietary choices and human health

Nitrogen is an essential nutrient for agricultural products and foods. However, over-consumption of food (especially, calories and saturated fats) and protein also leads to negative human health outcomes.

Diet and dietary preferences are critical determinants of human health, and over- or under-consumption of nitrate can lead to human health impacts. Humans require proteins to meet a variety of demands for amino acids (Figure 4.5.4). Although humans are essentially omnivorous, people have a variety of food preferences that are strongly affected by society, economy, tradition, culture, lifestyle (both personal and social aspects) and economic conditions. Gender and age also affect food preference.

Diet-related human health impacts include malnutrition when protein intake is insufficient, and potential for cardiovascular diseases when too much protein is consumed. Malnutrition is a problem in areas with high poverty rates, whereas overconsumption is common in economically developed countries. Possible effects of excess protein intake on human health are complicated because of additional interacting co-morbidity factors, including diabetes, obesity, cardio-vascular disease and gastro esophageal diseases (Table 4.5.3; WHO 2007; Willett et al. 2019). Consumption of red meat (beef, pork or lamb, especially their processed meat) is among several factors associated with increased risk of death from cardiovascular disease, stroke and type 2 diabetes (Willett et al. 2019). A comprehensive analysis of dietary risks in 195 countries from 1990 to 2017 showed that high consumption of red meat and processed meat were towards the bottom in ranking of dietary risks of non-communicable diseases (GBD 2019).

National guidelines can differ from the WHO guidelines (Figure 4.5.5 shows an example from Japan), and there are sources of uncertainty in studies that attempt to determine ideal protein intake amounts. Key sources of uncertainty include large variation in protein requirements among and within studies, the influence of energy balance on protein requirements, as well as the short duration (<2 weeks) of most N balance studies included in the meta-analysis (WHO 2007). The healthy reference diet can be a suitable world baseline when one focuses on relationships between different protein sources and long-term health impacts (Willett et al. 2019).

Special reports for the European Nitrogen Assessment process have shown how reducing meat and dairy intake in Europe would offer health benefits while substantially reducing nitrogen pollution threats to the environment (Westhoek et al. 2014). The most socially appealing strategies to achieve a target of halving nitrogen waste from the agri-food system were found to be those that combine a wide range of measures, including dietary change, food waste reduction and technical measures in agriculture and wastewater management (Leip et al. 2023).

Methods by which effects on human health are measured

Underconsumption or overconsumption of protein compared to protein intake recommendations is a health risk, but the relationship between protein intake and health status is difficult to quantify. Relationships between protein consumption and health outcomes can be obtained from prospective cohort studies (Willet et al. 2019). A cohort study is a long-term observational study to examine effects of target factor(s) on a group of people. Because the hierarchy of observational studies is not sufficiently high, meta-analysis based on relevant cohort studies is conducted to earn sufficiently high evidence of the cause–effect relationships.

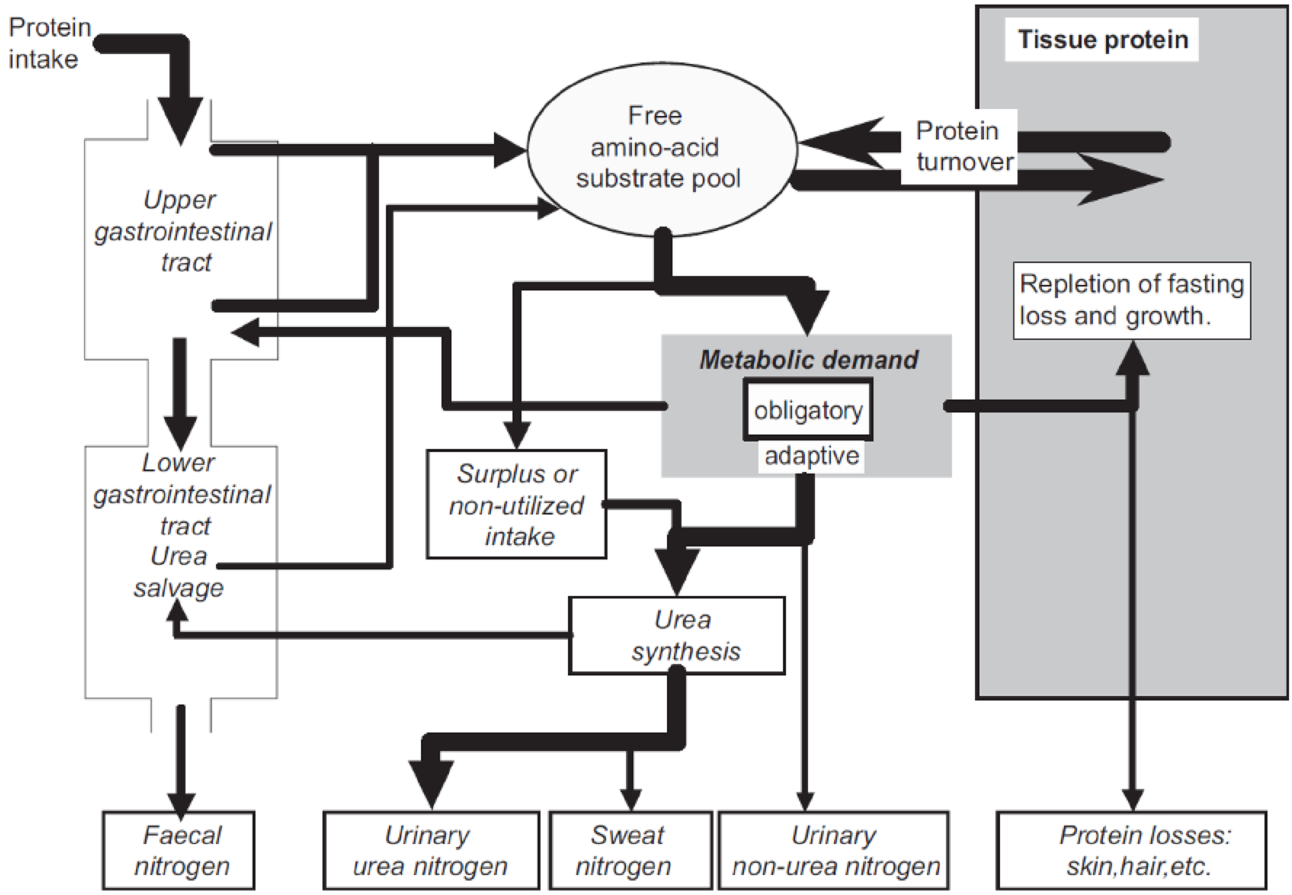


Figure 4.5.4. Schematic representation of the metabolic demands for amino acids (WHO 2007). Humans require protein with a balanced amino acids content. Public report © World Health Organization 2007

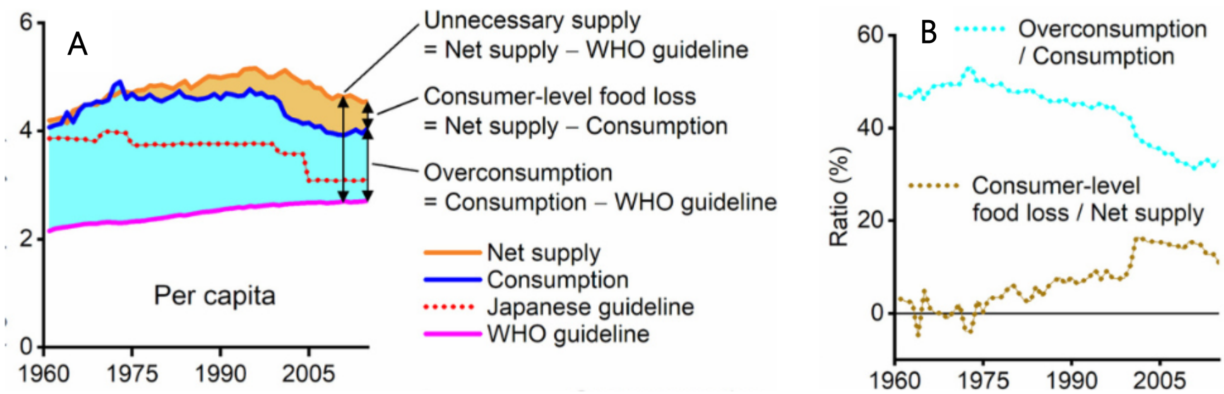


Figure. 4.5.5. A: Per capita food N flow in Japan 1961-2010 (Hayashi et al. 2018). Japanese guidelines for recommended protein intake were nearly double that of World Health Organization guidelines in 1961. **B:** Ratio of supply of edible parts of food-to-food waste (brown line); ratio of overconsumption to national guideline (turquoise line); and recommended protein intake provided by the World Health Organization WHO (black line). © 2018 The Author(s). Published by IOP Publishing Ltd.

Table 4.5.3. Negative effects of excess protein intake on human health (Source: WHO 2007; Willett et al. 2019).

Health status	Possible negative effects
Body growth	Protein malnutrition suppresses body growth especially for infants and children
Bone health	Bone mineral density can be reduced when acid-base chemistry is disrupted by protein metabolism.
Renal function	Excess proteins contribute to kidney failure
Kidney stones	High animal protein intake may increase risk of kidney stones
Cardiovascular disease	Diets with a higher proportion of protein can be beneficial for the heart, protective influence on raised blood pressure Replacing calories from carbohydrates with protein reduces blood pressure and lipid concentrations Consumption of processed red meat (beef, pork, or lamb) is associated with increased risk of death from cardiovascular disease; unprocessed red meat is also weakly associated with cardiovascular disease Red meat consumption contributes to increased risk of coronary heart disease. Consumption of poultry, fish, nuts, and legumes reduce risks of cardiovascular disease Consumption of one egg per day is not associated with increased risk of cardiovascular disease except in people with diabetes Consumption of nuts is associated with reduced risk of cardiovascular disease
Stroke	Consumption of red meat is associated with increased risk of stroke
Type 2 diabetes	Consumption of red meat is associated with increased risk of type 2 diabetes Yogurt might reduce risk of diabetes Consumption of nuts is associated with reduced risk of type 2 diabetes
Cancer	Little effect of total protein intake on the incidence of cancer, but red or processed meat might increase the risk relative to plant protein Processed red meat is a group 1 carcinogen and unprocessed red meat is a group 2 carcinogen associated with colorectal cancer High intake of protein is associated with increased risk of breast cancer

Health status	Possible negative effects
	High milk consumption is associated with reduced risk of colorectal cancer, but increased risk of prostate cancer in men Soy foods reduce risk of breast cancer and other hormonally-related cancers
Mortality	Consumption of red meat is associated with increased risk of total mortality Replacing animal protein with plant protein substantially reduces overall mortality Consumption of nuts is associated with reduced risk of overall mortality

4.6 Cultural Impacts

4.6.1. Aerosols and visibility

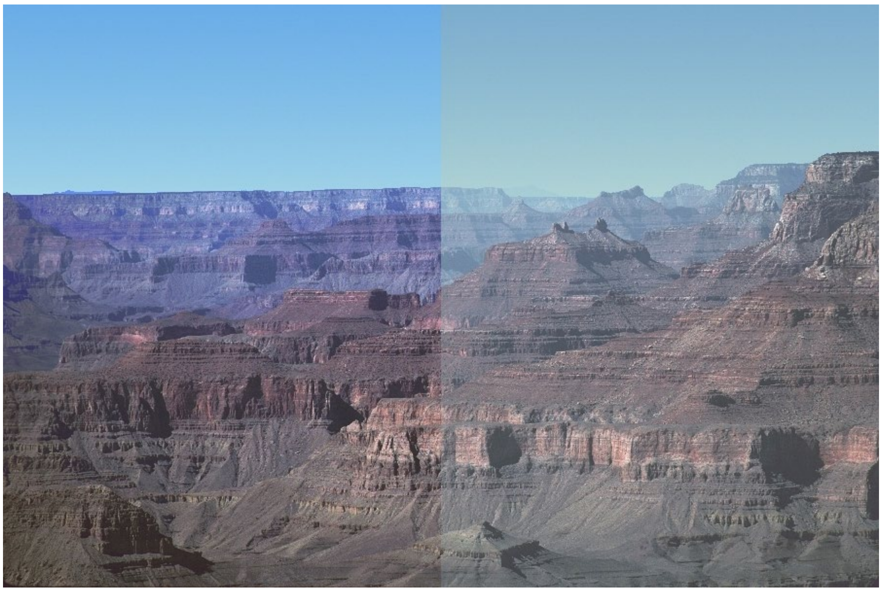
The presence of N compounds in the atmosphere influences cultural values by reducing visibility (Figure 1.4). Visibility refers to the visual quality of the view, the ability to perceive the illuminated forms, colours and textures of features in the landscape (Malm et al. 1980; Watson 2002; Hyslop 2009; Malm 2016). On very clear, high-visibility days, near objects have bright, crisp colours and textures, while objects over 200 km away are still visible (see Box 4.1). Even when there are no distant objects, a clear day produces vibrant blue skies that can contain bright white clouds with sharp edges. Air pollution can impair visibility by the loss of image-forming light and the addition of non-image-forming light between an observer and the objects being viewed. These changes to the light reaching the observer are the result of light extinction, the scattering and absorption of light by particles and gasses (Van de Hulst 1981).

Human caused visibility impairment is primarily due to fine particulate matter with diameters <2.5 μm ($PM_{2.5}$) (Malm et al. 1994; Hand & Malm 2007; Pitchford et al. 2007; Lowenthal et al. 2015). $PM_{2.5}$ is a complex mix of chemical species, including ammonium nitrate, ammonium sulfate and organic material all with differing optical properties.

Reactive N is a significant contributor to particulate matter and haze around the world (Wang et al. 2013; Bian et al. 2017). Like in the United States, particulate nitrate is expected to become more important globally in the future due to anticipated increases in emissions of ammonia and nitrate precursors and declines in ammonium sulfate, linked to reductions of SO_2 emissions (Bauer et al. 2007). The connection between nitrogen species and visibility impairment is well-studied and well-documented. Our confidence in the driver-impact relationship is high.

Box 4.1. Impact of air pollution on scenic view.

Scenic views with long sight paths and clear air that exist in many remote areas around the world are sensitive to small changes in loadings of fine particulate matter ($PM_{2.5}$), diminishing the view (EPA 1979; Latimer et al. 1981; Pitchford & Malm 1994). For example, the addition of 5 µg m^{-3} of a typical mix of $PM_{2.5}$ to clear air in the Grand Canyon National Park (below) creates a visible haze obscuring the scene. Good visibility is also important in more urban environments where visibility has been linked to human mental health (Jones & Bogat 1978; Evans et al. 1982; Campbell 1983; Zeidner & Shechter 1988; Mace et al. 2004; Velarde et al. 2007) and is important for safe aircraft, marine and automotive traffic operations (Haeffelin et al. 2010). The preference of people for clearer air is well-documented in the United States, Canada, and China (Malm et al. 2019; Pryor 1996; Fajardo et al. 2013).



Above: A scenic view from the south rim of Grand Canyon National Park with low particulate pollution (left) and modelled haze (Molenar et al. 1994) due to 5 µg m^{-3} of fine particulate matter (right) at 75% RH. From Colorado State University IMPROVE monitoring data. Download: http://vista.cira.colostate.edu/Improve/winhaze/.

Processes by which N impacts visibility

Reactive nitrogen plays an important role in the formation of particulate matter and the resulting haze causing visibility degradation. Particulate ammonium and nitrate directly contribute to particulate mass, increased particle size and light scattering (Hand et al. 2012a). Ammonium nitrate and ammonium sulphate particles are hygroscopic, which means that they absorb water at higher relative humidity (RH), increasing particle size and reducing visibility. Light scattering by ammonium nitrate particles is about four times greater at RH = 90% compared to dry conditions (Malm et al. 1994; EPA 2003; Lowenthal et al. 2015). Such particles containing water are often referred to as 'aerosols'.

Ammonia gas also indirectly affects $PM_{2.5}$ and haze levels. Increased ammonia can increase particulate ammonium nitrate concentrations that are in equilibrium with gas phase nitric acid (Stelson & Seinfeld, 1982). Ammonia also plays an important role in governing the acidity of particulate sulfate (Weber et al. 2015; Dennis et al. 2008), which affects its hygroscopicity, where a more neutralised sulfate is generally less hygroscopic (Gebhart et al. 1994; Lowenthal et al. 2015). This in turn influences the particulate water content and its contributions to haze and formation of secondary organic aerosols from biogenic emissions of volatile organic carbon compounds (Carleton et al. 2010; Carlton & Turpin 2013; Nguyen et al. 2015; Budisulistiorini et al. 2017; Malm et al. 2017).

Measurements for visibility

Particulate matter monitoring networks measure speciated $PM_{2.5}$ and PM_{10} mass, as well as other aerosols, with well-developed methods. The Interagency Monitoring of Protected Visual Environments (IMPROVE) monitoring programme serves the United States (Malm et al. 1994; Hand et al. 2012a; Pitchford et al. 2007). The European Commission, acting through the European Committee for Standardisation (CEN) has produced a series of Standard Methods for monitoring air pollutants (https://uk-air.defra.gov.uk/networks/monitoring-methods?view=eu-standards).

Although particulate ammonium is an important contributor to $PM_{2.5}$ and haze, it's measurement in current monitoring networks is problematic, since ammonium nitrate is semi-volatile and lost from the filters used to collect $PM_{2.5}$ samples for compositional analysis (Yu et al. 2006). This concern can be addressed by using denuder-filter combinations to separate the gases and aerosols (e.g., Perrino and Gherardi 1999; Tang et al. 2018, 2021). In situations where such approaches are not used, ammonium is sometimes inferred from the assumption that all measured $PM_{2.5}$ nitrate and sulfate are in the form of ammonium nitrate and ammonium sulfate.

The importance of reactive nitrogen species to human-caused visibility degradation across the United States is illustrated in Figure 4.6.1. On days with the highest human-caused haze, as measured in the IMPROVE monitoring network, ammonium nitrate and ammonium sulfate account for more than half of the light extinction coefficient (b_{ext}) of the particle throughout the eastern United States and Great Plains, as well as at many locations along the west coast. At some of these sites, ammonium nitrate is the largest contributor to b_{ext}, particularly near intensive agricultural regions in the Great Plains, Midwest, and California. This is a marked change from the early 2000's, when ammonium sulfate was the largest contributor to haze in these regions (Malm et al. 2001). However, with the substantial reductions in SO_2 emissions (Hand et al. 2012b), partly driven by effective air quality regulations (Xing et al. 2013), particulate sulfate concentrations have decreased at a faster rate than ammonium nitrate and today are more than a factor of 3 lower than in the early 2000's (Hand et al. 2012b). There has also been a corresponding widespread reduction in haze (Hand et al. 2014). These emission trends are expected to continue (Li et al. 2016).

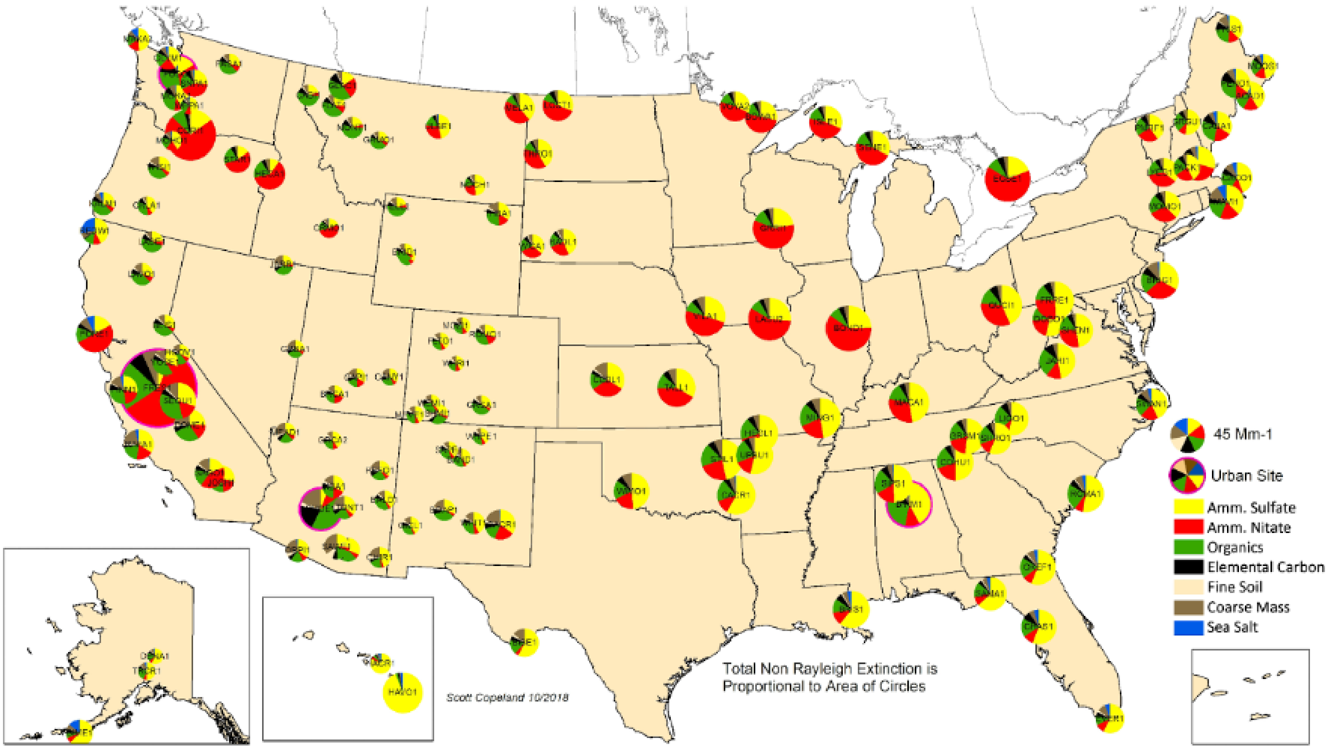


Figure 4.6.1. The light extinction coefficient (b_{ext}) from atmospheric particulate matter (larger circles indicate larger b_{ext}, i.e. lower visibility), showing the relative contributions from major aerosol species listed by colour. The figure shows the results for days in 2017 with the 20% highest human caused visibility impairment, as derived from

Colorado State University IMPROVE monitoring data. Download: http://vista.cira.colostate.edu/Improve/rhr-summary-data/.

4.6.2 Materials and cultural heritage

The effects of air pollution on materials and cultural heritage (Watt et al. 2009) is a well-studied subject within the field of atmospheric corrosion (Leygraf et al. 2016). Because of this, our confidence in the effects of N on materials and cultural heritage sites is high. In Figure 1.4 this impact is one that is relevant for cultural services. A wide range of materials are affected including different metals, stone materials, concrete, mortars, brick work, glass materials and wood. Each material has its unique sensitivity to pollutants, which in the outdoor environment include mainly sulphur dioxide, N oxides and nitric acid, ozone and particulate matter. The direct effect of air pollution on materials is, in general, detrimental, but can range from small or negligible to large and significant, depending on the material-pollution combination. There is no threshold level for pollution below which there is no effect. Due to pollution, the lifetime of technological products is shortened. Buildings and other structures, as well as objects of cultural heritage, deteriorate more rapidly when they are exposed to the atmosphere.

Processes by which N air pollution influences materials

The cause of the impact is chemical and electrochemical reactions that take place on the surface of the material. This occurs when the surface is sufficiently wet, for example due to high humidity, dew or after rain. All atmospheric particles, including those containing N, can contribute directly to the increased wetness depending on the hygroscopicity of the respective salt. The pollutants dissolve in water on the material surface causing an aggressive electrolyte, often acidic, resulting in a chemical transformation of the material into corrosion products that can either stay on the surface or be washed away by rain.

Historically, sulphur dioxide was the dominant air pollutant in Europe. However, since the turn of the 21st Century, due to successful implementation of air quality policy, sulphur dioxide has been greatly reduced. However, research has identified other pollutants including ozone, nitrogen oxides, nitric acid and particulate matter. The most striking example is the high corrosivity of nitric acid (HNO_3). Compared with SO_2, zinc is almost twenty times more sensitive to HNO_3 (see Figure 4.6.2; Samie, 2006).

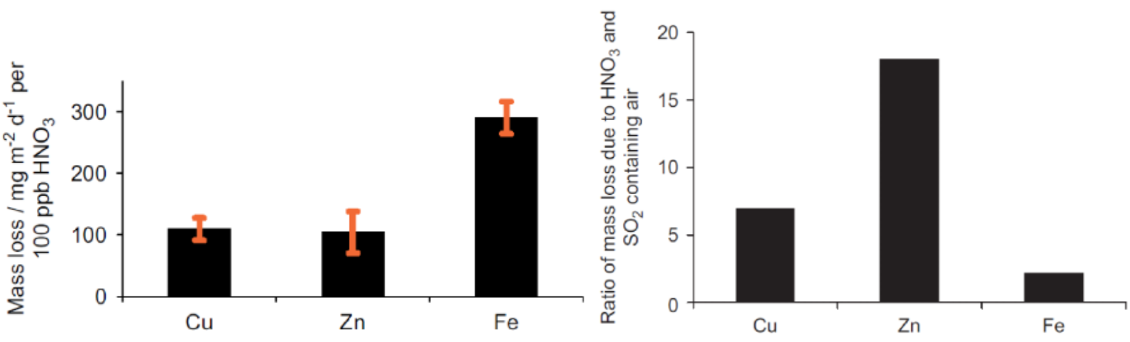


Figure 4.6.2. Effect of nitric acid (HNO_3) on copper (Cu), zinc (Zn) and carbon steel (Fe); Absolute corrosion measure as mass loss (left) and sensitivity relative to that of SO_2 at equal concentrations of HNO_3 and SO_2 (right). Error bars show the standard deviation of at least three samples. Original graphic produced for this document © UKCEH 2025.

Methods

Effects on different materials in the outdoor environment under field conditions are typically summarised in so-called dose-response functions. These enable the calculation of corrosion attack as a function of air pollution, climate and other factors. The basis for the functions is international exposure programs where standardised materials are exposed on racks with simultaneous extensive environmental characterisation (see Box 4.2).

The Pressure-Impacts (i.e., dose-response) functions are derived from a network of test sites within the framework of the Convention on Long-Range Transboundary Air Pollution (CLRTAP). These are based on data from about thirty test sites and twenty countries signatories to the convention. The functions have been used for mapping areas of increased risk of corrosion under different scenarios using methodologies described in the UNECE mapping manual (CLRTAP 2014). In order to use the functions, it is necessary to have access to pollution and climate data relevant for urban areas; gridded data at low resolution are not sufficient. For example, when using modelled output from the EMEP 50 km x 50 km there was a significant difference between modelled and measured pollution (and corrosion), while a recent study focusing on UNESCO sites using modelled output from the recent EMEP 0.1° x 0.1° showed acceptable agreement (Spezzano et al. 2018).

Box 4.2. Monitoring the effect of air pollution on materials

Above: Example of a test site with racks supporting standardised materials. So far, dose-response functions, including those for nitric acid, have been published for zinc and limestone, important materials for the industry and in cultural heritage. For limestone, the function is as follows:

$$R = 4.0 + 0.0059\,[SO_2]\,RH_{60} + 0.054\,\text{Rain}\,[H^+] + 0.078\,[HNO3]\,RH_{60} + 0.0258\,[PM_{10}]$$

where R is the corrosion attack in µm after one year of exposure, $[SO_2]$, $[HNO_3]$ and $[PM_{10}]$ are the concentrations in µg m^{-3}, RH_{60} is equal to the relative humidity above 60% (0 if below 60%), Rain is the amount of precipitation in mm and $[H^+]$ is the acidity of precipitation in mg l^{-1}. The equation illustrates the multi-pollutant nature of air pollution effects on materials, the necessity to have water present in enough quantity for corrosion to occur (relative humidity above 60%, rain and hygroscopicity of particles) and the importance of both dry deposition (pollutants and particles) and wet deposition (acid rain). See details in Doytchinov et al. (2013)

4.6.3 Recreation and algal blooms

Nitrogen that contributes to eutrophication of coastal and freshwaters can lead to increases in algal productivity called algal blooms. The algae can include green algae and cyanobacteria. Excessive algal blooms can promote hypoxia (depletion of oxygen in water) and anoxia (absence of oxygen). This occurs when the death of algae leads to microbial decomposition, which consumes oxygen dissolved in the water body. Cyanobacteria can multiply to produce harmful algal blooms (HABs) that can adversely affect human health, both physically, by causing skin, gastrointestinal or neurological disorders (see Section 4.5.3). Harmful algal blooms can also affect human well-being through unpleasant odours of decomposing biomass, and loss of recreational opportunity. Effects include loss of recreational fisheries (including shellfish), reduction of recreation in or on the water, such as swimming and boating, loss of tourism and decline in property values (Dodds et al. 2009; Birch et al. 2011; Compton et al. 2011).

Methods

There is not a very large literature on this topic, but the metrics for evaluating the consequences of eutrophication-caused algal blooms to recreation have mostly been through economic valuation (Birch et al. 2011). Two methods are applied to evaluate these costs. The first is developing estimates of the economic damage, abatement or mitigation incurred through loss of a targeted attribute, such as loss of recreational fishing. The second is willingness to pay for surveys.

Nitrogen damage can be attributed to a given source and given an economic value (Birch et al. 2011; Compton et al. 2011; van Grinsven et al. 2013). The change in damage cost (mitigation, remediation, direct damage or substitution) according to change in N loading can be calculated for specific N sources, such as synthetic fertilizer, and specific recreational or environmental impacts, such as damage to recreational fisheries or loss of property values (Birch et al. 2011; Sobota et al. 2015).

Some, but not all, of the benefits of recreation fall outside existing economic markets. For these, willingness to pay (WTP) surveys have become a standard method for evaluation. Some human benefits, such as their existence or conservation value, may not be directly related to any particular use of marine or freshwater resources. The value of these resources, or the value of protecting them from nutrient pollution, can be captured with the help of stated preference surveys, which measure WTP. Some studies have estimated N impacts on these less tangible services. For example, Jones et al. (2014) calculated impacts of N on biodiversity by linking N impacts to critical load exceedance via WTP values. This approach has been further developed by combining dose response functions for N impacts on species richness, and WTP to protect biodiversity. The study assessed the economic benefits of policies to reduce N emissions through the resulting improvements to biodiversity in the UK (Jones et al. 2018).

Interactions with other drivers

Other drivers contribute to eutrophication and HABs in addition to excess nitrogen. Other nutrients, climate, addition or removal of grazing or predatory animal species, and changes in hydrologic patterns can all influence primary productivity of water bodies. The contribution of N to effects arising from multiple causes has been termed the "N share (Gu et al. 2021)" and is calculated as a percentage of all other causes leading to the particular effect.

4.6.4 Odour

Wastewater treatment systems and agricultural operations, especially intensive livestock operations can generate offensive odours from air emissions (Schiffman et al.1998; Misselbrook 1993). While most wastewater odours come from sulphur-based compounds such as hydrogen sulfide, nitrogen-based compounds can also cause odours. Nitrogen-based odorous compounds include ammonia, the amine family of compounds (such as ethyl amine, trimethylamine), and especially indole and skatole (Schiffman et al.1998). Indole and skatole are volatile fatty acids that occur naturally in fecal materials of livestock and poultry. and are primary contributors to fecal odour.

Methods

Odour can be assessed by two criteria: strength, which is measured as concentration or intensity, and offensiveness (i.e., the perceived quality) (McCrory & Hobbs 2001; Zhu 1999). Odour threshold concentrations (OTC) are determined by filling polytetrafluoroethylene (PTFE) bags with feedlot air and measured with an olfactometer (Zhou et al. 2016). Olfactometers are devices built to present odour stimuli in a standardised computer-controlled manner with determined air flow, odour concentration, odour duration, onset and offset. The odour profile method (OPM) for offensiveness developed by Burlingame (1999) is used as the sensory evaluation of gas samples at the University of California Los Angeles (UCLA) by an 'odour panel' of trained personnel. Panelists are taught to identify multiple odour characters and their respective intensities in a single sample and to rate their odour intensity. The intensity of an odour characteristic is a measure of its odour strength, which is related to the log of its concentration via the Weber-Fechner law Eq. (Muñoz et al. 2010). OPM is a modification of Standard Method 2170: the Flavor Profile Analysis Method (FPA) (American Public Health Association 2012). The odour composition itself is identified via gas chromatography-olfactometry and gas chromatography- mass spectrometry techniques (Trabue et al. 2011).

Interactions with other drivers

There are other odour-causing substances, but in feedlots odour is primarily caused by N compounds. The relative importance of different odour causing substances changes with distance away from feedlots. For example, amines, including indole, increase in importance with distance. The relative contribution of substance to odour is influenced by climate drivers such as precipitation, temperature, wind velocities and direction.

4.7 Non-agricultural nitrogen products

4.7.1 Industrial nitrogen

Haber-Bosch nitrogen (N) has been increasingly used in a wide range of industrial products (e.g., plastics, explosives, fuels) in addition to its use for fertilizers. There are a very large number of industrial N compounds with definite consequences, but which have received little notice from an environmental perspective. The industrial N flux (excluding N fertilizer) is currently estimated to account for over 20% of total global Haber-Bosch N fixation (HBNF). Products include synthetic fibers, plastics, synthetic rubber, synthetic resins, glues, explosives, paints, rocket fuel and medicines. These products are the basis of modern industrial production and human life.

A great difference between industrial N uses from other uses is that the N compounds must be refined with higher purity for industrial products. For example, N compound gases for the semiconductor industry requires more than 99.999% purity. Similarly, urea for medical and cosmetic use needs higher purity than urea for fertilizer use (Gu et al. 2013).

The global industrial N flux has been estimated at 25.4 Tg N yr^{-1} (for around 2008), which is comparable to the NO_x emissions from fossil fuel combustion. More than 25% of industrial products (primarily structural forms, e.g., nylon) tend to accumulate in human settlements due to their long service lives. Emerging N species define new N-assimilation and decomposition pathways and change the way that N is released to the environment. The loss of these N species to the environment has significant negative human and ecosystem impacts (Gu et al. 2013).

The estimated per capita industrial N flux reaches 12-15 kg N yr^{-1} in Western Europe and North America, but it is only 0.3-0.6 kg N yr^{-1} in Africa and South Asia (Figure 4.7.1). This industrial N flux represents ~30% of HBNF in developed countries, but is generally less than 15% in developing countries. In Japan, which imports most of its food, more than 50% of Haber-Bosch N produced has been estimated to be used for industrial use (Katagiri et al. 2018).

Given the large differences in per capita industrial fluxes between developed and developing countries (Figure 4.7.1), an enormous potential increase of industrial N is expected in developing countries as economic development proceeds. If the global per capita consumption of industrial N reaches the current level found in developed countries, approximately 12-15 kg N yr^{-1}, then the total global industrial N flux would increase at least 3-4 times. These changes could be even bigger in future if the use of ammonia as a future fuel also becomes widespread (IRENA and AEA 2022; International Nitrogen Assessment, Chapter 5).

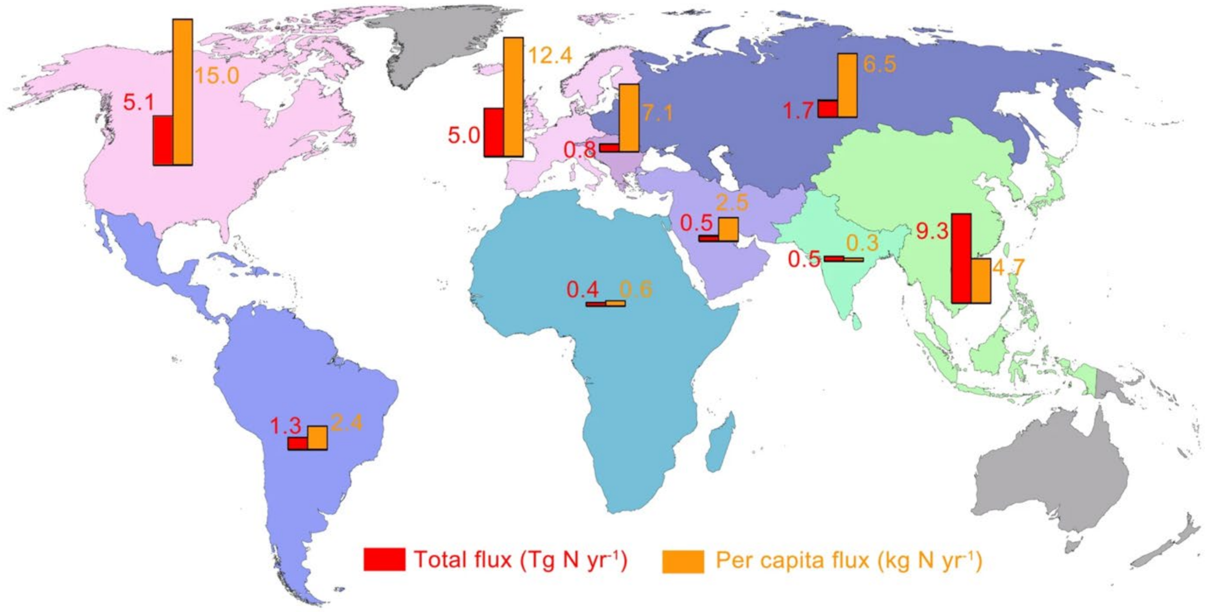


Figure 4.7.1. Industrial reactive nitrogen (N) fluxes in different continental regions. Coloured bars represent industrial N fluxes as totals and on a per capita basis. The global map was created with the GMAP function in SAS 9.1 (SAS Institute Inc., Cary, NC, USA). Map redrawn with permission (Gu et al. 2013). Reprinted with permission from the Authors. Copyright 2013 American Chemical Society.

Methods by which materials fluxes are estimated

The N fluxes of global structural industrial products can be estimated from total production and N concentrations (Table 4.7.1). However, information on the production and N concentration of non-structural industrial N is not available because many more N species are involved in this group than in structural industrial N. Gu et al. (2013) investigated the historical trends in the global industrial N flux by assuming that industrial N flux was approximately equal to the difference between total HBNF and N fertilizer production and distribution loss. Katagiri et al. (2018) applied an input-output analysis for estimating industrial N flux within the economy by calculating N budgets for five major economic sectors in Japan.

Processes by which industrial N contributes to reactive N in the environment

There are seven principal categories and 18 subcategories of N species, such as azo and diazo compounds, which are produced as intermediates in industrial processes. More than 50% of these intermediates represent to 'emerging nitrogen species (i.e., new synthetic substances), since they do not exist in the natural environment (Table 4.7.1).

The final products of industrial processes have also contributed new N species. For example 'nylon' is actually a family of polyamide fibre compounds, which includes approximately 140 or more species of N-containing compounds, estimated based on their molecular weights, including aliphatic, aliphatic-aromatic and aromatic nylon. Nitrogen-containing pesticides comprise more than 60 chemical species, including nitrile, amidine, heterocyclic, amine, cyanide, nitro and urea-based compounds. At least 118 species of azo dyes have been banned by the German government due to their carcinogenicity (Gu et al. 2013; https://etad.com/wp-content/uploads/ETAD-Information-Notice-No-6-rev.-2008-.pdf). The decomposition of industrial nitrogen products follows various pathways that are determined by the processes and uses associated with these products. For example, the burning of polyurethane releases hydrogen cyanide (HCN) and other N-containing compounds to the air, while landfills containing industrial products release N to the hydrosphere.

Table 4.7.1 Intermediates and final products of industrial production. Table produced for this chapter by Baojing Gu.

Category	Subcategory	Toxicity	Final products	Service life	Existing in nature
Nitrate ester ($-ONO_x$)[a]	Nitrate	low	fuel, medicine, explosive	< 1 yr	no
	Nitrite	low	medicine, reagent	< 1 yr	no
Amines ($-NH_yR_{2-y}$)[b]	Aliphatic	low	reagent, medicine	< 1 yr	many
	Aromatic	high	dye, resin, medicine, pesticide	> 1 yr	no
Amides ($-OCNH_2$)	Amide	low	reagent, fiber, plastic, rubber, resin	> 1 yr	few
	Urea	low	medicine	< 1 yr	many
	Hydrazone	low	dye	> 1 yr	few
	Amidine	low	pesticide	< 1 yr	no
Nitriles ($-C{\equiv}N$)	Nitrile	high	fiber, plastic, rubber	> 1 yr	some
	Isonitrile	high	pesticide, medicine	< 1 yr	some
	Isocyanate	high	pesticide, medicine, fiber, plastic, rubber	> 1 yr	no
	Oxime	high	medicine, fiber, reagent	> 1 yr	no
Nitro ($-NO_x$)	Nitro	high	explosive, medicine, paint, dye, reagent	< 1 yr	few
	Nitroso	high	medicine, reagent	< 1 yr	few
Azo ($-N{=}N-$)	Azo	high	dye, reagent	> 1 yr	no
	Diazo	high	dye	> 1 yr	no

Category	Subcategory	Toxicity	Final products	Service life	Existing in nature
	Hydrazine	high	fuel, medicine, rubber, resin	< 1 yr	some
Heterocyclic	N Heterocyclic	high	dye, medicine, pesticide	< 1 yr	many

a, x = 1 or 2; b, y = 0, 1 or 2.

Industrial production processes typically show a relatively low proportion of N loss. This proportion of N loss can be expressed as N input for production not contained in the final products, divided by total N input for production). Values are estimated at e.g., ~2% for the United States in 1996 and ~5% for China in 2008. Compared with the high initial proportion of N loss in agricultural production (50% to 80%), the low proportion of N loss in industrial production only resulted in 0.5-1.3 Tg N yr^{-1} released to the environment in 2008 worldwide (based on a 2-5% N loss rate) (cf. Gu et al., 2013).

Nevertheless, the N release during industrial production belongs to the category of point emissions that usually have an extremely high N concentration and cause deterioration in the local environment. For example, it has been suggested that groundwater nitrate concentration beneath industrial land use is much higher than that beneath other land uses, e.g., cropland and human settlements (Gu et al. 2013). Moreover, the composition of the N emissions is complex due to the massive contribution of N species during industrial production, while recognizing that approximately one-half of the intermediates and their end products are toxic (Table 4.7.1).

The emissions of these complex toxic N species can have serious effects on the health of people who work in or live near the factories. For example, the chemical hydrazine (N_2H_2) is used, amongst others, as a rocket propellant, a corrosion inhibitor and as an intermediate for agricultural chemicals. It is highly toxic, and the leakage of hydrazine can irritate eyes and cause delayed inflammation and can also have strong corrosive effects on skin if the exposure involves a concentration greater than 0.06 mg m^{-3} (Choudhary and Hansen 1998).

In addition to effects of specific nitrogen compounds, emitted N species, such as NO_x emissions from the production of nitric acid, can react with other components (e.g., volatile organic compounds, VOCs) to form near ground-level ozone and fine particulate matter that penetrate and are deposited in the lungs, causing serious harm to human health (Peel et al. 2013).

To understand the environmental fates of industrial products, Gu et al. (2013) classified industrial N compounds into structural and non-structural N. The category of structural N is defined as the set of N-containing products that offer physical support services (e.g., housing, furniture, clothing, transportation) or combine with physical support services (e.g., dye, paint). Based on their functions, structural industrial products are divided into 6 principal categories: fiber, plastic, rubber, resin, dye, and paint (Table 4.7.2). Each category includes hundreds to thousands of N species.

Structural nitrogen has a low N turnover rate because of its long service life, the length of time over which an industrial product can maintain its function. The long service life of structural industrial N leads to the persistent accumulation of N in human settlements prior to its release to the environment. The current N release from structural N is from materials produced years to decades ago, not from recently produced materials. Hence, the current patterns and fluxes of structural industrial N release may reflect the historical industrial N cycle more accurately than they reflect the contemporary industrial N cycle.

Table 4.7.2. Quantities and N fluxes of structural industrial products produced worldwide circa 2008. Table produced for this chapter by Baojing Gu. **Key:** *AS (SAN): styrene-acrylonitrile copolymers; ABS: acrylonitrile-butadiene-styrene copolymers; PU: polyurethanes. ** no data available; # the average value is 6.3.

Species	Quantity (Tg)	N content (%)	N flux (Tg)
Synthetic fiber			
Polyamide fiber	3.9	12.4-32.6	0.5-1.3
Acrylic fibers	2.0	26.4	0.5
Polyurethane fiber	0.3	23.7	0.1
Plastic*			
AS (SAN)	7.5	8.9	0.7
ABS	12.3	4-9	0.5-1.1
PU	13.5	23.7	3.2
Synthetic rubber			
Nitrile rubber	0.8	13	0.1
Acryl-nitrile rubber		4.7-12.1	
Dye	1.3	**	**
Paint	36.6	**	**
Resin	**	**	**
Total	>78.2		>5.6-7.0#

The accumulated structural industrial N can be inadvertently released into the environment. Concrete admixtures, plates, furniture and decorative materials can release NH_3 and toluene diisocyanate (TDI) to produce indoor air pollution. Toxic N can also be released during fires or other emergencies. The burning of polyacrylonitrile materials can release 0.06 g HCN per gram polyacrylonitrile. HCN is highly toxic and sometimes fatal to humans. More than 50 people were killed in a fire disaster in Shanghai in November 2010. These deaths were largely attributed to the emission of HCN from the burning of polyacrylonitrile and polyurethane foam used in the redecoration of the building. Burning for energy recycling is an important type of waste treatment involving structural industrial N, and it could cause direct environmental and health risks for local or downwind residents. This process results in aggravated reactive airways disease (RAD), coughs, asthma and chronic respiratory disease if no steps are taken to reduce the N contained in the burning structural industrial products. In landfills containing structural industrial N, the groundwater quality around the sites has been seriously impacted by the leaching of pollutants, especially in developing countries with rapid urbanisation, such as China, India and Mexico (Gu et al. 2013).

In contrast, **non-structural industrial nitrogen** generally does not involve physical support services, but is associated with chemical reactions or biological metabolism. The principal non-structural industrial products are explosives, rocket fuel, medicines, pesticides and reagents. In Japan, non-structural industrial products are used and essential for five major industries (as based on annual Japanese Yen expenditures): medical service, office supplies, health and hygiene, motor vehicle maintenance and machine repair service, and image and sound information production (Katagiri et el. 2018).

The turnover rate of non-structural industrial N, like that of food and fuel N, is relatively high because the N contained in these products is usually released to the environment within one year. The N contained in explosives and rocket fuel is released to the environment after use. For these compounds detonation and combustion are designed to produce non-reactive di-nitrogen (N_2). However, an uncertain fraction can also be released as reactive nitrogen compounds such as NO_x, N_2O and NH_3. Some forms of nitrogen compounds released to the environment are persistent. For example, dumping of the explosive trinitrotoluene (TNT) into the Baltic Sea after the 2nd World War has been associated with toxic accumulation in food chains (Schuster et al. 2021).

Pesticide N can persist in the environment for a relatively long time, but this duration is still less than one year. The half-lives of nitrile and amidine pesticides are as high as ~60 days; for heterocyclic and amine pesticides, the half-life is approximately 45 days; whereas for cyanide, nitro- and urea pesticides, the half-life ranges from 15 to 30 days prior to decomposition to simple N-containing compounds (Gu et al. 2013).

The quantity of N released to air, soil or water from non-structural industrial N depends on the processes and uses associated with these products. However, the total N released to the environment from non-structural industrial N should be smaller than its total flux, ~19.1 Tg N yr^{-1}. Generally, the use of explosives will cause rapid emissions of N_2 and reactive N (NO_x, N_2O, NH_3) to the atmosphere, and the emitted N would photochemically or chemically react with other chemicals (e.g., VOCs) and may be further deposited on land and water. Medicines decompose or are discharged directly into domestic wastewater through the human metabolic system. However, pesticides, depending on their relatively long half-lives, volatility and fat-solubility, disperse globally. They can bioaccumulate in food chains and affect human health and the health of other species far from the point of release and for many years after release.

In addition to the effects of N released to the environment, non-structural industrial N can also affect the environment and human health via direct usage. Medicines related to industrial N have made a profound and substantial contribution to the improvement of human health, although drug resistance and the side effects of certain medicines may also damage human health.

4.7.2 Nitrogen for munitions and explosives

> *'Haber's other motivation, not mentioned in his lecture* [for the Nobel Prize award], *was to provide the raw material for explosives to be used in weapons, which requires large amounts of reactive nitrogen. Haber's discovery has therefore had a major influence on both World Wars and all subsequent conflicts'* (Erisman et al. 2008).

Nitrogen has always been very important for warfare. In early times, when settlements started to produce more food than the community needed, not all men were needed in the field and that led to the possibility of using this capacity for warfare. Food for soldiers became a requirement and this put pressure on agricultural production to fuel wars (cf. Erisman et al. 2008).

It is thought that gunpowder was discovered in China, perhaps around 1000 AD, including the recognition that saltpetre (potassium nitrate) is needed to produce it. In the mid-13th century, Roger Bacon mentioned gunpowder, which may have been imported from China (Needham et al., 1987, pp 47-50). The availability of gunpowder led to a change in warfare with guns and canons for more effective destruction of the enemy. It also increased the demand for saltpetre. In across Europe so-called nitre beds were created to harvest saltpetre: soil with sheep dung, ashed wood and straw. In the UK there is mention of the saltpetre collectors, the Peterman, who collected saltpetre from the houses (Buchanan 2006; Gorman 2014).

The central role that N has in the manufacture of explosives is reflected not only in the Nobel prizes awarded to Haber and Bosch, but in the very origin of the Nobel Prize itself. Alfred Nobel's wealth was built on the development of safe methods for using nitro-glycerine, and his patents for dynamite and gelignite eventually financed the Nobel Foundation (Erisman et al. 2008).

Methods for assessing impacts of Haber-Bosch production of munitions

Haber's discovery therefore fuelled the First World War, and, ironically, prevented what might have been a swift victory for the Allied Forces. Since then, reactive N produced by the Haber–Bosch process has become the central foundation of the world's ammunition supplies. As such, its use can be directly linked to 100–150 million deaths in armed conflicts throughout the twentieth century (Erisman et al. 2008).

Currently, there are estimates about the annual deaths by guns of 500,000 people (Nuwer 2018). In addition to this there is a steady increase in the number of people killed by explosives, currently estimated to be around 44.000 people annually (Norton-Taylor 2016). Apart from people, the use of gunpowder also led to the loss of many animal lives (although this has not been quantified).

Chapter 5: Examples of nitrogen impact-related policies around the globe

5.1 Introduction

Nitrogen impacts are measured for a variety of reasons, but always with some further step in mind, whether for the development of policy to reduce negative effects or increase positive ones, or to gain further insight into the economic, health or ecological dimensions of the N cascade. Methods for assessing N threats or benefits may then constitute the beginning of a process that leads next to the review of N impact data by stakeholders and finally some action taken by the stakeholders. This chapter provides examples of how N impact data have been used to drive public policy. Flux-impact path models and economic analysis are addressed in depth in elsewhere in INMS (e.g. van Grinsven et al. 2026; Gu et al., 2026 and in Part C of the International Nitrogen Assessment).

For this chapter, we invited experts from several countries and regions to submit examples illustrating how the science behind understanding positive and negative N impacts has been applied at international, national, regional and institutional scales. We then placed each submitted example into the Driver, Pressure, State, Impact, Response (DPSIR) framework and context (Chapter 1). The examples illustrate very different stages of thinking about impacts of excess reactive N to health and the environment. In some examples, such as that of the Marine Strategy Framework Directive (MSFD), public perception (response) of environmental damage prompted the establishment of scientific programs to identify drivers, pressures, states and impacts. Monitoring states and impacts provides information over space and time. Assessments relate the states and impacts to the drivers. For the MSFD, as presented by the contributors, the policies remain informational for nations or regions to use as they see fit. In other examples, scientific results can drive the policy, as with examples from the Netherlands, Aotearoa New Zealand and Rocky Mountain National Park, where there is a full DPSIR connection between scientific understanding of impacts leading to policies. The actions then lead to actions toward drivers and pressures. Other examples in this chapter illustrate where the policy and responses have not yet incorporated environmental impact assessments. Combined, the examples represent a continuum of thinking systematically about the full threats and benefits of N to society and environment.

The DPSIR framework was originally conceived as one way of providing information for policy makers. As such, it is not the only way to use scientific findings to refine the use of N for beneficial purposes and minimize environmental and health damages. Some examples in this chapter were never intended to inform policy or have not been considered in that context. Such examples are less representative of DPSIR, but the framework could serve as a guide for future considerations to visualize how drivers and pressures affect states and impacts, leading toward meaningful policy responses.

5.2 Agricultural policies to promote nitrogen use in Kenya

In Sub-Saharan Africa (SSA), agriculture plays a vital role in the rural economy. In Kenya, the sector accounts for 65% of export earnings and employs more than 70% of Kenya's rural people, meeting food security needs and providing employment and income (Argus 2016). Increased fertilizer use contributes greatly to increased food production. However, SSA countries apply only 17 kg of fertilizer per hectare (kg ha^{-1}) of arable land on average, which is far below the global average of 135 kg ha^{-1} (FAO 2014).

There have been several policy interventions in SSA to promote increased fertilizer use. The “Abuja Declaration on Fertilizer the African Green Revolution of 2006” recommended increasing fertilizer use 50 kg/ha by 2015 (AfDB 2017). The Kenyan Institute of Policy Research Analysis (KIPPRA 2014) increased fertilizer use to boost productivity of small farms. However, despite all these declarations to increase fertilizer use in SSA countries, fertilizer consumption in some countries is still low, including Kenya (Figure 5.1). In December 2017, the consumption of fertilizers in Kenya was 800,000 Mtonne (amount of all fertilizer products), with 53% and 17% of the country's fertilizer consumed by maize and tea, respectively (Onyango 2018).

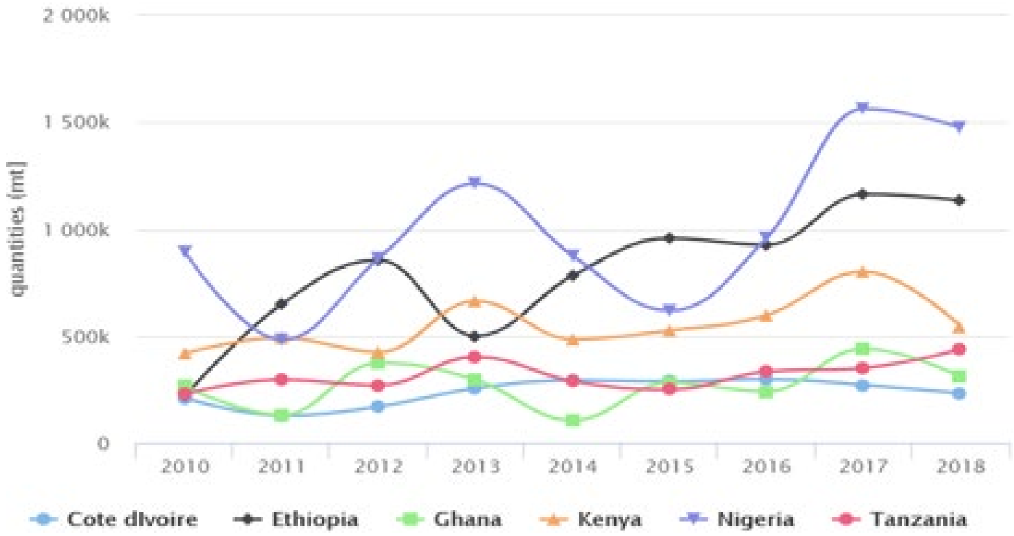


Figure 5.1. Fertilizer consumption trends in some Sub-Saharan Africa (SSA) countries from 2010 to 2018 (Data: https://africafertilizerise/statistics/; Figure: provided by Mekonnen Giweta and Cargele Masso from informal case study).

Agriculture is the backbone of Kenya's economy and accounts for 30% of gross domestic product (Boulanger et al. 2017). However, like many of the SSA countries, low productivity is a key challenge. Low use of fertilizer and poor access to market were identified as the major bottlenecks for Kenyan agriculture (Mason et al. 2016; Boulanger et al. 2017). To address this, in the last couple of decades, the Kenyan government introduced policy changes to promote an increase in fertilizer use by smallholder farmers (Boulanger et al. 2017). Kenya liberalised the fertilizer market in the 1990s, and it is considered an example of successful private sector-led fertilizer market development (Mason et al. 2016). Even though the fertilizer market was liberalised Kenya, the high cost prevented small-scale farmers from being able to afford enough fertilizer (Mason et al. 2016. The Kenyan government's policy documents, including Vision 2030 and Agriculture Sector Development Strategy (ASDS), prioritised reduction of fertilizer cost as one of the policy interventions to increase agriculture growth by an annual average of about 6.4% (Boulanger et al. 2017; Government of Kenya 2007, 2010).

Targeted input subsidy programs, such as the National Accelerated Agricultural Inputs Access Programme (NAAIAP), were also initiated by the Kenyan government in 2007 for improved seed and inorganic fertilizers (Ariga & Jayne, 2010; FAO 2009). NAAIAP aims to promote agricultural input use, input market development, agricultural productivity and food security, and the programme was expanded to reach 2.5 million disadvantaged farmers/households with maize seed and fertilizer 0.4 ha through a voucher scheme (FAO 2009).

According to the Food and Agriculture Organization of the United Nations (FAO 2009), the fertilizer subsidy programme has potential benefits in the long term in terms of efficient use of inputs, rapid adoption of new technologies, and an increase in the income of rural households. However, implementation of the programme faced challenges, such as poor infrastructure for the development of efficient input marketing system; financial burden on the government; and requirements for additional investment in agricultural extension services. A policy scenario analysis indicated that introduction of a fertilizer subsidy programme increases fertilizer use and crop productivity (Ayenew et al. 2016). However, it should be accompanied with training on sustainable agricultural practices to increase fertilizer use efficiency (FUE) and reduce negative impacts on the environment (Ayenew et al. 2016). In line with this, another study concluded that fertilizer use policy interventions should be accompanied by effective extension and efficient market access for fertilizers and agricultural products (Boulanger et al. 2017). In addition to input subsidies, the Kenyan government built a large fertilizer manufacturing plant at Eldoret as a cost reduction strategy (Andae 2015). Currently 10,000 MT of single super phosphate (SSP) are produced in Kenya, while most other fertilizers are imported (Onyango 2018; Figure 5.2). In Kenya, the fertilizer industry is mainly driven by the private sector, which imports most fertilizer (Onyango 2018).

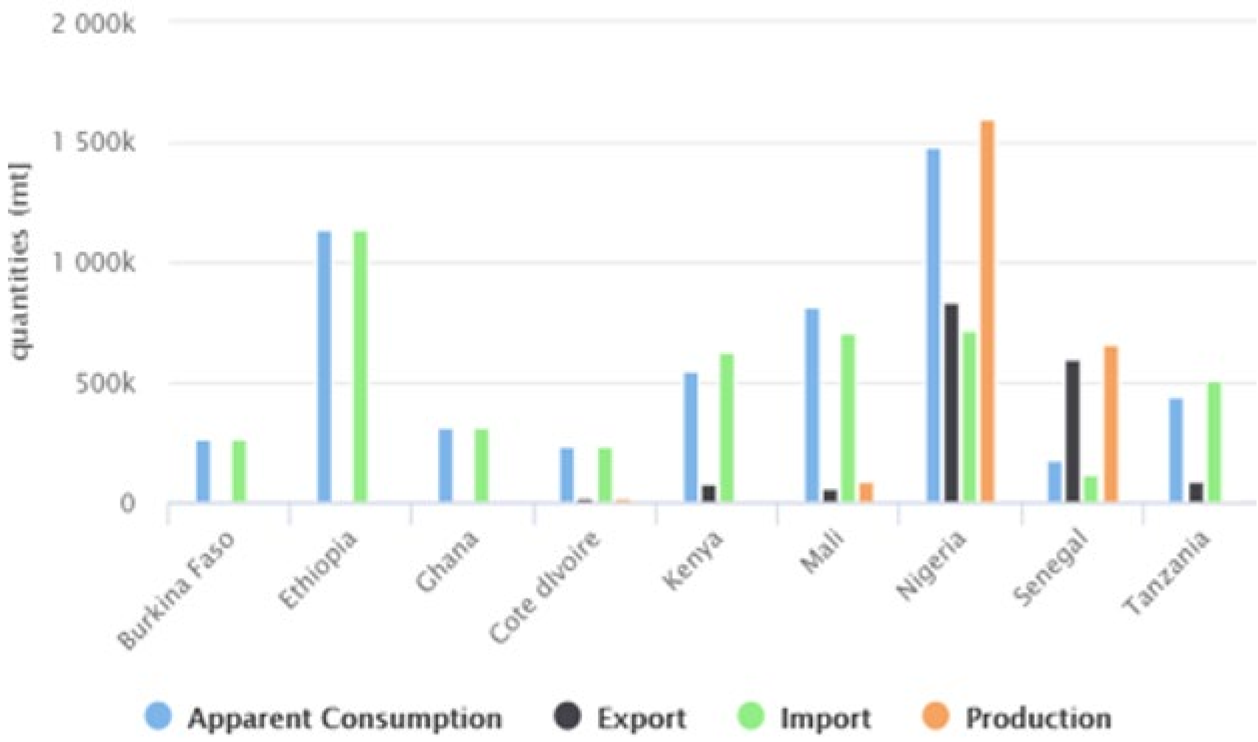


Figure 5.2. Fertilizer production, imports, exports, and apparent consumption in 2018 in nine Sub-Saharan Africa (SSA) countries (Data: https://africafertilizerise/statistics/. Figure provided by Mekonnen Giweta and Cargele Masso in informal case study).

The Fertilizers and Animal Foodstuffs Act (amendment) 2015, Cap 345, was passed to regulate the importation, manufacture and sale of agricultural fertilizers and animal foodstuffs. However, implementing this policy remains a challenge (Tarus et al. 2015). For example, although Kenya Plant Health Inspectorate Service (KEPHIS) performs a risk assessment before introduction of live organisms such as bio-fertilizers from abroad, it has no legal mandate to regulate bio-fertilizers (Tarus et al. 2015). Therefore, there are no penalties for non-compliance in the importing, registering, distributing and regulating processes of bio-fertilizers (Tarus et al. 2015). Establishment of relevant regulatory bodies is needed to implement existing agricultural policies and laws, as well as the development of a soil health inputs policy to provide guidelines for improved adoption of soil health technologies (Mangale et al. 2016). The major challenges for implementation of these polices include:

i. inadequate laboratories, equipment, and personnel to carry out quality control,
ii. high cost of fertilizers,
iii. inadequate capacity for soil analysis, and
iv. lack of knowledge (Onyango 2018).

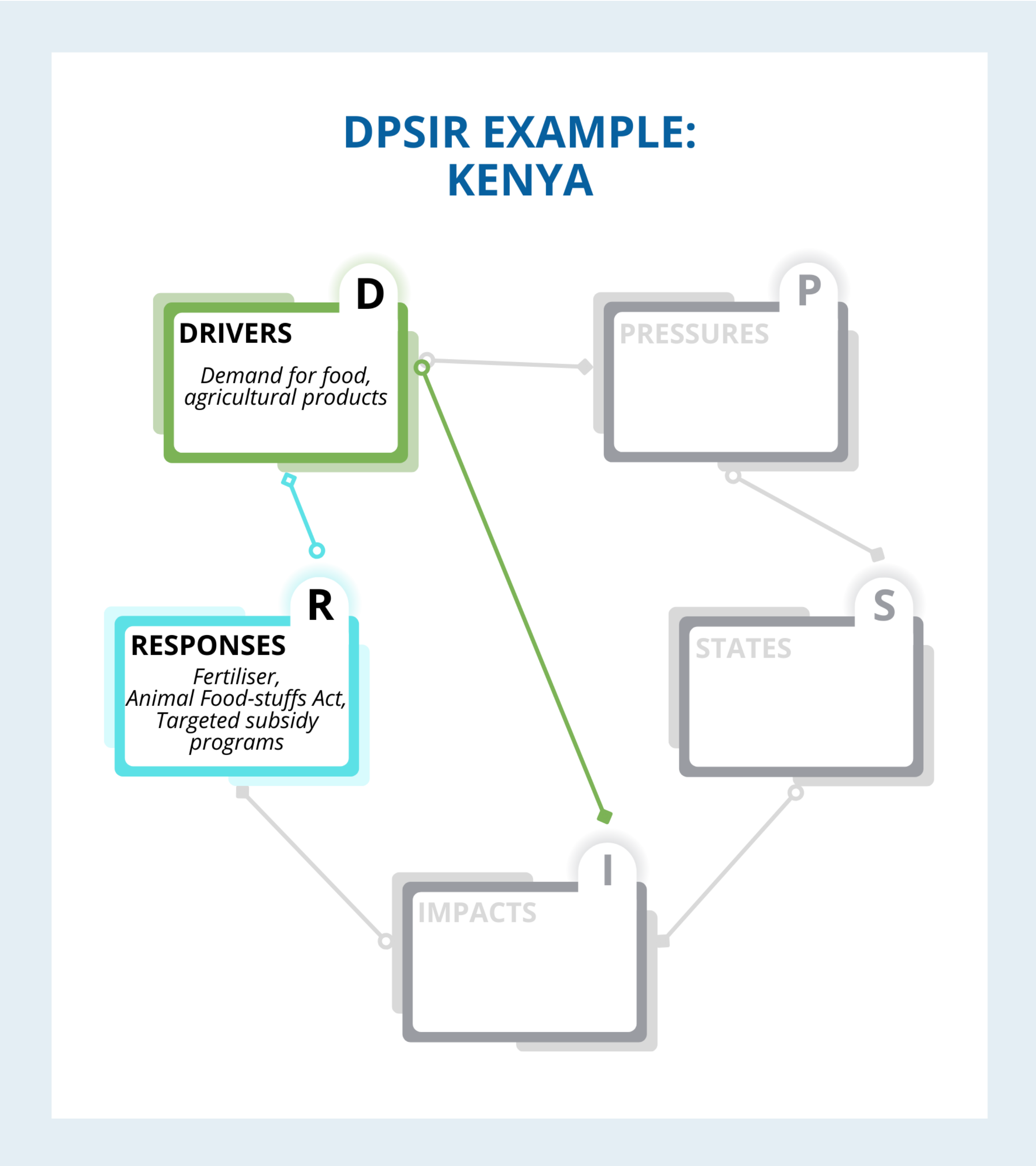


Figure 5.3. Drivers, Pressures, States, Impacts and Responses (DPSIR) diagram for agricultural nitrogen (N) policies from Kenya. Grey arrows indicate currently unquantified pathways. Original graphic produced for this document © UKCEH 2025.

The example provided from Kenya provides a perspective on N that relates solely to food production. While critically important, there is an opportunity to also consider policies that can help to minimize environmental impact. The INMS East Africa Demonstration Project, centered on Lake Victoria Basin, represents the paradox of too little N available for some agriculture and too much in others.

> 'The N used for production is too little due to insufficient use of N inputs, whereas the eutrophication of Lake Victoria at selected sections has been related to high loading of nutrients mainly N and P. This is a consequence of unsustainable agricultural practices, deforestation, erosion, encroachment to marginal

lands because of population pressure and low crop productivity on unit area and nutrient inputs from municipal wastewater' (https://www.inms.international/regional-demos/east-africa-demonstration).

Nitrogen pollution of Lake Victoria comes from other sources in addition to agriculture, and the contribution from all sectors (municipal/industrial waste, atmospheric deposition, agriculture and natural areas) is not well understood. Studies are ongoing to develop a more complete view that includes states, impacts, and responses to N inadequacy and excess.

5.3 Impacts of government policies on fertilizer use in India

The introduction of India's Green Revolution in 1965 championed by M.S. Swaminathan marked a period in which agricultural practices shifted from traditional to technology-based methods and eventually to intensification. The Green Revolution introduced high-yielding, disease-resistant varieties (HYVs) of crops, which benefited from fertilizer addition and led to increased fertilizer use throughout the country.

The increase in usage was further aided by the government's decision to subsidize fertilizers through favourable price policy, and this has continued in recent decades. For example, the consumption of nitrogen (N), phosphorus (P) and potassium (K) fertilizers has increased from 17.36 Mtonne in 2001-2002 to 26.59 MMT in 2017-2018. The prevalent trend among farmers is to employ primarily those fertilizers that are subsidised by the government. In particular, high subsidies on urea have resulted in irrational and unbalanced use, even though the soil needs secondary and micronutrients as well to maximize crop production. Excessive use of urea also has various negative environmental impacts. As plants uptake only a fraction of the nitrogen-based fertilizers, the remainder accumulates in the soil, is lost as run-off, or leaches into groundwater. It therefore requires the commitment of several relevant governmental bodies to regulate the production, distribution and balanced consumption of fertilizers to increase crop production without harming ecosystems.

Since independence, the Indian government has been regulating the sale, price, and quality of fertilizers, which were recognised as essential commodities under the Essential Commodities Act of 1955. Under the Fertilizer Control Order (FCO) of 1985, the government holds the right to decide the price, distribution and quality of fertilizers. In 2008, the Department of Fertilizer announced the Uniform Freight Policy (UFP), the objective of which was to ensure that every part of the country, especially remote villages, had access to fertilizers at the same price. However, the government's Nutrient Based Subsidy (NBS) policy, effective April 1, 2010, allowed private corporations to decide the maximum retail price of fertilizers, which resulted in a substantial increase in the price of P&K fertilizers, causing a spike in the use of government-subsidised urea. As of 2016, the consumption ratio of N, P and K fertilizers in India was 6.7:2.4:1 against their recommended dose of 4:2:1. In Punjab and Haryana, the ratios are 31.4:8.0:1 and 27.7:6.1:1, respectively (Deshpande 2017).

To cut excessive consumption of urea and to promote balanced use of fertilizers, the government has more recently decided to sell urea in 45 kg bags instead of 50 kg bags at 242 Indian Rupees (ET Bureau 2019). This has reduced urea use by 10%, as farmers are mostly accustomed to using the packaged urea on the field instead of specifically weighing fertilizer as per the recommended rate on an area basis.

Excessive use of urea has several detrimental effects on ecosystems. Plants take up only 30-40% of the urea-N while another 30% remains trapped in the soil. The rest is either leached into underground aquifers or to surface water, causing groundwater pollution or surface water eutrophication, or is denitrified as gaseous N_2 or as N_2O and NO_x, causing atmospheric pollution.

India's Cabinet Committee on Economic Affairs (CCEA) approved the Neem Coated Urea policy in 2015 (India PIB 2015). The policy made it mandatory for all indigenous urea producers to produce 100% of their total output of urea as neem-coated urea. The policy was designed to avoid diversion of subsidised urea towards non-agricultural purposes including industry, as well as to reduce detrimental effects of N on the environment and to increase the overall N-use efficiency by crop plants, This form of urea, which is prepared by spraying 0.03% neem (Margosa) oil at the final prilling stage, inhibits nitrification, reducing the nitrification-related losses, making fertilizer-N available at the root zone for a longer period. Initial estimates show lower consumption of urea by 15-20% (PTI 2015). As per Fertilizer Association of India (FAI) statistics, N-fertilizer consumption in India increased from 16.75 Mtonne in 2013-14 to 17.37 MMT in 2015-16. If the same rate of increase were maintained, the expected urea consumption in 2018-19 would have been 18.30 MMT growing at an average annual rate of 3.81%. But with the introduction of neem-coated urea in 2015, the actual consumption in 2018-19 was 17.64 MMT — a reduction in use of 0.66 MMT.

The Government of India has also tried to create awareness on the balanced use of fertilizers among farmers by conducting training, field demonstrations and frontline demonstrations in association with the state government, Indian Council of Agricultural Research, South Asian Universities and fertilizer industry.

Evaluation of DPSIR approach for India

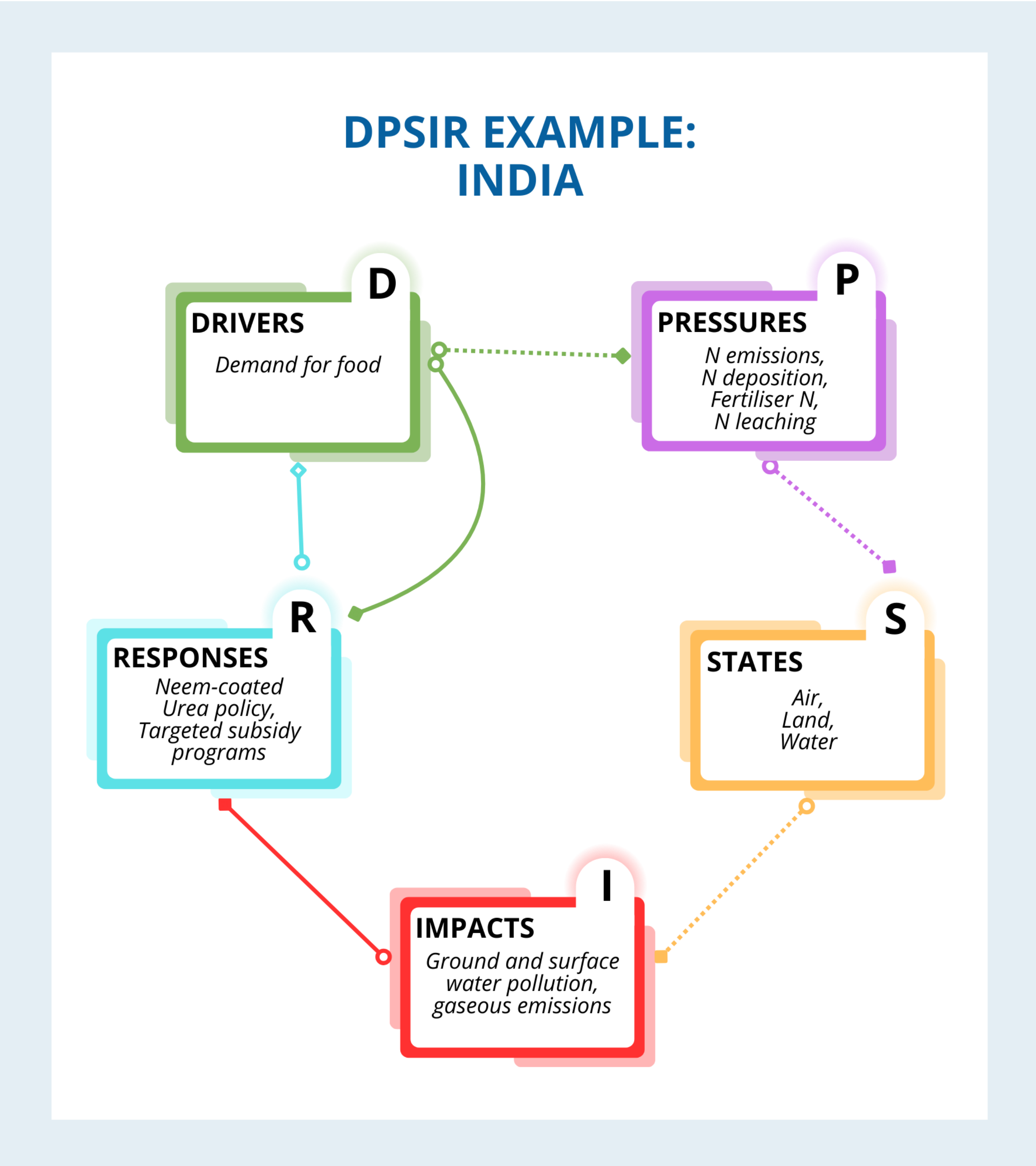


Figure 5.4. Drivers, Pressures, States, Impacts and Responses for nitrogen (N) policies in India. Solid lines indicate quantified pathways and dashed lines indicate aspirational goals. Consumption of fertilizer in India has grown substantially since the 1960s with accompanying environmental impacts. A 2017 workshop to stimulate cooperation and to develop a long-term vision for N research and its policy applications in India focused on improving agricultural practices to increase nitrogen use efficiency (NUE). However, participants acknowledged the need to address sustainability goals that include reducing environmental and human health impacts (Móring et al. 2021). These aspirational goals are noted with dotted lines in the DPSIR diagram. Original graphic produced for this document © UKCEH 2025.

5.4 The rise, fall and resurrection of the Mineral Accounting System in the Netherlands

Livestock densities and N and P surpluses per hectare of agricultural land in the Netherlands are among the highest in the world. These cause high emissions of NH_3 and NO_x to air and high concentrations of N and P in surface waters. To implement the European Union Nitrates Directive (EUND) for reducing nitrate pollution of waters, the Netherlands introduced, among other strategies, the Mineral Accounting System (MINAS) in 1998. MINAS comprised a farmgate balance approach to estimate surpluses of N and P, as well as a set of seven soil- and farm- type dependent statutory maximum surpluses, which decreased stepwise over time or imposed progressive levies if maximum surpluses were exceeded. Both the farmgate balance and the statutory maximum surpluses were based on scientific principles, and, therefore, MINAS is a classic example of science-based policy. However, a ruling of the Court of Justice of the European Union (CJEU) led to an early abandonment of MINAS in 2003 and forced the Netherlands to implement soil- and crop-type specific N application standards. Here, we describe the rise and fall of MINAS and discuss its apparent recent resurrection in the so-called KringloopWijzer (Annual Nutrient Cycling Assessment) farm tool.

The Netherlands has high agricultural productivity and exports much agricultural produce. In the 1990's livestock density was close to four Livestock Units (dairy cow equivalent) per hectare. Before regulations were put in place, total N inputs from fertilizer and manure to agricultural soils exceeded 500 kg N ha^{-1} yr^{-1} and N surpluses exceeded 300 kg N $ha^{-1}yr^{-1}$. Mean concentrations of nitrate (NO_3^-) in the upper groundwater of sandy soils ranged between 100 and 200 mg l^{-1} NO_3^- with peaks up to 250 mg l^{-1} in the south (van Grinsven et al. 2016). In 1990, agriculture emitted around 300 kt of NH_3 to air, which was a major source of acidification and eutrophication. Based on the environmental concerns of Dutch citizens and the introduction of European environmental directives, a series of regulations on the use of manure and fertilizer livestock number were implemented from the 1985 onward, but often only became effective with some delay. Core elements of the EUND, introduced in 1992 and designed with a strong input of the Netherlands, was to limit manure production to 170 kg N ha^{-1} yr^{-1} (the equivalent of about two dairy cows) and to restrict manure application to the growing season. The Dutch, science-based policy approach to implement the EUND allowed for a farm-specific and stepwise implementation in practice. MINAS was introduced in 1998 on high-density livestock farms with more than 2.5 livestock units per hectare, then on all other farms in 2001.

The MINAS farmgate N balance accounts for inputs of N (and P) via fertilizers, animal feed, animal manure, compost and other sources, as well as the N output (export at farm level) in harvested products and animal manure (Oenema and Berentsen 2005). The difference between total N inputs and outputs should not exceed certain 'acceptable' (levy-free) surpluses for N. The values of acceptable surpluses were based on scientific assessments of inevitable N losses from manure storages and the relation between N surpluses and exceedance of the nitrate standard for drinking water (50 mg l^{-1} NO_3^-) in the upper groundwater at farm level, as function of soil type and land use (grassland or arable land). In case of exceedance of levy-free surpluses, farmers had to pay levies proportional to the exceedance. Both levy-free surpluses and levies were tightened in the years after the introduction of MINAS in 1998. These were strong incentives to reduce the N and P surplus, mainly by: a) Reducing the input of N and P fertilizer or manure; b) reducing protein-N and P contents in purchased feed; and c) enhancing the output via harvested produce.

Between 1998 and 2006, when MINAS was in place, it was highly effective. The use of mineral N fertilizer decreased from 385 to 270 kt, and the use of N in feed concentrates decreased from 450 to 380 kt, causing an increase of NUE of the national agricultural sector from 31% to 40% (Figure 5.3).

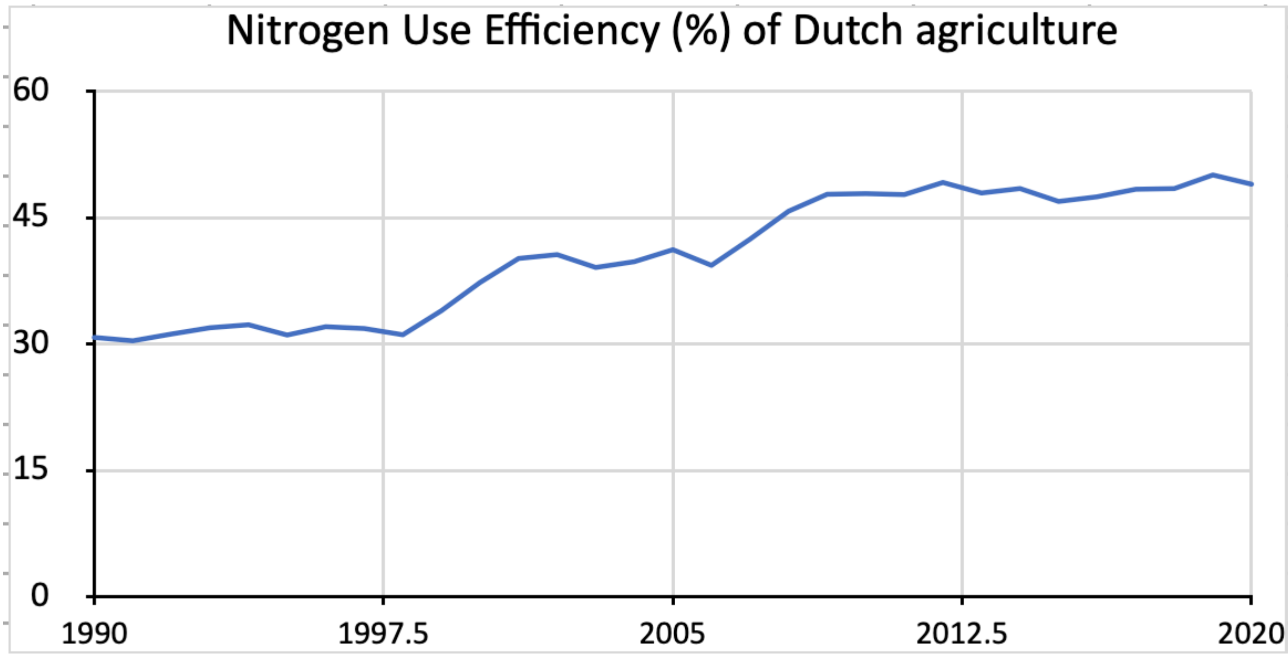


Figure 5.5. Evolution of the nitrogen use efficiency (NUE) of agricultural sector in the Netherlands. The useful outputs in NUE are specified as including crop, animal and manure products. Figure re-drawn based on data and figure from the Government of Netherlands available at https://www.clo.nl/indicatoren/nl060902-stikstof-efficientie-van-de-nederlandse-landbouw-1990-2020. Original graphic produced for this document © UKCEH 2025.

The enormous success of MINAS was only due to the levies, but there was a learning effect, especially for dairy farmers, who are the largest sector both regarding use of N and land. Most dairy farmers were able to meet the levy-free surpluses, and discussion of farmgate balances with other farmers and advisors made them aware of the imbalance between N inputs and crop and animal requirements, as well as the potential to save money on fertilizer and feed. However, pig and poultry farmers were mostly landless and did not have these MINAS benefits through optimisation. Instead, they had to pay hefty levies and costs for manure disposal and also because the default values for inevitable N losses from manure storages tended to be low. Hence, farmers had higher N surpluses on paper than in reality. The calculation of annual N and P surpluses per hectare for these intensive farms, with often less than 10 hectares, was inaccurate and highly variable as MINAS could not accurately account for stock changes of feed, manure and animals. In contrast, MINAS was profitable for farmers who benefited from the increased supply of manure from pig farmers who paid them bonuses to accept their manure. MINAS did not provide many incentives to arable farms, because levy-free surpluses were relatively high.

In 2003, the CJEU ruled that MINAS failed to comply with the requirements of the EUND on principle grounds; MINAS did not enforce the application standard of 170 kg N ha^{-1} yr^{-1} in manure or crop-specific application standards. Further, the MINAS farm gate balance could only be established after the growing season, when nitrate pollution may have occurred already, and farmers could in principle buy off exceedance by paying the levy (Wright & Mallia 2008). By 2003 the support of farmers organisations and political parties for MINAS had in part eroded due to increasing complexity, high administrative costs for farmers and taxpayers, and the malfunctioning of intensive livestock farms.

The new system with manure and N application standards complied with the maximum allowable application rate of manure (170 kg N ha^{-1} yr^{-1} or 250 kg N ha^{-1} yr^{-1} in case of derogation) stipulated in the EUND did not bring much relief to the farmers. However, it contributed to further increase in NUE from 40% to 50%, and to achieve, on average, the nitrate target of 50 mg l^{-1} in the sandy region (van Grinsven et al. 2016). The increase of NUE after 2006, when MINAS was abolished, is due to a decrease of use synthetic N fertilizer by nearly one third (from 270 to 200 million kg N). But the need for a flexible and farm-specific nutrient accounting system remained.

Driven by recent impulses for resource efficiency and circularity, and facilitated by improved availability of N budget data from feed and dairy industry and data processing tools, a MINAS-like system was introduced in the form of the KringloopWijzer (Aarts et al. 2015), which is now enforced on almost all dairy farms by the dairy cooperative Friesland-Campina, which sources ~80% of all raw milk in the EU. The KringloopWijzer is a management tool, based in part on a farmgate balance approach. It allows for the quantification of N and P excretions, ammonia and greenhouse gas emissions, and stimulates increased NUE at farm, herd and soil/crop levels. It appears that the time was not right for a regulatory instrument based on farmgate balances at the turn of the millennium, but that could change in the near future.

Evaluation of the DPSIR approach for the Netherlands

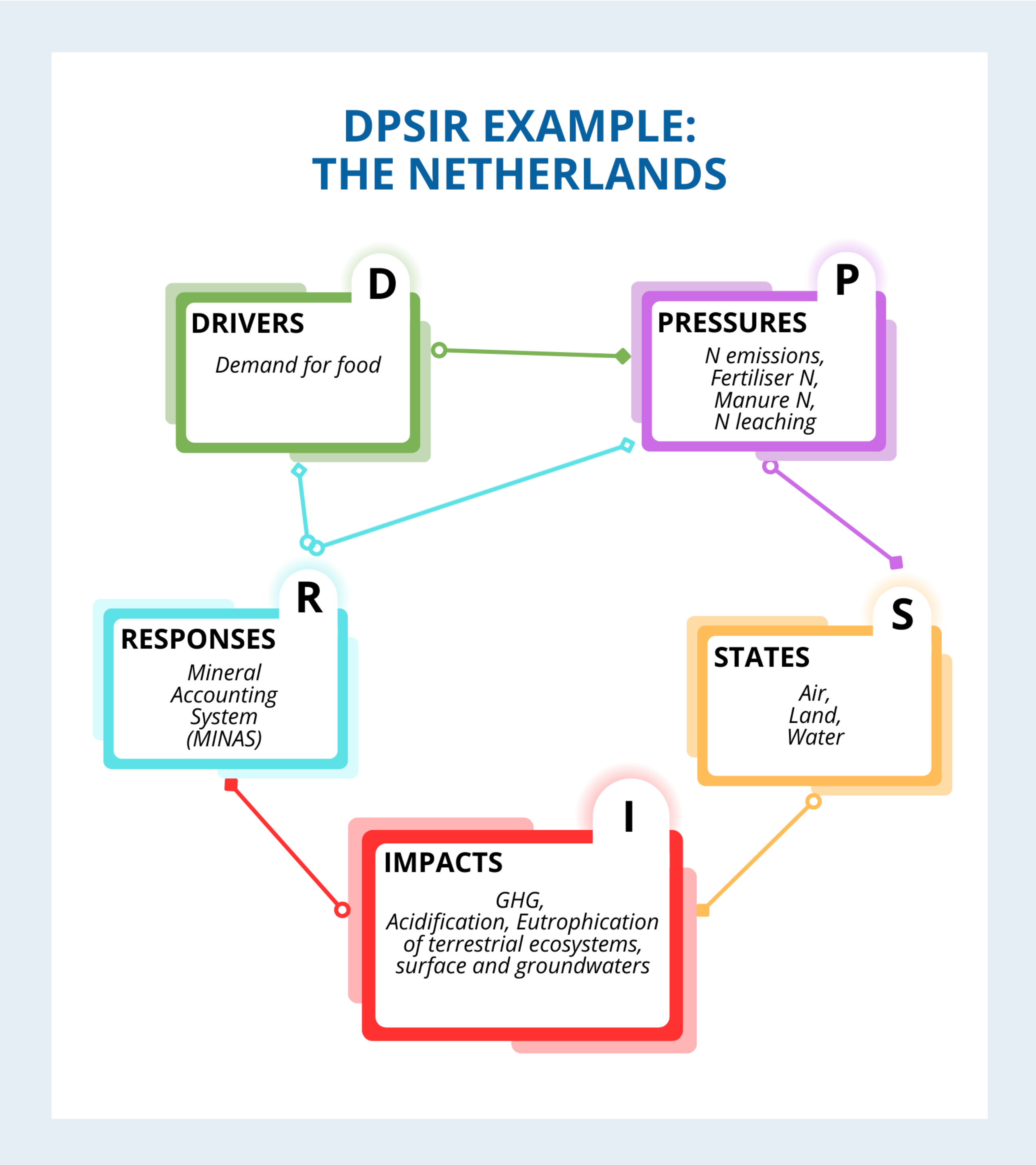


Figure 5.6. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for the Mineral Accounting System (MINAS) in the Netherlands. Nitrogen (N), greenhouse gas (GHG). Original graphic produced for this document © UKCEH 2025

Fertilizer policies in the Netherlands are driven by environmental concerns, and benefit from quantification of N flows and scientific expertise. The DPSIR diagram does not show evolution over time, but responses have changed, not least because of economic and social demands related to fairness of the policies. The recent application, KringoopWijzer, may help to increase NUE.

5.5 Aotearoa New Zealand's success and challenges with nitrogen cap and trade

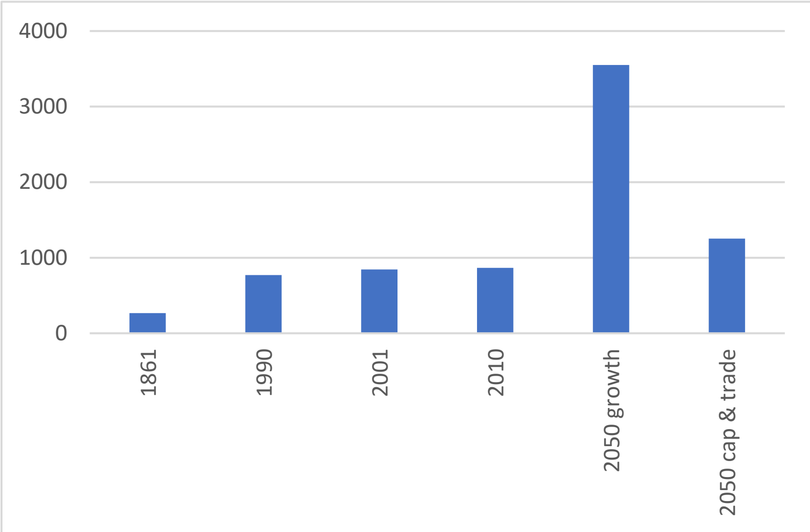
Aotearoa New Zealand has tested innovative N management policies within a remote land area slightly greater in size than the United Kingdom in the southern hemisphere's mid-latitude temperate zone. Productive agriculture comprises approximately 12 of 26.7 million hectares, and is dominated by year-round dairy, sheep and beef grazing systems that depended almost entirely on biological N fixation into the 1980s. A shift from sheep to dairy since that time has led to agricultural intensification, with increasing farm level N inputs and outputs (Figure 5.7). Since 1980, there have been notable increases in the use of N fertilizer, imported feed, commodity exports (Figure 5.8), the concentration of animal wastes and human population (Parfitt et al. 2006, 2012).

Figure 5.7. New Zealand's grass-fed pastures have intensified and shifted from sheep to dairy production over the last three decades. Photograph by Troy Baisden.

Agricultural N management issues present a focus area for Aotearoa New Zealand's environmental governance systems. These systems operate independently through local democracy in 16 regions that align local environmental legislation with catchment boundaries (Parfitt et al. 2012), implementing national policy guidance. Development of regulations and actions has targeted the most sensitive catchments, where the intrinsic value of water ecosystems and the economic value of tourism are greatest. The period between 2006–2008 saw the development of both a national N budget methodology and the world's first catchment-based, N cap-and-trade schemes in the catchments of two lakes, Taupō and Rotorua. The national and regional business-as-usual versus cap-and-trade scenarios signaled the potential to prevent an intensification of N inputs and outputs of more than 3-fold between 2001 and 2050 (Parfitt et al. 2012).

The case studies of Lakes Taupō and Rotorua, which are highly N-sensitive due to P-rich volcanic catchments, now illustrate both the success and challenges of using cap-and-trade policy tools. Good historic monitoring data over 20 years detected worsening trends, leading to the evaluation of N budgets and related science (Rutherford et al. 2019; Vant 2013; OECD 2015; Donald et al. 2019). The first important milestones notified the public of policy development that would benchmark their N budgets according to their practices in 2001. This enabled the worsening of N budgets and water quality to be halted, by comparison to the adjacent landscapes between the Taupō and Rotorua where agricultural development and intensification of N cycling continued, echoing national trends.

Figure 5.8. Aotearoa New Zealand's nitrogen exports (Gg N y^{-1}). Estimates: 1861 & 1850 (Parfitt et al. 2006); 1990, 2001 & 2010 (Parfitt et al. 2012) Original graphic produced for this document © UKCEH 2025.



The Lake Taupō catchment provides a relatively simple example of success with cap-and-trade. As Aotearoa New Zealand's largest lake (612 km^2), its ultra-oligotrophic status contributes to stunning water quality with a typical visual (Secchi) depth of 15 m (Verburg & Albert 2019). The legislation was finalised in 2006, confirmed following court challenges in 2008, and went into effect in 2010, implementing a catchment-wide reduction in N reaching the lake by 20%. Trading within a customised market mainly occurred at the onset of the scheme, allowing landowners a full array of choices ranging from intensification to reforestation. The system relied on an agricultural nutrient budget model to place N limits on N losses from individual farms (PCE 2018), which summed to a catchment-wide cap to achieve a 20% reduction in accounted N reaching the lake after including land-use change to forests. The concept of the N cascade (Galloway et al. 2008) proved compelling for simplifying policy to reflect inputs of new N to the catchment (Environment Court of New Zealand 2008). The 20% reduction was included to recognise that the catchment system had already overshot the selected policy target of 2001 lake water quality, due to legacies calculated as a 'load-to-come' based on decadal and centennial lag-times in groundwater N delivery.

The effort to achieve a 42% N reduction by 2032 in Lake Rotorua, an 80 km^2 eutrophic lake, demonstrates the increased challenges posed by additional complexity, including more intensive farming, wastewater from a significant population (Donald et al. 2019), and groundwater lag times that are greater and more complex than those in Lake Taupō (Morgenstern et al. 2015). Efforts to include science-based rules to account for groundwater lag times in the proposed cap-and-trade scheme improved modelled outcomes for the lake only slightly, while creating processes and rules that stakeholders found too complex (Anastasiadis et al. 2014).

Additional challenges identified in Lake Rotorua included changes in the state of the lake system and perceptions resulting from successful geo-engineering (including alum dosing to remove P) that were implemented rapidly knowing that reaping the benefits from a catchment-wide N cap would take decades (Rutherford et al. 2019; Donald et al. 2019). Perhaps the largest unresolved challenge to this and other catchments has been the allocation of units within the cap. Ongoing economic activity favours allocation that is proportional to N losses under current land use, rewarding farmers that have already invested in development beyond environmental limits. The finalisation of Lake Rotorua's cap-and-trade scheme has been delayed by repeated efforts to improve equity for owners of less-developed land, so that they retain a right to intensify their land use or sell unused development rights. This issue is repeated throughout Aotearoa New Zealand where considerable land and environmental governance responsibility has been returned to the Indigenous Māori people via a treaty settlement process in recent decades. The returned land commonly remained less developed agriculturally and economically than comparable areas (Waitangi Tribunal 2019), including significant land in the Lake Rotorua catchment (PCO 2006). A lesson has been that models and uncertainty developed mainly to define policies to reduce N loads may lack the detail required to compare different allocation schemes, including issues of fairness and equity (Daigneault et al. 2017).

Nationally, evidence for declining water quality (NZME 2022) has led to caps in some regions and in large river catchments. Some of the 16 regions have used natural capital (Dominati et al. 2010) as a basis for implementation of caps, with limited or no trading. The latest version of evolving national policy (NPS-FM 2020) strengthens requirements but maintains concentration-based limits on N at monitored sites and has been criticised for lacking protection for estuaries as downstream receiving environments (PCE 2020).

Drawing on Aotearoa New Zealand's experience implementing N cap-and-trade and other policy approaches over the past two decades, a tentative lesson can be drawn that local innovation has been possible due to having local environmental governance following catchment boundaries. Despite this, cap-and-trade has only one fully successful example, has not provided a widespread solution, and has been contentious and complex to implement. Where progress managing N has occurred, N budgets that provide projections decades ahead and allow the allocation of reductions in reactive N inputs and exports, appear to have supported effective policy and strategy.

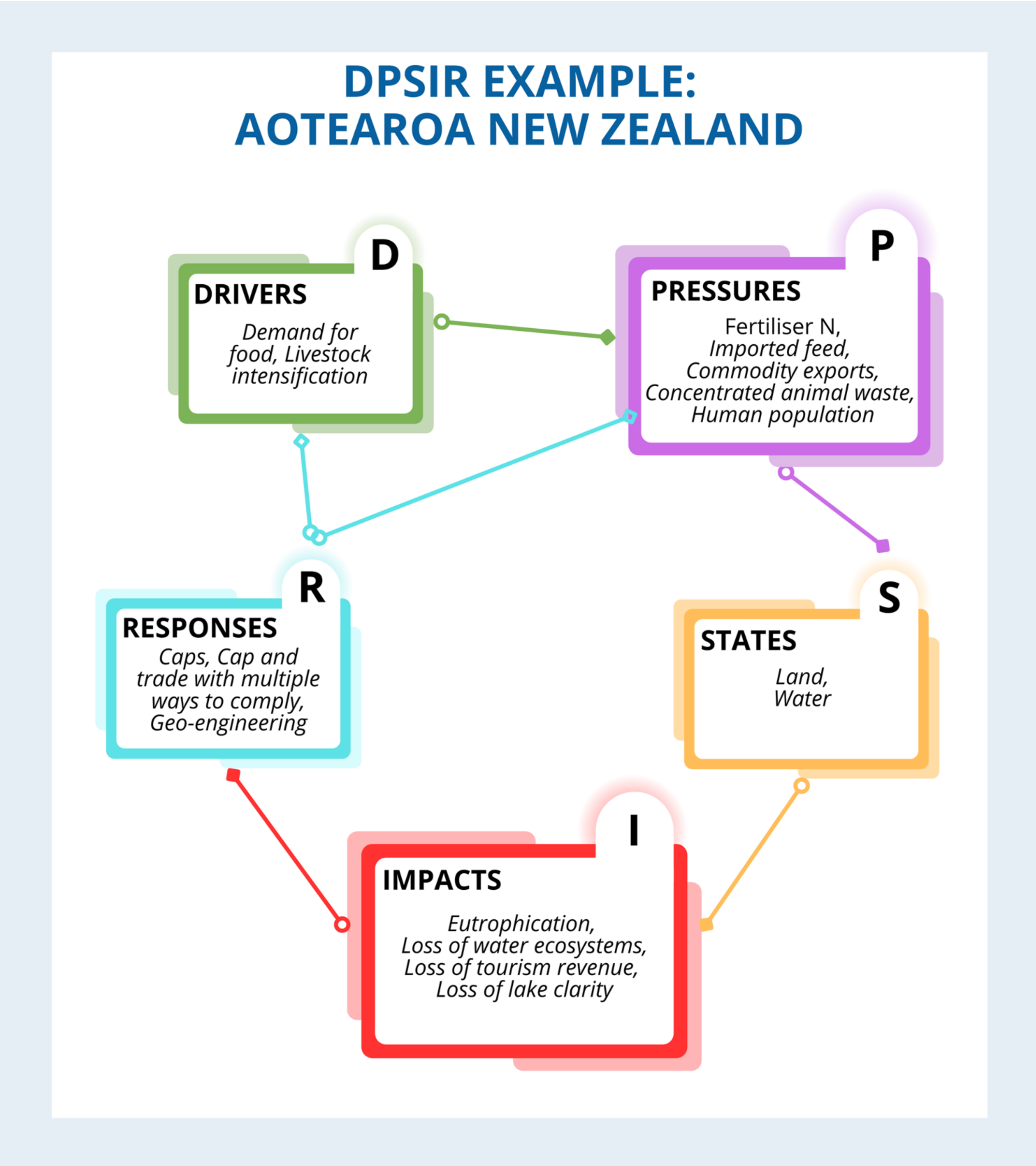


Figure 5.9. Drivers, Pressures, States, Impacts, and Reponses (DPSIR) diagram for the cap-and-trade system for lakes Taupo and Rotorua in Aotearoa New Zealand. Original graphic produced for this document © UKCEH 2025

This example illustrates the difficulties of implementing effective policies, even when the issues are similar (lake eutrophication) and local voices are heard and respected. Maintaining rigorous monitoring, interpretation and forecasting becomes ever more important to provide evidence when and where it is needed.

5.6 Water quality standards and mitigation approaches within international marine governance frameworks

The requirement to achieve and maintain healthy, productive and biologically diverse seas in the face of substantial anthropogenic terrestrial inputs of pollutants, including reactive N compounds, has led to a multitude of marine governance frameworks. The transboundary nature of the problem means these often take the form of multinational conventions (Table 5.1). These define monitoring and assessment approaches, which are then enacted at a national or regional level. Many of these conventions fit within the UNEP regional seas programmes and support UN Sustainable Development Goal 14 objectives:

- 'Conserve and sustainably use the oceans, seas and marine resources for sustainable development'
    - 14.1: 'By 2025, prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including marine debris and nutrient pollution'.

The process of monitoring and assessing eutrophication (nutrient pollution) is highly complex. The environmental response to a particular nutrient pollution input is dependent on many interrelated hydrodynamic, biochemical and geographic factors

- High costs of (and hence lack of) measurements
- Lack of information of a baseline ("pristine") state, which in many cases pre-dates the invention of suitable measurement approaches; and
- The high degree of natural variability.

This complexity has led to a lack of internationally accepted marine water quality standards, and subjective terms such as 'undesirable disturbance' are employed instead.

Table 5.1. Examples of marine governance frameworks.

Regions	Organisation: Countries/Region	Example Output	Reference
Europe	Oslo-Paris Commission (OSPAR): 15 countries in NE Atlantic	Third OSPAR Integrated Report on the Eutrophication Status of the OSPAR Maritime Area, 2006-2014: (2017) HASEC17/D505	oap.ospar.org
	Baltic Marine Environment Protection Commission–- Helsinki Commission (HELCOM): 9 Countries	HELCOM Baltic Sea Action Plan (2007)	helcom.fi
	Black Sea Commission: 6 Countries	Strategic Action Plan for the Rehabilitation and Protection of the Black Sea (2009)	https://www.unep.org/bucharest-convention
	Mediterranean Action Plan–- Barcelona Convention: 22 Countries	A Strategic Action Plan to eliminate pollution from land-based sources in the Mediterranean: MEDPOL Programme (2006)	unep.org/unepmap
Asia	Water Environment Partnership in Asia: 13 Countries in E and SE Asia	WEPA Outlook on Water Environmental Management in Asia (2015)	wepa-db.net
	Association of Southeast Asian Nations: 10 Countries	ASEAN Marine water quality Management Guidelines and Monitoring Manual (2008)	environment.asean.org

Regions	Organisation: Countries/Region	Example Output	Reference
	Partnerships in Environmental Management for the Seas of East Asia (PEMSEA): 14 Countries	SDS-SEA Implementation Plan 2018-2022 (2018)	https://www.pemsea.org/
Americas	Gulf of Mexico Programme: USA	Annual Report for the Gulf of Mexico Programme (2018)	epa.gov/gulfofmexico
	The Convention for the Protection and Development of the Marine Environment in the Wider Caribbean Region (WCR); Cartagena Convention: 20 countries	Protocol Concerning Pollution from Land-Based Sources and Activities (1999)	www.unep.org/cep/who-we-are/cartagena-convention
	International Joint Commission between the USA and Canada	Great Lakes Water Quality Agreement	https://www.ijc.org/en/what/glwqa-ijc

The assessment to inform governance frameworks is through monitoring programmes using a diverse range of approaches (Karydis & Kitsiou 2013). The range of parameters and spatiotemporal coverage are limited by practical considerations including the availability of sampling platforms (research ships, moorings) and measurement approaches, although both are developing rapidly through autonomous sampling and miniaturisation technologies. Generally, the parameters include temperature, salinity, water transparency, dissolved oxygen, chlorophyll and various forms of nutrients. Monitoring biological parameters (phyto- and zooplankton species and benthos) is highly desirable, but complex and expensive. The monitoring information is then used in multivariant analysis to evaluate indicators of water quality. Water quality standards are generally defined and applied at a national level, and so there is a great diversity of approaches, (e.g., Ferreira et al. 2011).

The only practical management approach to mitigate coastal eutrophication at anything but a very local scale is through the control of nutrient inputs, both direct and via rivers and estuaries. Such inputs come from wastewater, industrial and agricultural sources. Mitigation measures include modifying agricultural and industrial practices, sewage treatment and discharge approaches. Taking the Oslo Paris Commission (OSPAR) as an example of a governance framework, this commission publishes an annual report: 'Riverine Inputs and Direct Discharges to Convention Waters'. This contributes to the Joint Assessment and Monitoring Programme, and this, in turn, to regular assessment reports (full report in 2010, an Intermediate Assessment in 2017). The OSPAR Comprehensive Procedure summarizes this information into simple colour-coded maps indicating if an assessment region is a “Problem Area”; “Potential Problem Area” or “Non-problem Area”; underlying this is a wealth of supporting and descriptive information. This information is used in a six-year cycle to inform each nation’s assessment of compliance with the Marine Strategy Framework Directive, a legislative framework that aims to achieve “Good Environmental Status (GES) of EU marine waters by 2020”. This includes a descriptor on eutrophication. Ultimately, compliance or non-compliance is under the scrutiny of the Court of Justice of the European Union.

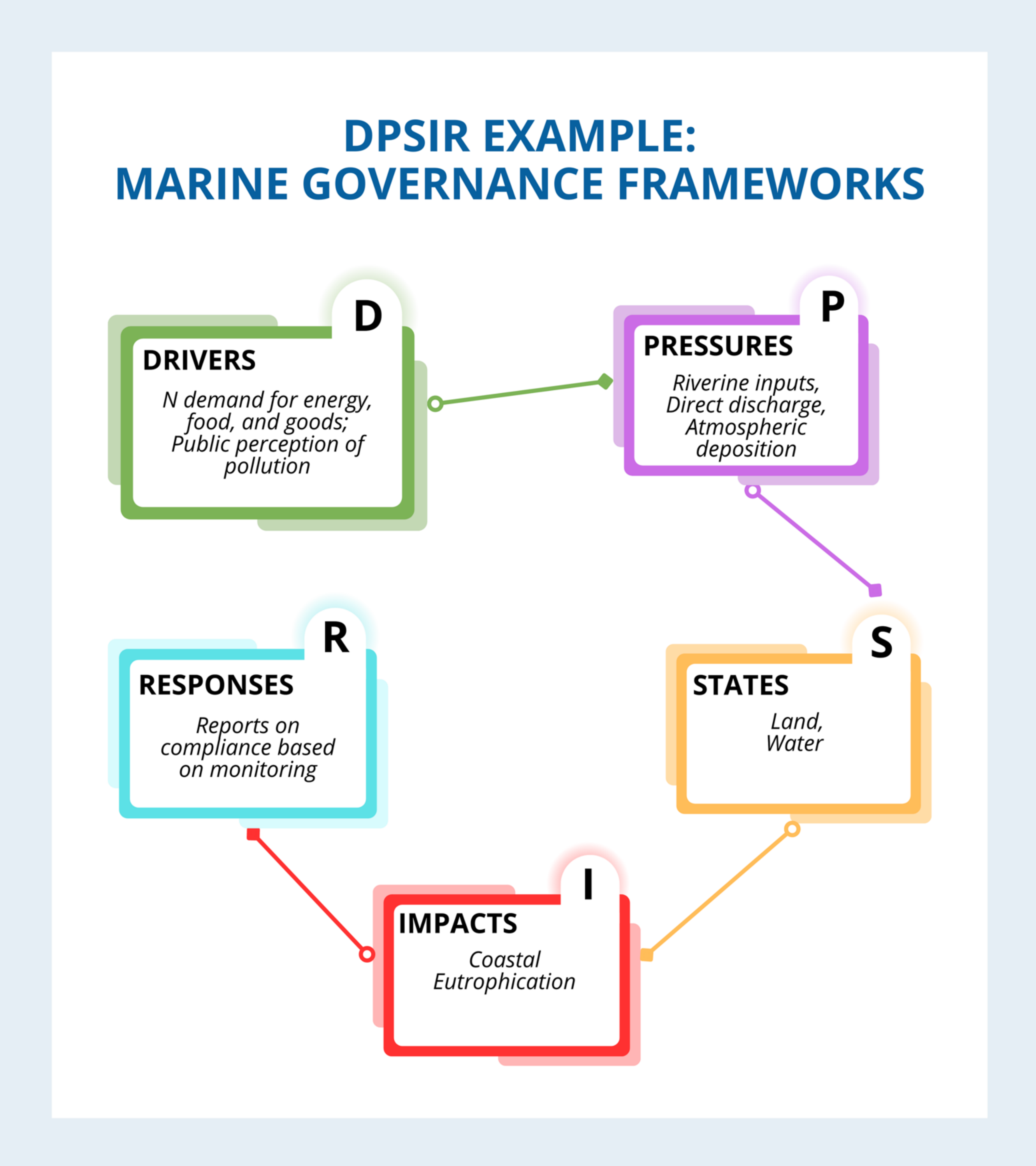


Figure 5.10. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for the marine governance frameworks worldwide. Original graphic produced for this document © UKCEH 2025

The many marine governance frameworks provide valuable services. However, these do not necessarily lead to action to manage drivers. This is not to criticize any individual marine governance framework, but to emphasize a) that more detail review of each framework is needed, and b) that fair assessment should seek to demonstrate how the frameworks can move beyond reporting to implementing practice changes that reduce nitrogen pollution substantially.

5.7 International air pollution controls via the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP)

The Convention on Long-range Transboundary Air Pollution (CLRTAP) was adopted by national parties across region of the United Nations Economic Commission for Europe (UNECE) in 1979. It is a framework for mitigating air pollution that has advanced into binding obligations through several protocols over the years. The framework addresses transboundary (i.e., cross-border) pollution and builds on cooperation, linking member states and involving scientists, to conduct common monitoring and evaluation methods (Kauffmann & Saffirio 2020), and inform pollution strategy development. Scientific input comes primarily via the Working Group on Effects (WGE) and the European Monitoring and Evaluation Programme (EMEP) to provide understanding of the harmful effects of air pollution to support effective air pollution control. The WGE includes six International Cooperative Programmes (ICPs) on Forests, Waters, Materials, Vegetation, Integrated Monitoring, Modelling and Mapping, and the Task Force on Health identify the most endangered areas, ecosystems, and other receptors by considering damage to human health, terrestrial and aquatic ecosystems and materials. An important part of this work is long-term monitoring. The work is underpinned by scientific research on dose-response, critical loads (CLs) and levels, and damage evaluation via integrated assessment models. The work of EMEP includes atmospheric monitoring and modeling of transboundary fluxes and levels, as well as the Task Force on Integrated Assessment Modelling.

The main strategy development and negotiating body of the convention, the Working Group on Strategies and Review (WGSR) also includes significant scientific input, provided especially through the Task Force on Reactive Nitrogen (TFRN), the Task Force on Technical and Economic Issues (TFTEI) and the Forum for International Cooperation on Air Pollution (FICAP). Whereas TFRN (established in 2007) focuses on developing a holistic approach to nitrogen pollution mitigation, with special emphasis on agricultural ammonia, TFTEI focuses on combustion and other industrial sources, especially of nitrogen oxides.

In 1999, air pollution impacts of both nitrogen oxides and ammonia were specifically included as part of the "Gothenburg Protocol to Abate Acidification, Eutrophication, and Ground-Level Ozone" (UNECE 1999)

An example of how the Air Convention works comes from the International Cooperative Programme on Modelling and Mapping of Critical Levels and Loads and Air Pollution Effects, Risks and Trends (ICP Modelling and Mapping, https://unece.org/modelling-and-mapping) which was established in 1988 to:

- Determine receptor-specific critical loads for indirect effects of the long-term deposition of various air pollutants and critical levels for direct effects of gaseous air pollutants,
- Map pollutant depositions and concentrations which exceed critical thresholds,
- Establish appropriate methods as a basis for assessing potential damage, e.g., via dynamic modeling.

About 30 nations contribute data that are used to develop CL maps as input to European Union air quality policy. Critical loads and maps are reviewed and updated about every 5 years, taking information from many publications ranging from the open literature to reports and dissertations. Data from European studies take precedence unless absent, in which case results from other countries are considered. The Mapping Manual of the Convention defines specific methods in detail (https://www.umweltbundesamt.de/en/cce-manual; CLRTAP 2014, 2017). Causality is determined primarily from the results of long-term field addition experiments, deposition gradient studies, and steady-state and dynamic models. Other than excluding specific approaches (e.g., unrealistically high pollutant additions, microcosm experiments), no other criteria for study quality are specified. Critical loads are ecosystem specific, while critical levels for nitrogen refer to major vegetation groups (see Chapters 3 and 4 of the present guidance document). Reviews and updates are drafted by a small team; the drafts become background documents for meetings of experts who develop final documents. Consensus on the CLs and maps is not required but often occurs. The exceedances of CLs and critical levels are used to quantify the risk for environmental damage by air pollution and to support the development of optimised abatement strategies (Gregor et al. 2001).

An effectiveness assessment of the CLRTAP regime indicates that it has helped states to reach agreement on contentious issues and achieve results in air pollution reduction. The scientific tools developed under the convention have been instrumental in the development of EU air pollution policies, including the development of the Clean Air For Europe (CAFÉ) programme, the Thematic Strategy for Air Pollution, and the establishment of national air pollution emissions ceilings (Kauffmann & Saffirio 2020). There are continuing challenges with regard to participation, implementation procedures, empowerment of domestic stakeholders and funding. However, the CLRTAP and its collaborative methods are being used as a model for other international regions, including northeast Asia: China, Korea, and Japan (Byrne 2015; Kauffmann & Saffirio 2020).

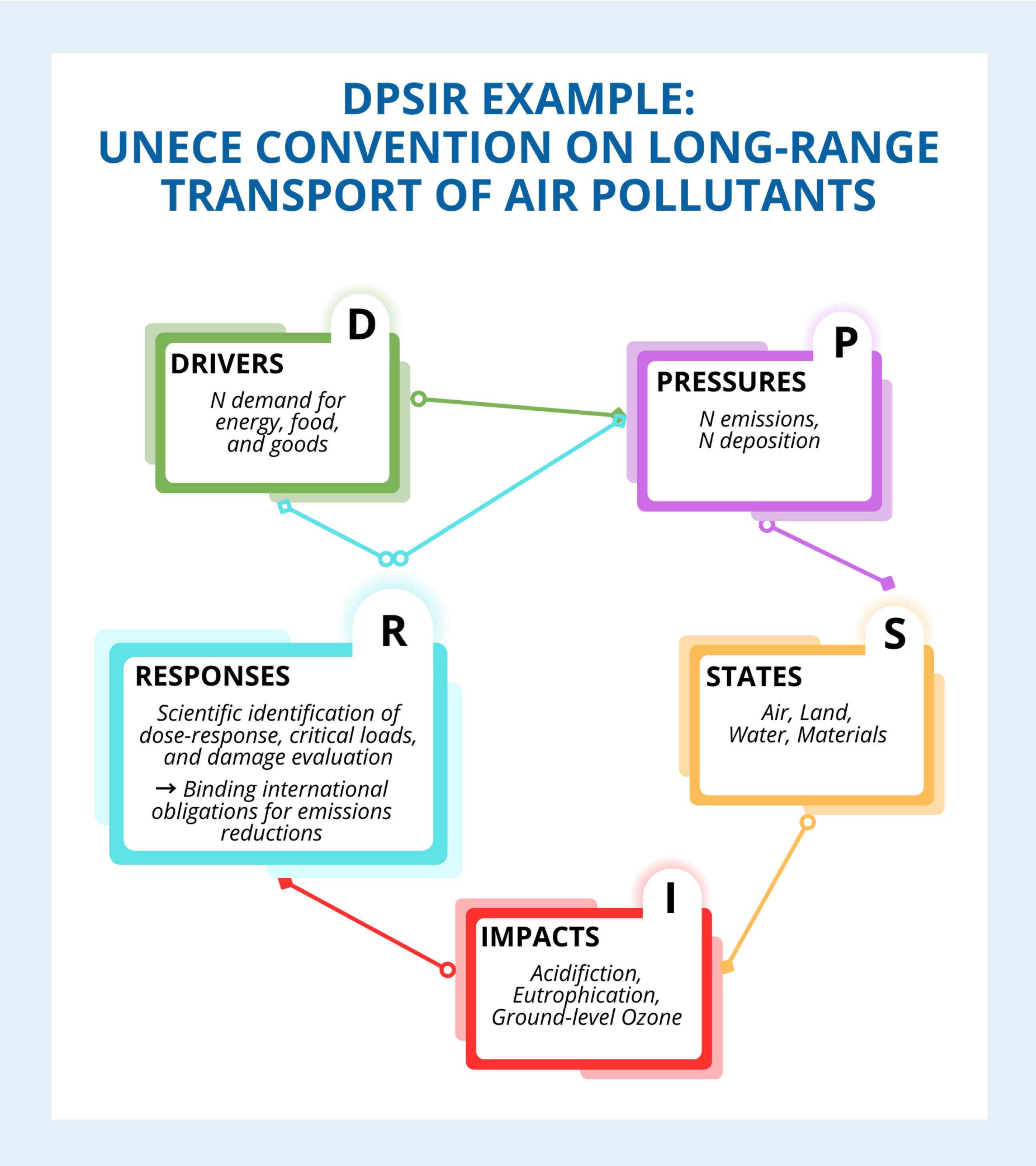


Figure 5.11. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP), informally the 'Air Convention'. The DPSIR process is successful in identifying drivers of change, i.e., nitrogen (N) demand), monitoring sources and consequences of change, and evaluating changes against common benchmarks, which led to establishment and enforcement of national air pollution emission ceilings. Original graphic produced for this document © UKCEH 2025

5.8 Policies to improve nitrogen pollution in water in Taiwan

Taiwan's fresh and coastal waters are threatened by reactive nitrogen (N) from human activities. Taiwan has a population of 23.5 million, a population density up to 651 persons per km^2, and a growing economy: the gross domestic product (GDP) per capita increased by 43% from 2004 to 2013. Vigorous development of the Taiwan livestock industry in the past 40 years has polluted water with N. Additional N comes from tourism (e.g., tourist hotels), coal-fired power plants and industries, including electroplating and leather manufacturing.

The major sources of N pollution are domestic sewage, animal husbandry, agriculture and industry. In the past five decades, water pollution prevention and control regulations were focused on the control of organic pollutants and other specific toxic chemical substances from industry. However, agricultural intensification has also contributed to N pollution, and with increasing urbanisation, domestic sewage has become the primary source of N pollution in urban catchments. Water quality management measures to control N began to be enforced in 2014.

River water quality has been improving over time through a combination of science-based policies across agricultural, sewage treatment and industrial sectors, and through enforcement actions by the government of Taiwan beginning in 1974. Water pollution control measures for pig farms were put in place after publication of a paper showing nearly substantial N pollution in rivers was caused by livestock (Kao et al. 2003) Liu 2009). Between 2010 and 2019 the numbers of pig farms were reduced from 10,000 to 6,700 farms, and the total number of husbandry pigs declined approximately 10%. Beginning in 1998 the Taiwan Council of Agriculture began to promote proper fertilizer application methods after becoming aware that overuse of synthetic N fertilizer was impacting environmental quality in Taiwanese rivers and estuaries. The implementation of the system of 'reasonable fertilizer application guidelines' was supported by Taiwan's Council of Agriculture.

According to statistics, the average annual synthetic N fertilizer consumption from 2005 to 2007 was 1.14 MMT. From 2008 to 2013, the average yearly consumption was lower at 1.02 Mtonne, and the annual average decreased by 120,000 Mtonne. Total N fertilizer consumption dropped in 2014 to 972,000 Mtonne, showing significant decreases in fertilizer use from 2008 to 2014 (Chou et al. 2016). The Environmental Protection Administration of Taiwan (EPAT) proposed a series of measures to improve river pollution after they estimated that the effectiveness of strict control of wastewater discharge standards on main rivers could reduce NH_3-N use by 976 kg per day. These measures accelerated the construction of sewer systems, domestic sewage treatment plants, and artificial wetlands from 2014 to 2015. Effluent standards for NH_3-N and total N were revised in 2017 to improve water quality from industry. It is expected that the strict control of wastewater effluent from industry will reduce NH_3-N use by 30,108 kg per day (EPAT 2020).

Analysis of Taiwan's legislative history of water pollution prevention policies and regulations can be separated into three stages. The first stage started with the Water Pollution Control Act in 1974 that focused on organic pollution management. The second stage began in 2002 and involved amendment of water pollution regulations for toxic substance control. Ammonia N management, which can be the critical factor in reducing water pollution, did not begin until the third stage in 2014. The EPAT's River Pollution Index (RPI) is used to demonstrate the condition of water quality, and includes four water quality measurements: Dissolved oxygen (DO), biological oxygen demand (BOD), suspended solids (SS), and NH_3-N. The RPI is used to classify river water quality into four pollution categories: Severely polluted, moderately polluted, slightly polluted and non-polluted.

The percentage of severely polluted river kilometres (km) dropped significantly after 2003, as there moved into the moderately polluted category (Figure 5.12). The concentration of NH_3-N decreased gradually during the second stage, but not enough to increase the portion of the non-polluted length. The amended Water Pollution Control Act improved organic pollution during the second stage, but not the N threat. The average concentration of NH_3-N decreased after 2014 when the third stage of amended water pollution related regulation and policies was implemented. Since then, the percentage of non-polluted river lengths has increased significantly. The river length in the severely polluted category dropped 12% over 17 years. Nitrogen pollution management strategies with laws and regulations effectively improved the water quality by reducing the N threat to the aquatic system (Figure 5.12). Water quality control measures continue to be implemented: In 2020, the EPAT launched the Sustainable Water Quality Promotion Plan (2020-2023) to reduce NH_3-N. The objective of the project is to protect aquatic ecosystems and begin to work toward sustainability goals.

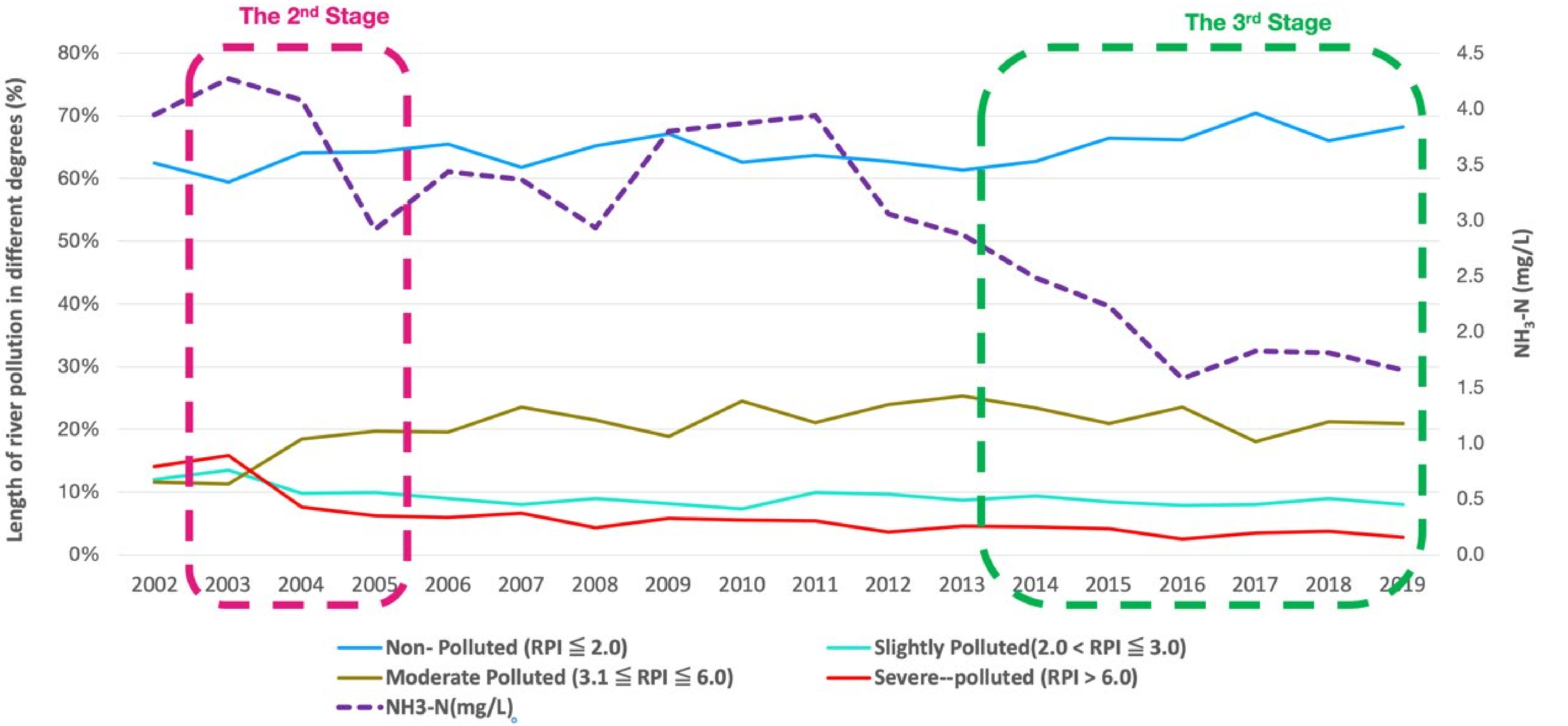


Figure 5.12. Trends in levels of the River Pollution Index (RPI) for Taiwan during stage two and three water pollution control periods, as well as the percentage of the length of rivers in the pollution categories: Non-polluted, slightly polluted, moderately polluted and severely polluted. Original graphic produced for this document © UKCEH 2025.

Evaluation of the DPSIR approach for water quality in Taiwan:

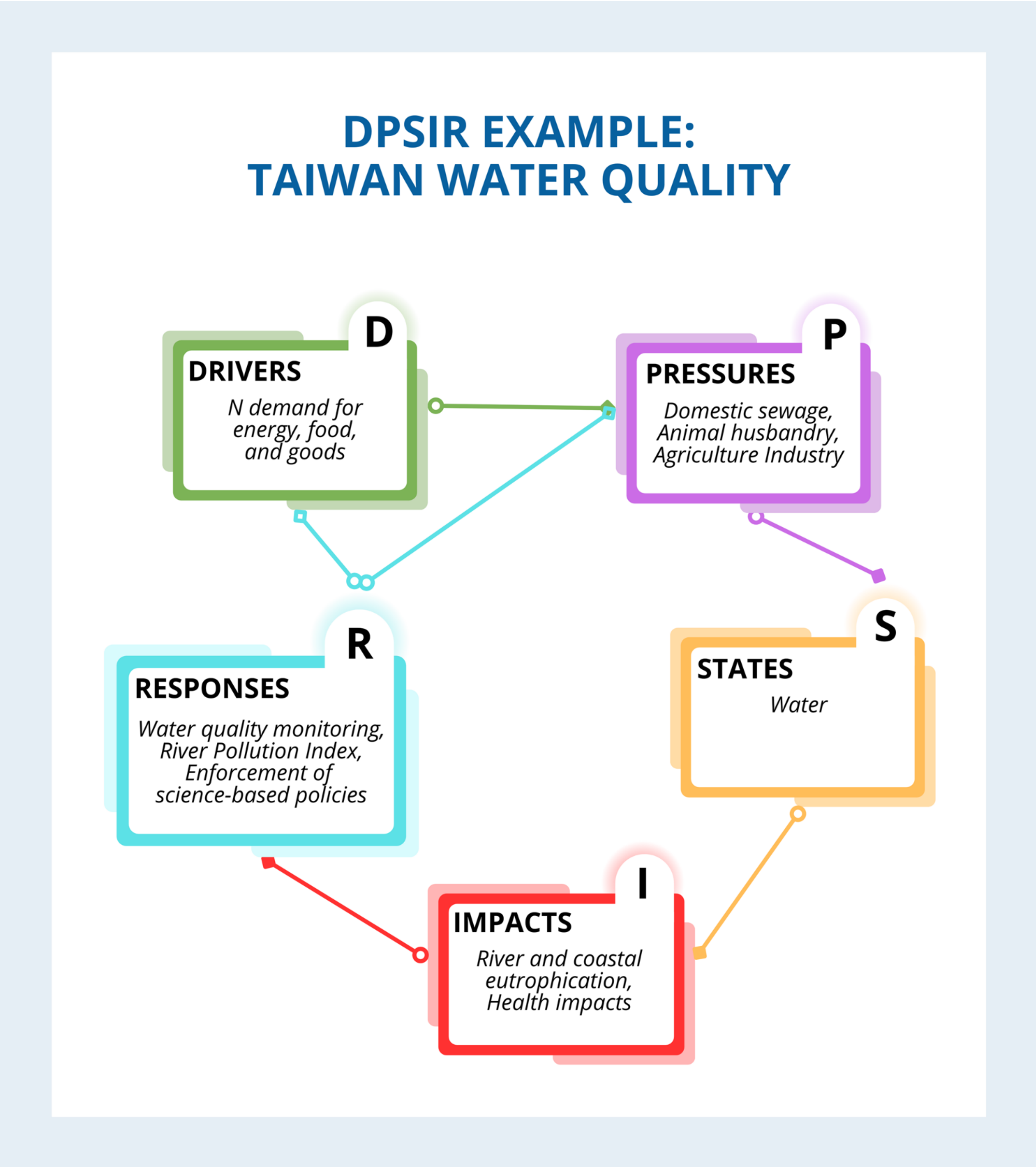


Figure 5.13. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for nitrogen (N) demand and water quality in Taiwan. Original graphic produced for this document © UKCEH 2025.

Taiwan is a successful example DPSIR can lead to establishment and enforcement of national water pollution controls by:

- Identifying drivers of change through a sequence of increasingly specific studies,
- Monitoring sources and consequences of change, and
- Evaluating changes against common benchmarks.

5.9 Rocky Mountain National Park Nitrogen Deposition Reduction Plan

Rocky Mountain National Park (RMNP) in the United States protects 1,100 km^2 of spectacular high-mountain environments. The park is home to alpine and subalpine lakes, forests, wetlands, and is home to the headwaters of the Colorado and Missouri Rivers. In 1977 the U.S. Clean Air Act gave special protection to RMNP, along with other protected areas, requiring that air quality and environmental attributes not be allowed to deteriorate due to air pollution. Annual wet atmospheric N deposition is low by most countries' standards (mean 1998-2019 deposition was 3.3 kg N ha^{-1} yr^{-1}), but even this amount exceeds background values by an order of magnitude (Baron 2006). Accumulated research since 1983 found that atmospheric N deposition was responsible for significant ecological and biogeochemical impacts to soils and soil food webs, plant diversity, and lake productivity and species composition (Baron et al. 2000; Burns 2004). After external assessments of the validity of the results, and consideration of a large body of evidence, the National Park Service concluded atmospheric N deposition was damaging RMNP and needed to work with regulatory agencies to achieve their legal requirement to protect air quality related values from air pollution. A statistically determined critical load (CL) became the organizing principle for a policy response to N in RMNP (Baron 2006).

Much of the N deposition originates in the urban and agricultural regions of Colorado east of RMNP from human activities that include intensive animal and crop agriculture, transportation and fossil-fuel burning (Benedict et al. 2018). An interagency collaboration between the Colorado Department of Public Health and Environment (CDPHE), the U.S. Environmental Protection Agency (EPA) and the National Park Service produced a policy strategy aimed at reducing N emissions and subsequent N deposition in RMNP to below the level of harmful impact, or CL (CDPHE 2023). This Nitrogen Deposition Reduction Plan (NDRP) lays out the methods by which wet N deposition can be reduced to achieve the goal. Scientists and administrators from the agencies met with stakeholders representing industrial, municipal and agricultural interests to explain the scientific evidence and engage their assistance in reducing N emissions. The plan presents various methods of reducing NO_3^- deposition (mainly generated by combustion) and NH_4^+ deposition (mainly generated from agriculture) but allows N emitters flexibility in choosing best practices. The methods range from reducing power-plant emissions and retrofitting diesel-powered buses, to better management of animal waste from industrial livestock facilities. The plan uses a 'glidepath' approach to lower N emissions gradually until they stabilize below the critical load of 1.5 kg wet N ha^{-1} yr^{-1} in 2032 and remain low until 2064. Continuous monitoring of N deposition allows progress on the glidepath to be examined at five-year intervals. Adoption of N reduction measures and agricultural best management practices and innovations is currently voluntary; however, the regulatory agencies can enforce emissions reduction rules if their goal is not achieved in the expected timeframe.

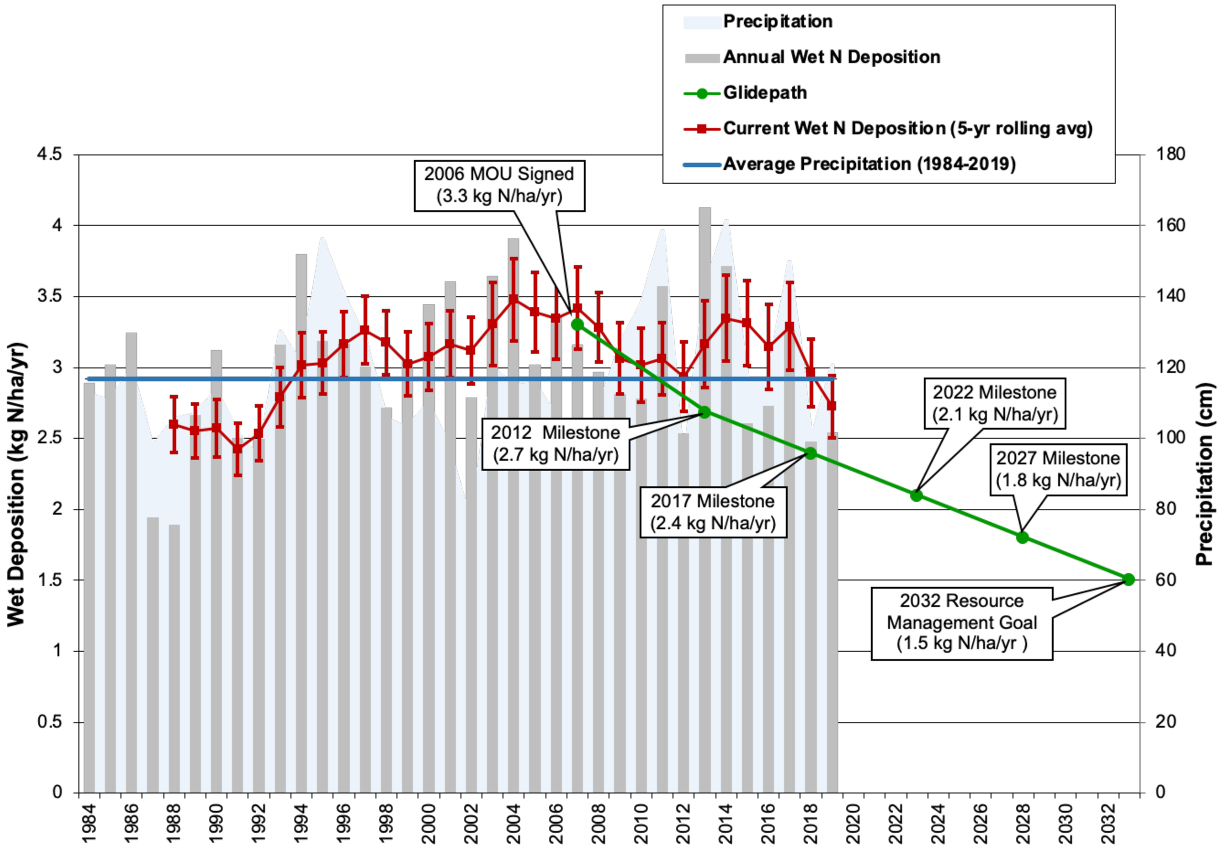


Figure 5.14. Wet nitrogen (N) deposition and precipitation at Loch Vale long-term research and monitoring watershed in Rocky Mountain National Park, compared to the Nitrogen Deposition Reduction Plan glidepath. Original graphic produced for this document © UKCEH 2025.

Prior to the NDRP's implementation in 2007, total wet N deposition had been increasing. Since 2007, the five-year rolling average of wet N deposition shows variability between wet and dry years, but no overall decline (Figure 5.14). Wet N deposition to Rocky Mountain National Park is currently above its glidepath value of 2.4 kg N ha^{-1} yr^{-1} (Morris et al. 2019). Development of agriculture and fossil fuel combustion greatly increased U.S. N emissions to the atmosphere in the second half of the 20th century, but recent regulatory actions and economic factors have lowered NO_x emissions from combustion of fossil fuels by vehicles, electric power generating units and industrial facilities, resulting in decreased wet NO_3^- deposition (Li et al. 2016). Levels of wet NH_4^+ deposition, by contrast, have increased in many regions. Reduced N as NH_4^+ is now greater than 50% of wet N deposition in RMNP (Morris et al. 2019).

The NDRP is an example of N-impacts research leading to a policy response, but the outcome is not yet known. While NO_x emissions and consequent NO_3^- deposition are expected to continue declining in Colorado and elsewhere due to an energy transition from fossil fuels toward renewable energy sources, emissions of NH_3 from agricultural sources are unregulated. Voluntary innovations in animal feed composition or manure management may help reduce emissions.

Figure 5.15. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for the Rocky Mountain National Park Nitrogen (N) Deposition Reduction Plan. The solid lines indicate known factors and implemented policies, while the dashed line indicates that the effects of responses on the pressures are not yet clear. Original graphic produced for this document © UKCEH 2025.

The DPSIR process is successful in identifying drivers of change, monitoring sources and consequences of change, and evaluating changes against common benchmarks, but voluntary pollution reduction has not reduced pressures. This example illustrates some of the complexities involved with implementation, even in a small region where nearly all stakeholders agree on an outcome.

5.10 The development and implementation of the University of Virginia Nitrogen Action Plan

The University of Virginia (UVA) is a public research university in Charlottesville, Virginia, USA, with about 22,000 enrolled students and 16,000 total faculty and staff. In 2010, UVA became the first institution to calculate its N footprint (Figure 5.16), and in 2013, the UVA Board of Visitors passed a resolution to reduce University-wide N emissions 25% below 2010 levels by 2025. In December 2019, a new goal was established to reduce solid and liquid waste to 30% below 2010 levels by 2030. Nitrogen waste from UVA contributes to N pollution in Chesapeake Bay and is one of the great environmental concerns and interests in UVA.

The UVA Nitrogen Working Group (NWG) was formed in 2014 with the goal of tracking and reducing the N footprint to reach various reduction goals. The UVA N footprint was calculated by tabulating all the N that:

- Entered the institution (e.g., food purchases)
- Was generated by activities at the institution (e.g., fossil fuel combustion in steam generators and buses)
- Was generated due to activities at the institution (e.g., commuting, food production)

Over the years, the NWG has worked with stakeholders throughout the university to collect data for tracking the footprint and develop feasible reduction goals. The N footprint tool, developed by Leach et al. (2012) was used to test scenarios for the most effective ways to decrease the N footprint of the university. Assistance from many stakeholders was crucial in publishing the first UVA Nitrogen Action Plan (NAP) in 2019 (UVA 2019). The strategies in the NAP have been incrementally implemented at UVA. Several energy strategies have been implemented at the university by facilities management including a large impact scenario with the purchase of two solar fields to reduce UVA's dependence on fossil fuels. Several dining strategies have been implemented as well, including transitioning a meat-focused café to a plant-based café and adopting a blended burger (80% beef, 20% mushrooms) in residential dining halls. These strategies were adopted by stakeholders due, in part, to the environmental benefits suggested in the NAP.

The N emissions from the energy and wastewater sectors have decreased since 2010 due to less purchased electricity and coal consumption and the City of Charlottesville adopting tertiary wastewater treatment. There have been efforts to decrease N emissions from the food sector as well. The full long-term effects of the scenarios listed in the NAP have yet to be analyzed, as the goal year of 2025 has not been reached. However, the N footprint tracking calculation completed for 2017 shows a 24% reduction in UVA's footprint since 2010 in its N footprint relative to 2010 baseline levels. Most of the decrease from 2010 to 2017 was driven by a reduction in the food and energy sectors (Figure 5.16).

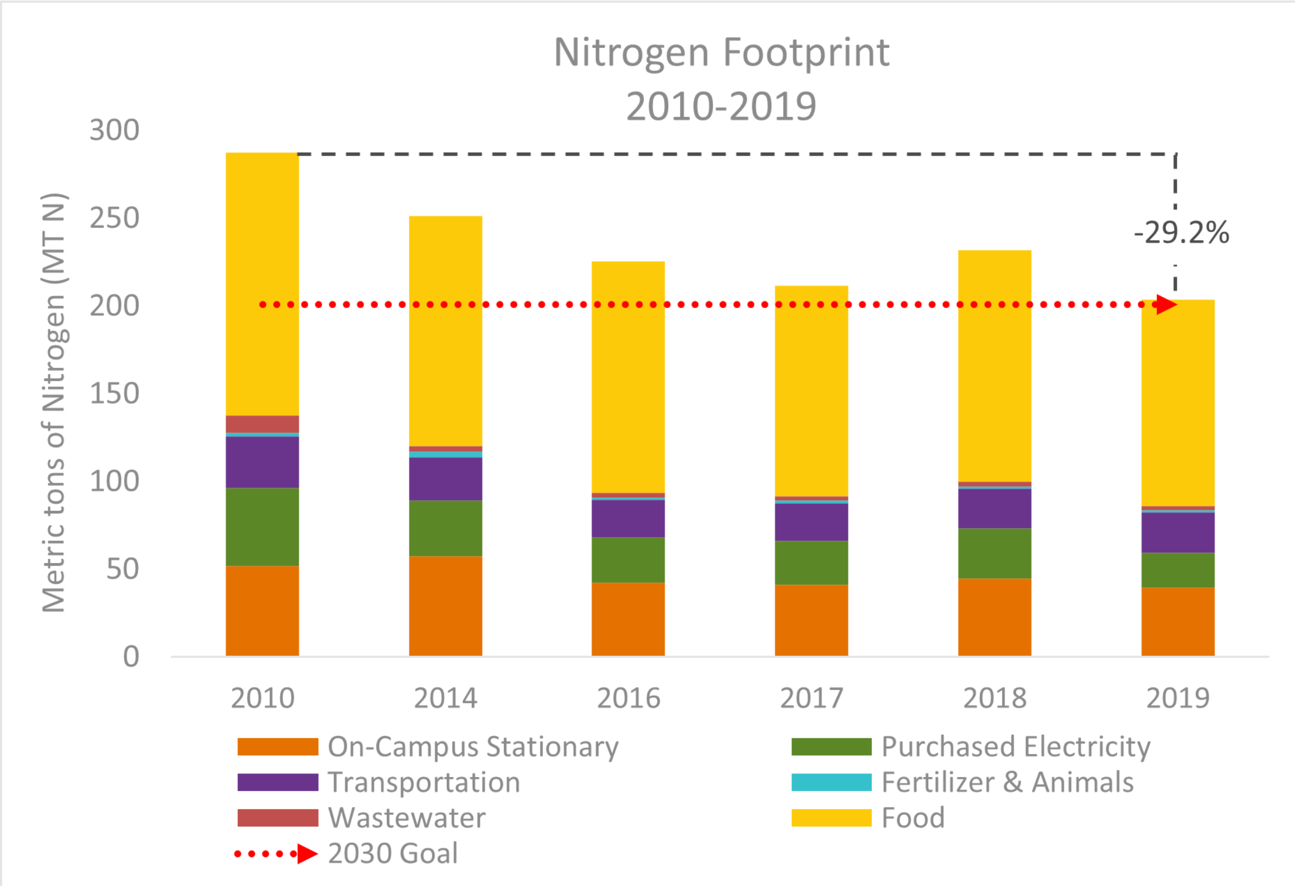


Figure 5.16. The University of Virginia's (UVA's) nitrogen (N) footprint from 2010 to 2017. Source: Sustainability Indicator Management and Analysis Platform (SIMAP). Copyright © 2023 University of Virginia.

There are several potential connections that the NWG's work has with local and global N initiatives. Examples include improving air quality by accounting for and reducing emissions from purchased and stationary electricity, as well as reducing the amount of reactive N present in Chesapeake Bay by promoting local, plant-based food products. The country-specific N footprint tool, N-Calculator, was developed at the University of Virginia in 2010, and the first tools were calculated for the United States and the Netherlands (Leach et al., 2012). The web-based tool (https://n-print.org/) allowed individuals in different countries to estimate their contribution to N losses to the environment through their food consumption, energy use (e.g., transportation, electricity use), and the purchase of goods and use of services. Since published, N footprint tools have been developed for the U.K., Japan, Austria, Tanzania and Australia (Stevens et al. 2014; Shibata et al. 2014; Pierer et al. 2014; Hutton et al. 2017; Liang et al. 2016). With INMS funding, footprint tools are now being developed for Brazil, Denmark, Portugal and Ukraine.

The DPSIR analysis of the University of Virginia Nitrogen Action Plan is shown in Figure 5.17. Overall, it can be concluded that the action plan has led to significant reductions in N waste. The DPSIR process is informative in identifying the drivers of change, monitoring sources, the consequences of change, evaluating changes against agreed-upon benchmarks, and ultimately, responses that led to significant reductions in the university N footprint.

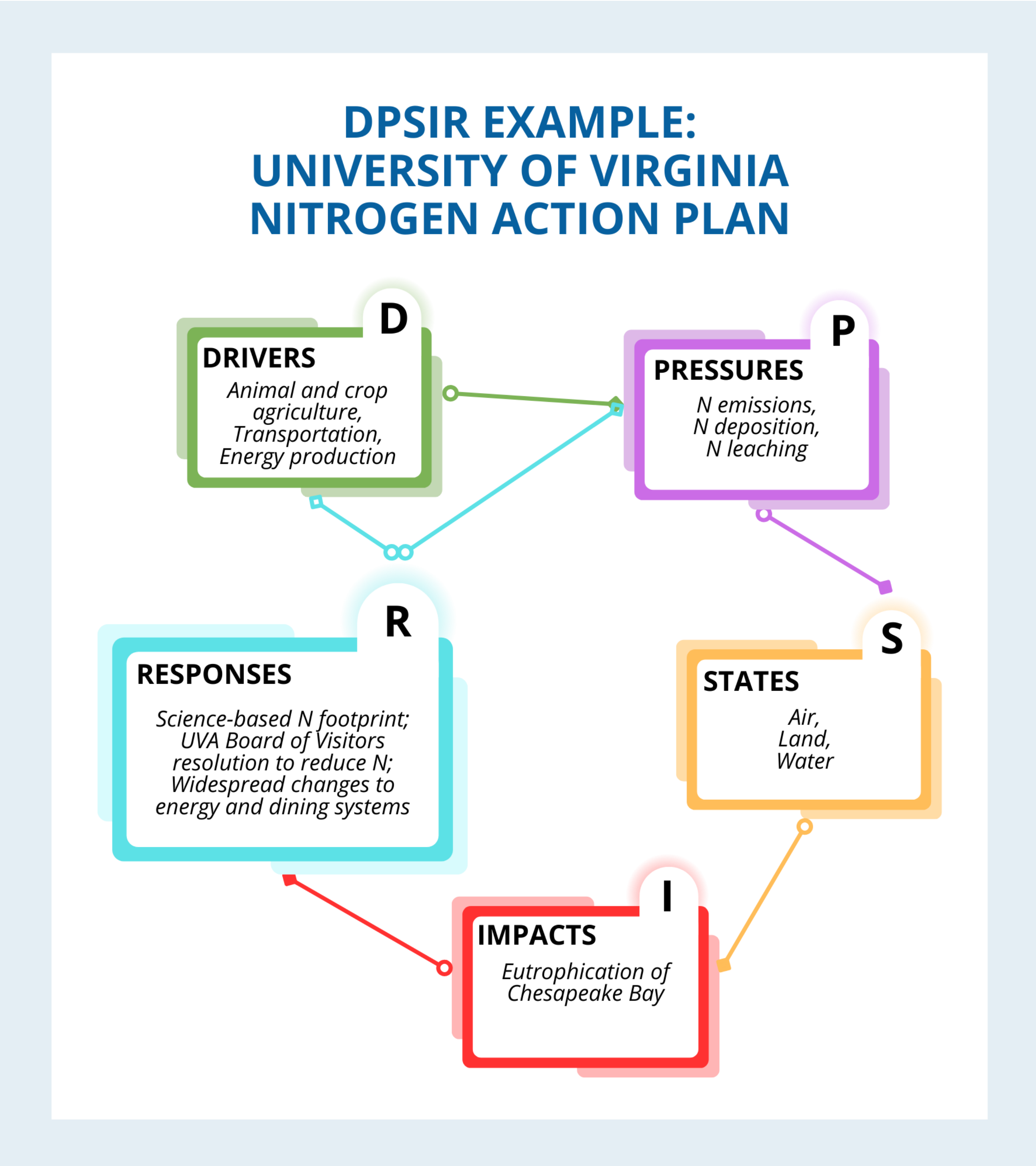


Figure 5.17. Drivers, Pressures, States, Impacts and Reponses (DPSIR) diagram for the University of Virginia (UVA) Nitrogen Action Plan. The DPSIR process is successful in identifying drivers of change, monitoring sources and consequences of change, evaluating changes against agreed-upon benchmarks, leading to significant reductions in nitrogen (N) waste. Original graphic produced for this document © UKCEH 2025.

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Supplementary File

N-MIP matrix

You can download this xlsm file at: https://www.inms.international/N-MIP-Matrix

This INMS Guidance Document on Impact Assessment Methods provides practical guidelines for assessing beneficial and detrimental nitrogen (N) impacts on the environment and humans at all scales from local to regional to global. Written with input from scientists worldwide, this Guidance Document serves as a foundation for improved integrated nitrogen assessment and policy support.

The Guidance Document introduces concepts of reactive N impacts and provides a framework that describes transformation processes of N drivers, pressures and impacts to describe and analyze the positive and negative effects of altered reactive N cycles in different environments. We introduce a comprehensive Nitrogen Matrix of Impacts and Pressures (N-MIP), which is an interactive tool to link N impacts with brief information on underlying mechanisms and determine the category of the impacts. Six different integrated methodologies provide policy makers and other practitioners ways of examining the pathways and trade-offs involved with nitrogen fluxes. These cover: nitrogen budgets, nitrogen footprints, nitrogen use efficiency, planetary boundaries, critical loads, environmental performance index and cost-benefit assessments. The Guidance Document concludes with national or sub-national examples that illustrate how nitrogen impact methodologies have been applied.