A COMPREHENSIVE APPROACH TO NITROGEN IN THE UK

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Glossary

Acidification (of soil)

The loss of nutrient bases (calcium, magnesium, potassium) in the soil, through leaching, and their replacement by acidic elements (hydrogen and aluminium). Pollutant nitrogen deposition (e.g. nitrogen oxides and ammonia) enhances the rate of acidification.

Atmospheric Deposition

Removal of suspended material from the atmosphere, this can be classed as either 'wet' or 'dry'. Wet deposition occurs when material is removed from the atmosphere by precipitation. In dry deposition, the material is removed from the atmosphere by contact with a surface.

Biodiversity

Biodiversity is the variability among living organisms, from genes to the biosphere. The value of biodiversity is multi-fold, from preserving the integrity of the biosphere as a whole, to providing food and medicine, to spiritual and aesthetic wellbeing.

Carbon dioxide equivalent (CO_2e)

Carbon dioxide equivalent describes how much global warming a given type and amount of greenhouse gas may cause, using the functionally equivalent amount or concentration of carbon dioxide (CO_2) as the reference.

Carbon leakage

Carbon leakage occurs when emissions are reduced in the UK due to industry moving offshore where it is cheaper to operate because carbon policies are less ambitious or non-existent.

Carbon sequestration

The capture and removal of carbon dioxide from the atmosphere and storing it in an alternative carbon related reservoir, e.g. soil organic matter, charcoal, tree growth.

Circular Economy

A circular economy aims to maintain the value of products, materials and resources for as long as possible by returning them into the product cycle at the end of their use, while minimising the generation of waste.

Critical Level

Concentration or cumulative exposure of atmospheric pollutants above which direct adverse effects on sensitive vegetation may occur according to present knowledge.

Critical Load

A quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge.

Dead Zone

"Dead zone" is a more common term for hypoxia, which refers to a reduced level of oxygen in the water

Ecosystem services

The benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling that maintain the conditions for life on Earth.

Eutrophication

The enrichment of the nutrient load in ecosystems (terrestrial and aquatic), especially compounds of nitrogen and/or phosphorus. This leads to an undesirable disturbance to the balance of organisms in the ecosystem, affecting terrestrial and aquatic biodiversity and water quality.

Exceedance

The amount of pollution above a 'critical level' or 'critical load', expressed in different ways, such as accumulated area of exceedance.

Forb

A herbaceous flowering plant other than a grass

Global Warming Potential

The global warming potential of a gas or particle refers to an estimate of the total contribution to global warming over a particular time that results from the emission of one unit of that gas or particle relative to one unit of the reference gas, carbon dioxide, which is assigned a value of 1.

Leaching

The washing out of soluble ions and compounds by water draining through soil e.g. nitrate leaching to water bodies

Nitrogen cascade

A term used to describe the passage of reactive nitrogen (Nr) through the environment.

Nitrogen Fixation

Any natural or industrial process that causes free nitrogen (N_2), which is a relatively inert gas plentiful in air, to combine chemically with other elements to form more-reactive nitrogen compounds such as ammonia, nitrates, or nitrites e.g. nitrogen fixing bacteria, lightning or combustion.

Nutrient Nitrogen Critical Load

Empirical nutrient nitrogen critical loads are based on observed changes in the structure or function of ecosystems as reported in the refereed literature from the results of experimental or field studies, or in a few cases dynamic ecosystem modelling.

Ozone

Ozone (O_3), the triatomic form of oxygen, is a gaseous atmospheric constituent. In the troposphere (also referred to as ground level), it is created both naturally and by photochemical reactions involving gases resulting from human activities (it is a primary component of photochemical smog). In high concentrations, tropospheric ozone can be harmful to a wide range of living organisms. Tropospheric ozone acts as a greenhouse gas. In the stratosphere, ozone is created by the interaction between solar ultraviolet radiation and molecular oxygen (O_2). Stratospheric ozone plays a decisive role in the stratospheric radiative balance. Depletion of stratospheric ozone results in an increased ground-level flux of ultraviolet (UV-) B radiation.

Planetary Boundary

Planetary boundaries define the safe operating space for humanity with respect to the Earth system and are associated with the planet's biophysical subsystems or processes.

Pollution swapping

Pollution swapping can be defined as the increase in one pollutant as a result of a measure introduced to reduce a different pollutant.

Reactive nitrogen

Collectively any chemical form of nitrogen other than di-nitrogen (N_2), the unreactive gas which makes up around 78% of the atmosphere. Reactive nitrogen (Nr) compounds include ammonia (NH_3), nitric oxide and nitrogen dioxide (NOx), nitrous oxide (N_2O), nitrate (NO_3^-) and many other chemical forms, and are involved in a wide range of chemical, biological and physical processes.

Tropospheric

The lowest region of the atmosphere between the earth's surface and the tropopause, characterized by decreasing temperature with increasing altitude

Acronyms and Abbreviations

AD	Anaerobic digestion
AEZ	Agro-ecological Zone
AMF	Arbuscular Mycorrhizal Fungi
AR5	IPCC Fifth Assessment Report
ASSI	Areas of Special Scientific Interest (Northern Ireland)
AUE	Agronomic Use Efficiency, i.e. mass of yield increase per mass of applied nutrient, such as nitrogen
BNF	Biological Nitrogen Fixation
CAP	Common Agricultural Policy of the European Union
CBD	Convention of Biological Diversity
CCAC	Climate and Clean Air Coalition
CCC	Committee on Climate Change
CCS	Carbon capture and storage
CEA	Controlled Environment Agriculture
CFCs	Chlorofluorocarbons
CE	Capture Efficiency, i.e. the amount of a nutrient in the harvested product compared with the total nutrient uptake by the crop
CH_4	Methane
CHP	Combined Heat and Power
CL	Critical Load
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
CPR	Committee of Permanent Representatives of the United Nations Environment Programme
DAERA	The Department of Agriculture, Environment and Rural Affairs (Northern Ireland)
DAP	Di-ammonium phosphate, used as a mineral fertilizer
DEFRA	Department for Environment, Food and Rural Affairs (UK)
DM	Dry Matter
DMPP	3,4-dimethylpyrazole phosphate – a nitrification inhibitor

DNMARK	Danish Nitrogen Mitigation Assessment
DON	Dissolved Organic Nitrogen
EC	European Commission
EF	Emission factor
EGR	Exhaust gas recirculation
EIONET	European Environment Information and Observation Network
ELMS	Environment Land Management Schemes
EMEP	European Monitoring and Evaluation Programme
ENA	European Nitrogen Assessment
EPNB	Expert Panel on Nitrogen Budgets of the TFRN
ES	Ecosystem Services
EU	European Union
EU-NEP	European Union Expert Nitrogen Panel
EU27	European Union 27 Member States
FCR	Feed Conversion Ratio
FAO	Food and Agriculture Organization of the United Nations
FFCC	Food, Farming and Countryside Commission
FRfW	Farming Rules for Water
FSA	Food Standards Agency
FYM	Farmyard Manure
GAP	Good Agricultural Practices
GAW	Global Atmospheric Watch
GDP	Gross Domestic Product
GEF	Global Environment Facility
GHG	Greenhouse Gas
GIS	Geographic Information Systems
GM	Genetically Modified
GMO	Genetically Modified Organism
GPA	Global Programme of Action for the Protection of the Marine Environment
GPNM	UNEP Global Partnership on Nutrient Management
GWP	Global Warming Potential

HAB	Harmful Algal Bloom
HB	Haber-Bosch
HI	Harvest Index
HLPF	High-Level Political Forum on Sustainable Development
IDDRI	Institut du développement durable et des relations internationals (Institute for sustainable development and international relations)
IIR	Informative Inventory Report
INA	International Nitrogen Assessment
INC	Internal Nitrogen Cycle
INE	Internal Nutrient Efficiency
INI	International Nitrogen Initiative
INS	Indigenous Nutrient Supply
INMS	International Nitrogen Management System
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISTM	Integrated Soil Fertility Management
IVC	In-Vessel Composting
K	Potassium
KAP	Knowledge, attitudes, practices
LAQM	Local Air Quality Management
LCA	Life Cycle Analysis
LED	Light-Emitting Diode
LRTAP	UNECE Convention on Long-range Transboundary Air Pollution (informally the 'UNECE Air Convention')
LUC	Land Use Change
MEA	Multilateral Environmental Agreement
MPA	Marine Protected Area
MRIO	Multi-Regional Input Output modelling using economic statistics
MSC	Marine Stewardship Council
NAEI	National Atmospheric Emissions Inventory
NCS	Nitrogen Credit System

NDC	Nationally Determined Contribution under the UNFCCC
NECD	National Emission Ceiling Directive
NGO	Non-governmental Organization
NPN	Non-protein Nitrogen
Nr	Reactive Nitrogen, chemically active forms of nitrogen that interact with the environment and support plant growth, they are typically scarce in the natural environment. Collectively any chemical form of nitrogen other than di-nitrogen (N_2), the unreactive gas which makes up around 78% of the atmosphere. Reactive nitrogen (Nr) compounds include ammonia (NH_3), nitric oxide (NO) and nitrogen dioxide (NO_2) (NO + NO_2 = NOx), nitrous oxide (N_2O), nitrate (NO_3^-) and many other chemical forms, and are involved in a wide range of chemical, biological and physical processes.
NRW	Natural Resources Wales
N_2	Di-nitrogen, a colourless and odourless diatomic gas, forming about 78% of Earth's atmosphere
N_2O	Nitrous oxide – a powerful greenhouse gas
NB	Nitrogen Balance (partial), i.e. the different between inputs (e.g. fertilizer, biological nitrogen fixation, manure) and outputs (crop harvest and other removed residues)
N balance	Difference between N inputs and outputs of a system, where a positive value is typically termed the N surplus. May be defined at field, farm and regional scales
NBPT	N-(n-butyl) thiophosphoric triamide – a urease inhibitor that slows the conversion of urea to NH_x
NCE	Nitrogen Capture Efficiency, the amount of nitrogen taken up or 'captured' by a crop as a fraction of the N added as input to the soil (i.e., availability') from external supply and internal supply (mineralization)
NGO	Non-governmental Organisation
NFS	National Food Strategy
NF_3	Nitrogen trifluoride
NH_3	Ammonia - an air and water pollutant and the primary nitrogen form in biological systems
NH_4^+	Ammonium – present in biological systems and soils, while forming a pollutant in atmospheric PM and aquatic systems
NH_x	Total ammoniacal nitrogen sometimes referred to as TAN
NI	Nitrification Inhibitor
Nnet	Nitrogen Human Environment Network

NO	Nitric oxide – a tropospheric air pollutant
NO_2	Nitrogen dioxide – a tropospheric air pollutant
NO_3^-	Nitrate – present as a secondary pollutant in atmospheric PM and a eutrophying pollutant of aquatic systems
NO_x	Nitrogen oxides – a combination of NO and NO_2
NPK	Nitrogen, Phosphorus and Potassium in combination
N_r	Reactive Nitrogen, a term used for a variety of nitrogen compounds that support growth directly or indirectly, as opposed to N_2 which is inert
NUE	Nitrogen Use Efficiency. Typically defined as the ratio of N in outputs divided by the N in inputs. May be defined for different systems such as crops, livestock, food chain and the whole economy
NVZ	Nitrate Vulnerable Zone
O_3	Ozone
OECD	Organization for Economic Co-operation and Development
P	Phosphorus
POM	Particulate Organic Matter
PB	Planetary boundary
PHE	Public Health England
PM	Particulate Matter, which includes NH_4^+ and NO_3^- as major components; PM_{10} and $_{2.5}$ refer to atmospheric particulate matter (PM) that has a diameter of less than 10 and 2.5 micrometers respectively. $PM_{2.5}$ is also known as Fine Particulate Matter
PTE	Potentially Toxic Element
PUE	Phosphorus Use Efficiency
RE	Recovery Efficiency, i.e. mass increase of nutrients in harvested crop as a fraction of the mass of nutrients applied
RF	Rain-Fed
RIVM	National Institute for Public Health and the Environment, the Netherlands
RNA	Ribonucleic acid
RSPB	Royal Society for the Protection of Birds
SAC	Special Areas of Conservation
SACEP	South Asia Co-operative Environment Programme
SAFFO	Silage, Slurry and Agricultural Fuel Oil (England) Regulations

SCR	Selective catalyst reduction
SDGs	Sustainable Development Goals
SI	Supplementary Irrigation
SIA	Secondary inorganic aerosol
SFI	Sustainable Farming Initiative (England)
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SSA	Sub-Saharan Africa
SSC	Soil Supply Capacity, i.e. ability of the soil system to replenish a given plant nutrient in the soil solution for plant uptake
SNAP	Site Nitrogen Action Plan
SNBS	Scottish Nitrogen Balance Sheet
SRUC	Scotland's Rural College
SSSI	Site of Special Scientific Interest
STFR	Soil testing and fertilizer recommendation
TAN	Total Ammoniacal Nitrogen
TFRN	Task Force on Reactive Nitrogen of the UNECE Convention on Long-range Transboundary Air Pollution
TN	Total nitrogen
TSP	Triple Super Phosphate
TWh	Terawatt Hour
UN	United Nations
UNDP	United Nations Development Programme
UNDSD	United Nations Division for Sustainable Development
UNEA	United Nations Environment Assembly
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework on Climate Change
UNIDO	United Nations Industrial Development Organization
USD	United States Dollars

VCR	Value-Cost-Ratio, i.e. the ratio of the price of additional yield (e.g. crop yield increment) following application of inputs (e.g. fertilizer, but excluding seeds) to the cost of the inputs
WAGES	Water, Air, GHG, Ecosystems/Biodiversity and Soils
WEL	Wales Environment Link
WFD	Water Framework Directive
WHO	World Health Organization
WMO	World Meteorological Organization
WRAP	Waste and Resources Action Programme
WWTP	Wastewater Treatment Plant

1. Nitrogen losses to the environment and its impacts

1.1. Introduction

The sustainability of our world depends fundamentally on the use of nitrogen (N), which is a vital element in all forms of life. Nitrogen accounts for 80% of the atmosphere on Earth in its inert gaseous form (N_2). Natural cycles have developed in such a way that globally, approximately 200 Tg N yr^{-1} is converted from inert N into reactive N (Nr) compounds via biological nitrogen fixation (BNF), lightning and natural combustion (Figure 1.2.1). In order to feed >7 billion people, humans have more than doubled global land-based cycling of N since pre-industrial times (Fowler et al., 2013). Since the 1960s, human use of synthetic N fertilizers has increased 9-fold globally, and a further substantial increase of around 40-50% is expected over the next 40 years in order to feed the growing world population and because of current trends in dietary lifestyles, with increasing consumption of N intensive animal products (Sutton et al., 2013). These changes will exacerbate current environmental and climate-induced problems unless urgent action is taken to reduce and improve the efficiency of N use, to reduce the waste of valuable N resources, and to re-evaluate societal ambitions for future per capita consumption patterns.

Major inequalities still exist between those parts of the world with surplus nutrients and those that do not have enough. The key regions mobilizing excess nutrients include North America, Europe, and parts of South and South East Asia and Latin America. In Africa, Latin America and parts of Asia there are wide regions with insufficient nutrients to meet crop demand and food security needs (Sutton et al., 2013). While the distribution and application of commercial N fertilizers has provided large benefits to the world's human population, the collective use of commercial fertilizers, manure, and legume crops, to provide N to grow crops for human consumption (18%) and animal feed (82%), needs to be more efficient to avoid N losses causing risks on public health, the economy, and the environment (e.g. see Section 1.1.1 to 1.1.4). These risks include:

- reductions in biodiversity (i.e. degradation of sensitive habitats);
- accelerated climate change via the production of nitrous oxide gas (N_2O);
- widespread air and water pollution leading to growing incidences of upper respiratory disease and cancer in humans, including the role of oxidized N in tropospheric (ground-level) ozone formation (a potent GHG that can also impact on human health and crop yields);
- depletion of stratospheric ozone layer via the production of nitrous oxide gas (N_2O);
- eutrophication and hypoxic “dead zones” in the coastal ocean; and
- acidification of soils and forests of natural ecosystems.

It can be seen that tackling N pollution by tightening the N cycle will have multiple benefits across the environmental, economic and social pillars of sustainable development, including meeting the ‘Triple Challenge’ of meeting the food needs of the world, while tackling the climate crisis and reversing the loss of nature (Baldwin-Cantello et al., 2020).

The social cost of impacts of N pollution in the EU27 in 2008 was estimated between €75–485 billion per year for all sources (Van Grinsven et al., 2013). The economic benefit of N in primary agricultural production in this period ranged between €20−80 billion yr^{-1} and was lower than the annual cost of pollution by agricultural N which is in the range of €35−230 billion yr^{-1}. Similarly, in the United States, potential health and environmental damages of anthropogenic N at the national scale in the early 2000s totaled $210 billion yr^{-1} USD (range: $81–$441 billion yr^{-1}) (Sobota et al., 2015). Although not reported directly in literature, in this report we estimate that a cost of approximately £10.9 (2.7 - 27.1) billion per year of societal costs can be attributed to N pollution in the UK (see Section 1.6). Of these costs, approximately 60% are attributed to the impact on human health, predominantly that of oxidized N (NO_x) and reduced (NH_3) emissions.

On a global scale, the planetary boundary (PB) for N has been estimated to be exceeded by at least a factor 2 (Steffen et al., 2015; Erisman et al., 2015; and see Section 1.2). This means that for N, the safe operating space of humanity with respect to the earth system has been seriously transgressed. The PB for N has been taken from the comprehensive analysis of de Vries et al. (2013), which proposed a PB for eutrophication of aquatic eco-systems of 62 Tg N $year^{-1}$ from industrial and intentional biological N fixation, using the most stringent water quality criterion, although regional distribution of fertilizer N is critical for impacts. This can be compared to a current value of industrial and intentional biological fixation of N at global level of 150 Tg N $year^{-1}$.

A recent report by WWF and 3keel entitled 'Thriving Within Our Planetary Means' (Jennings et al., 2021), has taken PB analysis one step further by combining national consumption-based environmental footprints to "downscaled" planetary boundaries (e.g. Fang et al., 2015). This analysis yielded a UK per capita footprint of 72.9 kg N yr^{-1} and this was compared to a global per capita PB of 7.9 to give a required reduction in the UK per capita footprint of 89%. This top–down approach was used as it assigns an equal share of the PB on a per capita basis, to explore the benefits that could be universally achieved if resources were distributed equally. The different footprint approaches are described in Section 1.4.3.

A key concern with Nr is that it can move through the environment causing multiple effects in the atmosphere, in terrestrial ecosystems, in freshwater and marine systems, and on human health. This phenomenon is known as the 'Nitrogen Cascade', which can amplify Nr effects through both time and space and make them difficult to manage (Section 1.4.3). Immediate action is therefore needed to reduce the use of Nr and to better manage N losses in order to limit its cascading effects. However, despite its relevance to most UN Sustainable Development Goals (SDGs), nitrogen pollution still lacks broad visibility and coordinated global governance. A new goal to ''halve nitrogen waste'' by 2030 is estimated to save US$100 billion annually (Sutton et al., 2021), contributing to post-coronavirus disease 2019 (COVID-19) economic recovery and multiple SDGs. The scientific community is working with the UN to coordinate and accelerate the necessary action (See Section 3.1.1).

1.1.1 Nitrogen impacts on Water Quality

Excess nitrate/nitrite exposure in food and water may be harmful to human health by contributing to the formation of carcinogens and teratogens (linked to pregnancy defects), causing changes in thyroid activity, and high nitrate levels in drinking water have been linked to methemoglobinemia in infants ("blue baby syndrome") and children (Brender, 2020). High nitrate concentrations in aquatic systems also have nutrient loading and acidification impacts on sensitive systems (Environment Agency, 2019).

In the UK, agriculture is the dominant source of nitrate in water (about 70% of total inputs), with sewage effluent a secondary contributor (25-30%) nationally (Environment Agency, 2019). In general, nitrate concentrations are greatest in the drier, arable-dominated southern and eastern areas of England, coinciding with areas most dependent on groundwater for public water supply and base flow to rivers. Around 2019, 55% of England was designated as a Nitrate Vulnerable Zone (NVZ) due primarily to elevated nitrate concentrations in groundwater and rivers, and to a lesser degree because of eutrophication of estuaries and lakes/reservoirs. NVZ action programmes to reduce agricultural nitrate pollution have been in place since the late 1990s, reducing river nitrate concentrations until more recently, when they have shown increases. Groundwater nitrate concentrations have been broadly stable in many places except in southern England where they have risen in some areas. This is partly explained by the lag time for the peak agricultural nitrate loadings of the 1980-90s to percolate through the water table. Changes in farming practice such as spreading more materials on land also have the potential to greatly increase nitrate loading locally.

In England, only 16% of water bodies meet the criteria for 'good' ecological status and none meet the criteria for 'good' chemical status, with the majority at around 60% being only of 'moderate' status in terms of pollution levels. The situation is better in Scotland where 50% of water bodies achieved good status as well as 13% achieving high status, and Wales where 40% of water bodies achieved good status (JNCC, 2020, Environment Agency, 2020).

1.1.2 Nitrogen impacts on Health

A major public concern is the rise in toxic fine particulate matter ($PM_{2.5}$ - fine particles in the air <2.5 µm in aerodynamic diameter) levels, a significant fraction of which is caused by N emissions, which can result in economic damages and health risks in downwind communities (Paulot & Jacob, 2014). The UK Committee on the Medical Effects of Air Pollution (COMEAP, 2018) report that their current estimate of the mortality burden of air pollution in the UK using a coefficient based on $PM_{2.5}$ (COMEAP, 2010) is equivalent to nearly 29,000 deaths and an associated loss of 340,000 life years across the population in a single year. The methodology used by COMEAP (2018) allows quantification using either $PM_{2.5}$ or NO_2 as the primary indicator of the mixture, and uses unadjusted coefficients to capture the effect of the mixture as a whole via single-pollutants analyses. The extent to which $PM_{2.5}$, NO_2, or other pollutants with which they are correlated contribute to the overall mortality burden of the air pollution mixture is not clear (e.g. correlation of NO_2 pollution with ground-level ozone production).

Secondary inorganic aerosol (SIA), formed in the atmosphere form primary emissions, has been estimated to contribute up to 40% of total $PM_{2.5}$ in the UK (AQEG 2018). The main contributors to SIA in the UK are nitrate (NO_3), ammonium (NH_4) and sulphate (SO_4), with NO_3 the dominant contributor by mass and NH_4 variable in time and space but generally contributing between a few percent and 20% (AQEG 2018). For NH_4 Part of the variability arises due to seasonal variations in emissions due to fertilizer and manure spreading, but the effects of weather are also very important, as temperature strongly influences aerosol formation from gases. Some of the largest contributions of NH_3 to $PM_{2.5}$ occur in spring, when emissions are high and temperatures are cool. In Paris during an air quality episode in spring 2014, 62% of $PM_{2.5}$ was estimated to originate from NH_3 (AQEG 2018). Furthermore, a modelling study by Vieno et al. (2016) has estimated that about 50% of the particulate NH_4 related PM in the UK may originate from gases emitted elsewhere in Europe. It has been estimated that a global halving of agricultural emissions could reduce the mortality attributed to $PM_{2.5}$ by ~250,000 globally and by 52,000 across Europe (Pozzer et al., 2017).

1.1.3 Nitrogen impacts on Climate change

UK territorial greenhouse gas emissions account for around 1% of the global total. In 2019, net territorial emissions in the UK of the basket of seven greenhouse gases covered by the Kyoto Protocol were estimated to be 454.8 million tonnes carbon dioxide equivalent ($MtCO_2e$) (43.8% lower than 1990), to which nitrous oxide contributed ~5% in 2019 (BEIS, 2021; see Section 1.4.2).

The land use, land use change and forestry (LULUCF) sector is now estimated to have had net emissions of 5.9 $MtCO_2e$ in 2019 (following a methodology revision to better represent emissions from drained and rewetted inland organic soils (peatlands)). This is down from a total of 18.0 $MtCO_2e$ in 1990, and the long-term fall has been driven by a reduction in emissions from cropland and grassland, and an increase in the sink provided by forest land, with an increasing uptake of carbon dioxide by trees as they reach maturity, in line with the historical planting pattern (BEIS, 2021b). There has also been some reduction in emissions since 1990 due to changes in agricultural practices.

1.1.4 Nitrogen impacts on Ecosystems and Soils

A report to Defra tracking progress in obtaining national (e.g. Clean Air Strategy, 2019) and international targets (e.g. Convention on Long-range Transboundary Air Pollution (CLRTAP) and EU National Emission Ceilings Regulations (NECR)) to protect ecosystems in the UK (Rowe et al., 2020), shows that:

(i) the area of N-sensitive habitats in the UK with exceedance of nutrient N critical loads fell from 75.0% in 1996, to 62.5% in 2012, but was still 57.6% (42,049 km^2) in 2017;

(ii) the area of acid-sensitive habitats in the UK with exceedance of acidity critical loads has fallen by more than one third, from 77.3% in 1996, to 47.4% in 2012, to 38.8% (27,253 km^2) in 2017, due mainly to decreases in sulphur deposition;

(iii) In 2016 just over 5% (12,433 km^2) of the UK land area was exposed to ammonia concentrations above the critical level set to protect higher plants (3 $\mu g\ m^{-3}$), and just over 60% (153,960 km^2) to ammonia at concentrations above the critical level set to protect lichens and mosses (1 $\mu g\ m^{-3}$). The area where the critical level for higher plants is exceeded has increased by 1.4% (3,321km^2) of UK land area since 2010. The area where the critical level for lichens and mosses is exceeded has decreased by 1% (2,482km^2) of UK land area since 2010.

Exceedance of critical loads or critical levels indicates that ecosystems are at risk from potential harmful effects, such as loss of biodiversity or changes in composition to more nitrogen loving species to the detriment of plants adapted to low nutrient conditions (e.g. forbs). The risk of harm is reduced when pollution decreases to below the critical load or level, but there may be delays to recovery.

As well as the negative effect on various sensitive habitats described above, atmospheric N deposition is also increasing carbon loss from peat bogs and about 15% of woodland soil in England and Wales is N saturated, that can increase nitrate leaching from soils and associated aluminum toxicity to the plant roots (Environment Agency 2019b). The Environment Agency (2019b) also report that soil biodiversity and the many biological processes and soil functions that it supports are thought to be under threat. UK soils currently store about 10 billion tonnes of carbon, roughly equal to 80 years of annual UK greenhouse gas emissions, but intensive agriculture has caused arable soils to lose about 40 to 60% of their organic carbon, plus wasting food and growing crops for bioenergy are putting additional pressure on soils. Almost 4 million hectares of soil are at risk of compaction in England and Wales, affecting soil fertility and water resources, and increasing the risk of flooding. Furthermore, when soils become compacted, they are more likely to become waterlogged and experience surface ponding that leads to run-off and flooding. This increases nutrient losses to watercourses causing pollution and reducing nutrient levels in soil. As a result, twice the amount of nitrogen fertiliser is needed to maintain yields (Environment Agency, 2019).

Reversing soil degradation and restoring fertility by 2030 is an aim of the government's 25 Year Environment Plan. The proposed Environmental Land Management (ELMS) scheme provides an opportunity to reward farmers for protecting and regenerating soils.

A relatively new threat, that requires further research, is the contamination of soils by nanoparticles as a result of sewage sludge spread to land and via pesticide applications. Silver nanoparticles are being applied to soils via sewage sludge and have been shown to be toxic to plants, affecting their root production (Environment Agency 2019b). Biosolids containing nanomaterials can disrupt plants' uptake of nitrogen and can change the types of microorganisms found in the soil, negatively affecting rates of plant growth. They have also been shown to be toxic to bacterial communities.

1.2 Global nitrogen: Sources, processes, drivers and flows

Total global fixation of reactive nitrogen (Nr) is estimated at 413 Tg per year (or million tonnes per year) (Fowler et al. 2013; see Figure 1.2.1), of which 210 Tg per year results from human activities. According to these estimates humans have thus doubled global supply of reactive N compounds, with the main sources being industrial fixation of N in the Haber-Bosch process, increased biological nitrogen fixation (BNF) through agricultural activities and increased fossil fuel combustion. It should be noted however, that natural oceanic N fixation is to a large extent separate from the terrestrial N cycle. Therefore, considering only the terrestrial part, it can be seen that human driven N fixation over three times larger than the natural sources (Figure 1.2.1) (For further details see Sutton et al., 2013, p 22). As a result of human activities the global N cycle is now out of balance, causing major environmental, health and economic problems.

Technological breakthroughs in the creation, distribution, and application of N fertilizers have underpinned major advances in food, fuel, and fiber production; but poor management practices and inefficient N fertilizer applications to agricultural lands are harming the economy, with several hundred million USD of annual financial losses ascribed to excess N use in developed nations (Sutton et al., 2011; Van Grinsven et al., 2013; Sobota et al., 2015). Unless action is taken, increases in population and per capita consumption of energy and animal products will exacerbate N losses, pollution levels and land degradation, further threatening the quality of our water, air and soils, affecting climate and biodiversity.

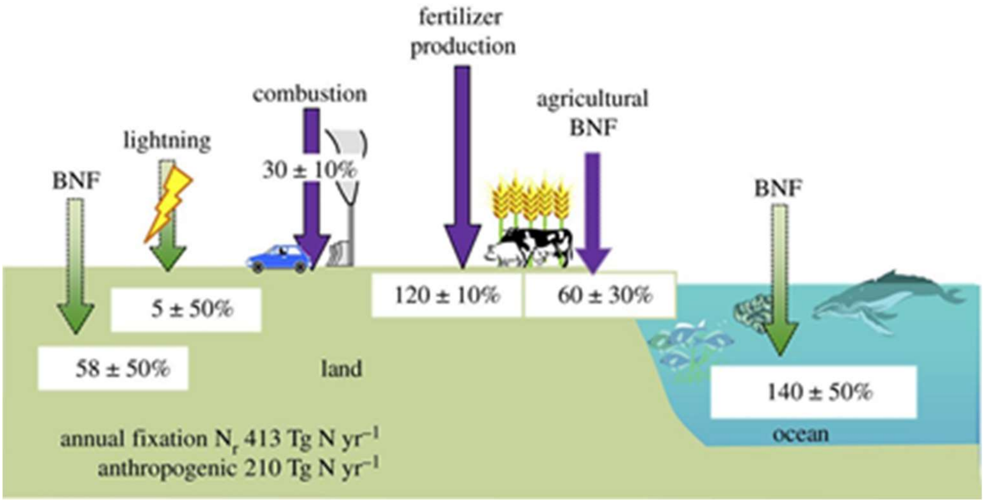

Figure 1.2.1. *Global nitrogen fixation, natural and anthropogenic in both oxidized and reduced forms through combustion, biological fixation, lightning and fertilizer and industrial production through the Haber–Bosch process for 2010. The arrows indicate a transfer from the atmospheric N_2 reservoir to terrestrial and marine ecosystems, regardless of the subsequent fate of the Nr. Green arrows represent natural sources (Biological Nitrogen Fixation, BNF), purple arrows represent anthropogenic sources (Fowler et al. 2013).*

The current N biogeochemical cycle is strongly affected by the agricultural system (Figure 1.2.2 & Table 1.2.1). Globally, humans introduce 120 Tg N per year of new N as synthetic fertiliser to sustain crop and grass production and as feedstock for many industrial processes (Figure 1.2.1; cf. Galloway et al., 2008), in addition to 50-70 Tg N which is fixed biologically by the agricultural system (Figure 1.2.1; cf. Herridge et al., 2008). Only around 16 to 20% of the N originally introduced in agricultural soils ends up in food for human consumption, and only 11% is consumed (after food waste is considered) while the rest is wasted to the wider environment (Billen et al., 2013; Sutton et al., 2013; Sutton & UNEP, 2013). Similarly, only a small fraction of N input to the livestock system is then consumed by humans as meat, whereas a large part is lost or recycled by agricultural soils (Billen et al., 2013) (more details below).

According to Erisman et al. (2015), the main drivers contributing to the overuse of N and the resulting impacts can be categorized as:

- The inefficient and unsustainable use of N-fertilizer and manure leading to large losses to terrestrial and aquatic ecosystems;
- Increased global consumption levels as a result of human population growth, increase in per capita consumption and a diet shift towards more protein-rich food which has led to an increased demand for agricultural products and consequently a rise in the use of N-fertilizers (and its inefficient use);
- Increased demand for fossil fuels, and the resulting release of Nr in the atmosphere during combustion.

N losses to the environment are represented mainly by ammonia (NH_3) emissions from agricultural and livestock production systems (37 Tg N per year, Sutton et al., 2013), soil denitrification (25 Tg N, Billen et al., 2013), and N leaching and runoff (95 Tg N, Billen et al., 2013). Humans contribute to the loss of N in the environment also by food waste, which according to FAO statistics represents one third of the food produced globally, and by sewage discharges, that are only partially treated. As a result, unintended emissions of N to soils, fresh waters and the atmosphere are produced.

Fresh waters receive around 39-95 Tg N per year from agricultural soils (Bouwman et al., 2011; Billen et al., 2013). Part of this remains in superficial aquifers, part is lost to the atmosphere by denitrification, contributing to the emission of the strong greenhouse gas nitrous oxide (N_2O), and part is discharged to coastal waters (40-66 Tg N per year, Voss et al., 2011; Seitzinger et al., 2005), where it fosters local water eutrophication and hypoxia. N fluxes within oceans are only partially known at global scale as many of the processes involved depend on microbial organisms and on the availability of the other nutrients (i.e. carbon (C), phosphorus (P) and silica (Si)). However, it is clear that human activities have increased N inputs from rivers and atmospheric deposition, while they have altered the stoichiometry of C, N, P and Si, especially in coastal waters (Voss et al., 2013). Globally, ocean N fixation is estimated to be 140 Tg N per year (Deutsch et al., 2007), deposition 30-67 Tg N (Fowler et al., 2013; Duce et al., 2008) and denitrification around 100-250 Tg N (Voss et al., 2013), contributing to the emission of 5.5 Tg N as N_2O (Duce et al., 2008). Fish landing represents only 4 Tg N per year (Maranger et al., 2008).

Table 1.2.1. Global nitrogen fluxes around year 2000-2010 reported in the literature (Sutton et al., 2013).

Legend/Global Nitrogen Fluxes	Tg N/yr	References	Additional References
1. Fertiliser consumption	120	(Fowler et al., 2013)	Galloway et al. (2008), Bouwman et al. (2011)
2. N_2 crop fixation	50-70	Fowler et al., 2013)	Herridge et al. (2008)
3. Crops & grass production	122	(Billen et al., 2013)	
4. Crops & grass for livestock production	100	(Billen et al., 2013)	
5. N back to agricultural soils	57		Based on (Billen et al., 2013; Sutton et al., 2013)
6. NH_3 emissions–agricultural system–from crops & grass	15	(Sutton et al., 2013)	
7. NH_3 emissions–agricultural system–from livestock	22	(Sutton et al., 2013)	
8. NH_3 emissions–agricultural system (total)	37	Sutton et al., 2013)	
9. Crops for human nutrition	22	(Billen et al., 2013)	
10. Livestock for human nutrition	6	(Billen et al., 2013)	
11. Fish landing	3.7	Voss et al. (2013)	Maranger et al. (2008)
12 .Food waste	13	(Billen et al., 2013)	
13. Human excretion	19	(Billen et al., 2013)	
14. Waste water treatment	13	(Billen et al., 2013)	
15. Sewage	6	(Billen et al., 2013)	
16. Riverine input to oceans	40-66	Voss et al. (2013)	Voss et al. (2011) & Seitzinger et al.(2005)
17. Surplus in agricultural soils	120	(Billen et al., 2013)	
18. Input from agricultural soils to aquifers and rivers	95	(Billen et al., 2013)	
19. Soil denitrification	25	(Billen et al., 2013)	
20. Denitrification in aquatic systems	52	(Billen et al., 2013)	
21. NO emissions from soils	10	Fowler et al., 2013)	
22. N_2O emissions from soils	13	Fowler et al., 2013)	
23. Lightning	2-10	Fowler et al., 2013)	Levy et al. (1996) & Tie et al. (2002)
24. N_2 natural fixation in terrestrial ecosystems	58	Fowler et al., 2013)	Vitousek et al. (2013)
25. NH_3 emissions–biomas burning	5.5	(Sutton et al., 2013)	
26. NH_3 emissions–natural soils	4.9	(Sutton et al., 2013)	
27. NH_3 emissions–natural ecosystems	10.4	(Sutton et al., 2013)	
28. Combustion	30-40	Fowler et al., 2013)	Van Vuuren et al. (2011a)
29. Wet and dry deposition on soils	70	Fowler et al., 2013)	Dentener et al. (2006) & Duce et al. (2008)
30. Wet and dry deposition on oceans	30	Fowler et al., 2013)	Dentener et al. (2006) & Duce et al. (2008)
31. NH_3 emissions–oceans (and volcanoes)	8.6	(Sutton et al., 2013)	
32. N_2O emissions from the ocean	5.5	Voss et al. (2013)	IPCC (2007) & Duce et al. (2008)
33. Denitrification in oceans	100-250	Voss et al. (2013)	Voss et al. (2011)
34. N_2 Fixation by oceans	140	Voss et al. (2013)	Deutsch et al. (2006) &Duce et al. (2008)
35. Burial in oceans	22	Voss et al. (2013)	
36. Flux from coastal ocean to open ocean	390	Voss et al. (2013)	
37. Flux from open ocean to coastal ocean	450-600	Voss et al. (2013)	
38. Wet and dry deposition of NHx and NOy on agricultural soils	50		Based on Dentener et al. (2006) & Duce et al. (2008)
39. Wet and dry deposition of NHx and NOy on natural soils	19		Based on Dentener et al. (2006) & Duce et al. (2008)

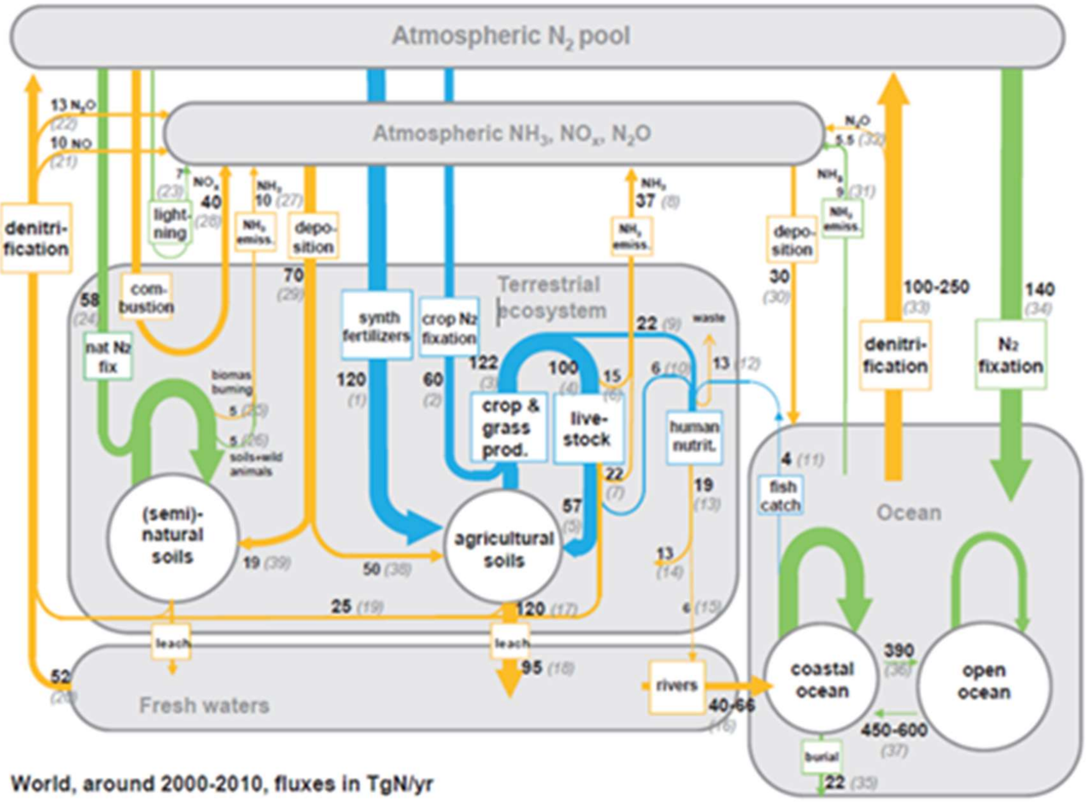

Figure 1.2.2. Global nitrogen cycle around years 2000-2010. The arrows show the nitrogen fluxes across environmental pools and compartments (green: natural fluxes, blue: intended fluxes, orange: unintended fluxes or substantially perturbed fluxes, more explanation is provided in the text). The figures in black indicate the nitrogen fluxes in Tg N per year and the figures within brackets refer to legend numbers in the accompanying table including the references for each flux (see Table 1.2.1) The diagram is based mainly on the values reported by Fowler et al., 2013; Billen et al. 2013; Voss et al., 2013; and Sutton et al., 2013; and the references cited therein (note that not all figures add exactly, due to the use of different data sources (Sutton, 2013)).

Combustion is responsible for the emission of 30-40 Tg N per year as NOx (Fowler et al., 2013; Van Vuuren et al., 2011), which is about five times the NOx naturally produced in the atmosphere by lightning (Fowler et al., 2013; Levy et al., 1996; Tie et al., 2002). Burning fossil fuels produces a significant additional Nr resource (~20% of human Nr production) that could be captured and used, but which is currently wasted as emissions of nitrogen oxide (NOx) to air, contributing to particulate matter and ground-level (tropospheric) ozone production in the atmosphere that adversely affect human health, ecosystems and food production systems. Ammonia emissions from agricultural systems are estimated at 37 Tg N per year with a further 15 Tg N from biomass burning, industrial and various waste sources, as compared to the 13 Tg N per year from natural systems and oceans, with a total annual emission of 65 Tg N (Sutton et al., 2013). Nitrogen wet and dry deposition is also influenced by N emissions and is estimated to be around 70 Tg N on terrestrial ecosystems and 30 Tg N on oceans annually (Fowler et al., 2013). In commenting on the major features of the global nitrogen cycle it is worth noting that, of 180 Tg N input through a combination of manufactured fertilizers and biological nitrogen fixation annually, only 28 Tg is available in food human consumption (i.e. 16%), with only 19 Tg (i.e. 11%) actually consumed, given levels of food waste prior to consumption.

Agriculture is one of the major contributors to the alteration of the global N cycle, especially in the production of N_2O and NH_3 emissions. The extensive use of synthetic N fertiliser, produced by the Haber–Bosch process, has sustained the increase in agricultural production and has provided food for a growing population (Erisman et al. 2008), while introducing substantial Nr inputs into the environment and significantly disturbing the natural N cycle, as discussed in Section 1.1. Bouwman et al. (2013) shows that in the beginning of the 20th century global nutrient budgets were either balanced, or surpluses were small. Between 1900 and 1950 global soil N surplus almost doubled to 36 Tg y^{-1} (Table 1.2.2). Between 1950 and 2000, the global surplus increased to 138 Tg y^{-1} of N. As N use in agriculture increases, so too do emissions of N_2O and NH_3, with an increase of 233 and 600% from the period 1900-2000 for these species, respectively (Table 1.2.2).

Table 1.2.2 *Global input terms (fertilizer, manure excluding NH_3 emission from animal houses and storage systems, biological N_2 fixation, and atmospheric N deposition), soil budget (total, arable land, and grassland) and the various loss terms for N [NH_3 volatilization, denitrification (excluding N_2O and NO), and N_2O and NO emission], nitrate leaching and runoff for 1900, 1950, 2000, and 2050 (predicted) (Table 1 in Bouwman et al., 2013).*

Input/output balance term	1900	1950	2000	2050
N Inputs				
N fertilizer	1	4	83	104
N manure	33	48	92	139
N_2 Fixation	14	23	39	54
N deposition	6	13	35	49
Total N inputs	**54**	**89**	**248**	**347**
N Fate/losses				
N withdrawal (plant uptake)	34	52	110	176
Soil N budget	20	36	138	170
Arable land	6	12	93	119
Grassland	14	24	45	52
NH_3 volatilization	4	7	24	36
Denitrification (N_2)	6	12	48	55
N_2O emission	3	4	7	9
NO emission	1	1	2	3
N leaching + runoff	6	12	57	68
NH_3 emission from animal houses and storage systems	2	4	10	15

Global crop production is often seen as the primary accelerator of N cycles in agriculture; however, the demand for animal feed produced from different crops and by-products of the food industry has rapidly increased in the past century (Figure 1.2.3). Livestock feed production systems are the largest cause of human alteration of the global N cycles. Grasses provide more than 70% of the global protein intake by animals and two-thirds of the remaining protein is supplied by feedstuffs and one-third by products like kitchen wastes. At present, about 30% of global arable land is used for producing animal feed, also involving a similar fraction of fertilizer use. In addition, total N in animal manure generated by livestock production exceeds the global N fertilizer use (Bouwman et al., 2009).

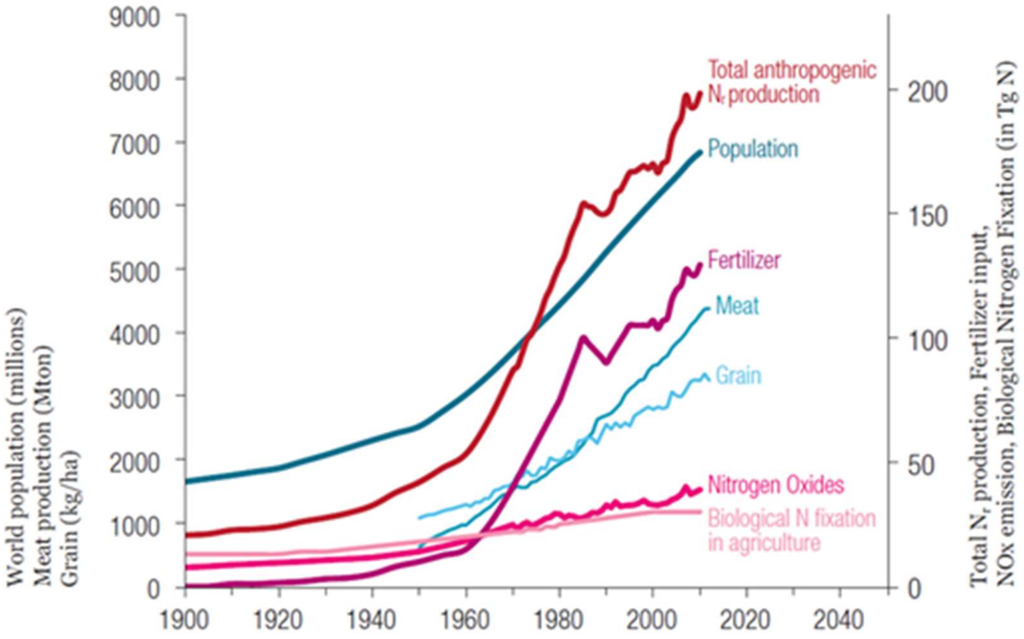

Figure 1.2.3. Global trends between 1900 and 2012 in human population and total anthropogenic reactive nitrogen creation throughout the 20th century (Erisman and Larsen, 2013).

It is relevant to compare the global N budget with the similar one established by the European Nitrogen Assessment (Figure 1.2.4, Sutton et al., 2011). In that study, 85% of the N from crop and grass production was consumed by livestock, with only 15% available for direct human consumption. The global fraction consumed by livestock is similar at 82%, with 18% estimated to be available for direct consumption by humans (Sutton et al., 2013). This emphasizes how, like the European cycle, the global nitrogen cycle is also dominated by humanity's use of Nr to raise livestock. Globally, a smaller fraction of the Nr in food comes from livestock than in Europe. In Europe, 53% of domestic Nr in food comes from livestock, while the global estimate (excluding marine fish) is only 27%. The apparent inconsistency relates to a lower estimated nutrient use efficiency for nitrogen (NUE) for livestock on the global scale. Livestock NUE indicates that only 6% of the Nr consumed by livestock globally reaches human food (prior to food waste), as compared with 19% in the European estimates.

Globally, 43% of the direct Nr inputs to agricultural soils (manufactured fertilizer, biological N fixation, atmospheric deposition and here including livestock manure) reach harvests and biomass production for consumption (feed and food), with the matching crop NUE figure for Europe being 58%. These values are much higher than the values given above for NUE for animal production, emphasizing the critical role of livestock in the low overall values NUE along the agri-food chain. If we consider all sources of anthropogenic Nr production, including NO_x emissions, fertilizer manufacture and agricultural biological nitrogen fixation (excluding natural and marine fixation), then this amounts to 227 Tg N per year. This may be compared with 19 Tg N that is actually consumed by people (accounting for food waste). Overall, this provides a full-chain NUE from all anthropogenic sources of Nr at 8%, emphasizing the necessity for, and the huge potential of, different options to improve the efficiency of Nr use.

Finally, it is worth noting that a substantial fraction (44%) of the Nr emitted to the atmosphere as NOx and NH_3 (113 Tg N per year) is estimated to be recycled back to agricultural cropland. While this provides a significant contribution to agricultural Nr inputs (22%) and productivity, it must be recognized that direct fertilization at the right time, and in the correct quantity, is more efficient, while avoiding the multiplicity of adverse effects.

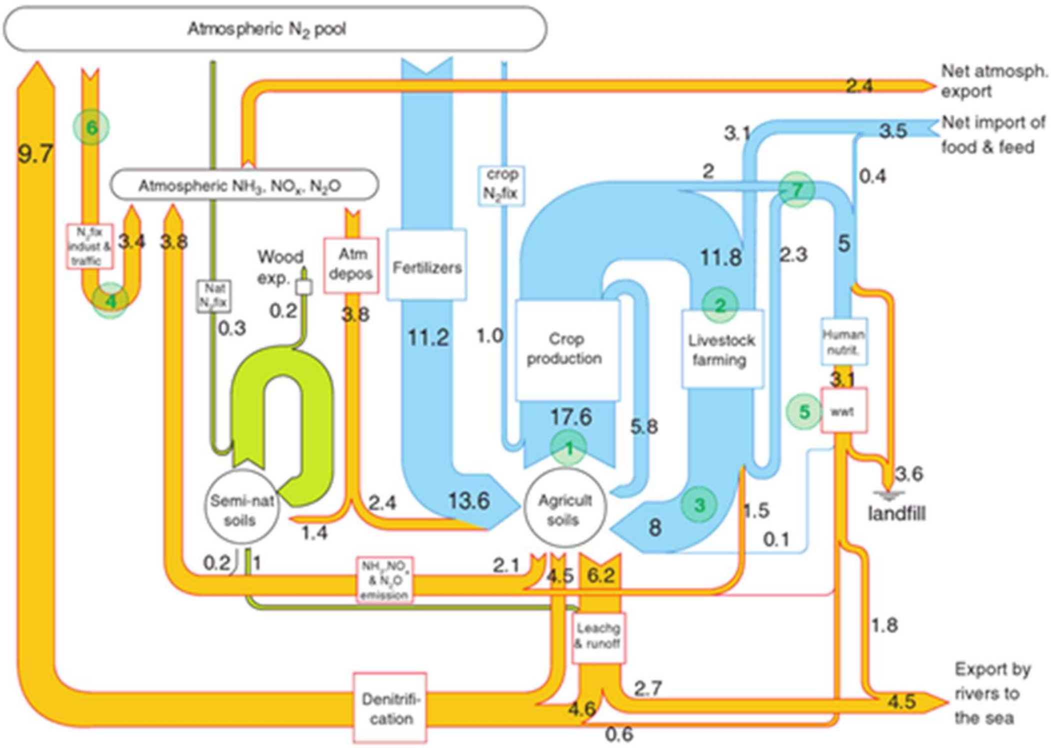

Figure 1.2.4. The N cycle at the scale of EU-27 for the year 2000. Fluxes in green refer to 'natural' fluxes (to some extent altered by atmospheric N r deposition), those in blue are intentional anthropogenic fluxes, those in orange are unintentional anthropogenic fluxes. The numbered green circles indicate a package of seven key actions for overall integrated management of the European nitrogen cycle (see: Sutton et al., 2011)

Beyond agricultural emissions, NO_x (both NO_2 and NO) has been among the air pollutants of greatest concern and under regulation in many countries for decades. NO_x remains a major contributor to air pollution due to rapid increases in fossil fuel consumption and uneven development among countries. Like most other major air pollutants, emissions of NO_x has increased over the past several decades; however, it is slowing (Figure 1.2.5). It was estimated that the global total annual emissions of NO_x from combustion and industrial sources in 2014 was 129 Tg (approx. 39 Tg N) (Huang et al., 2017). A large contributor to global production of NO_x is coal consumption and industry in Asia (Figure 1.2.6), where actions are now being taken to reduce emissions. Nations (China in particular) have continuously decreased coal consumption since 2014, and serious and in some nations (i.e. China) effective efforts have been undertaken toward denitration at all major power plants to reduce NO_x production.

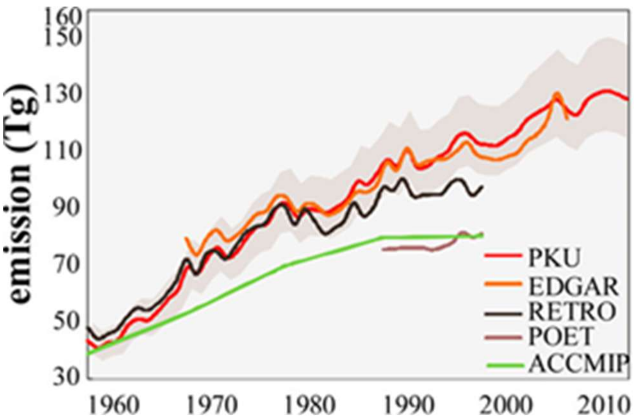

Figure 1.2.5 *Temporal trends in the total anthropogenic emissions of NOx from 1960 to 2014 as estimated by several models. Interquartile ranges (25th to 75th) from the Monte Carlo simulation are shown in shadows (Figure 3 in Huang et al., 2017; missions reported in molecular mass, to convert to TgN multiply by approx. 0.3).*

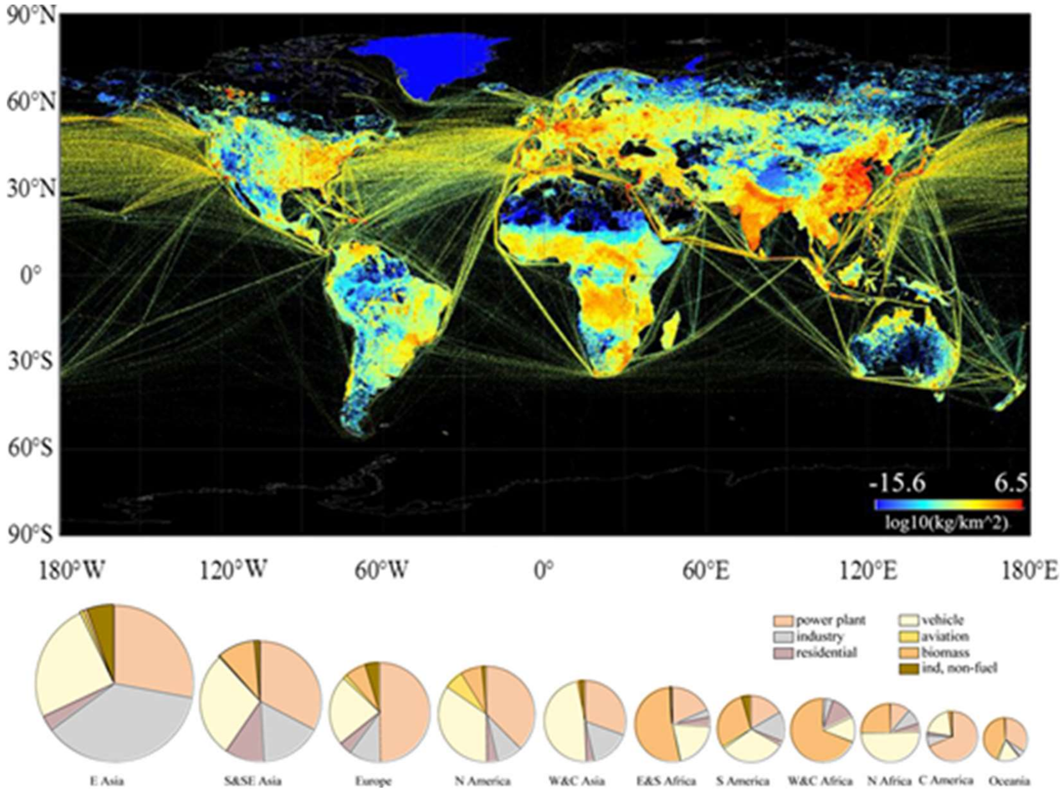

Figure 1.2.6. Geospatial distributions of NOx emission densities, with the exception of aviation in 2014. The source profiles of the emissions in various regions are shown as pie charts (Figure 2 in Huang et al., 2017)

Food waste contributes significantly to N waste/pollution in terms of both disposal and as a form of inefficiency in agricultural production. The proportion of food that ends up as waste is a key factor in determining the nitrogen requirements of growing a sufficient harvest. It is estimated that 8.3% of food is wasted at or around harvest and 7.0% during farm-stage post-harvest activities (WWF, 2021a). When food waste is high, farmers must raise levels of production, which in turn increases the amount of N required. In addition, if a large proportion of supplied food is lost from the supply chain to the environment, then the excess N in the wasted food can lead to a number of negative impacts on the ecosystem and human health, whilst also having considerable economic impacts for the food industry itself. It is estimated that global food waste on farms amounts to 1.2 billion tonnes per year, which represents a waste of approximately 15.3% of food produced globally, with a total value of $370 billion (WWF, 2021a).

It is not only the nitrogen implications of food waste that are of concern: food waste is estimated to contribute around 6 - 10 % of total anthropogenic greenhouse gas emissions (Ritchie, 2020, WWF 2021a). As food decomposes it produces methane (CH_4), a potent greenhouse gas. Grizzetti et al. (2013) estimate that the food lost at the consumption stage equates to 9% of total global food consumption. However, food is wasted at all stages of the supply chain, across its growth, harvest, storage, retail, and at final consumption. In particular, waste at the production and handling and storage phases can be significant contributors to overall losses. Food waste varies geographically and is largely dependent on diet, agricultural management and the availability of food (Figure 1.2.7). Weather (i.e. heat) and economic factors (access to refrigeration and adequate storage) also play into food waste. Where food is plentiful and consumers are more concerned with appearance and quality of the food they eat, losses are typically higher at the consumption end of the life cycle (i.e. USA and Europe). Alternatively, where food is in a more limited supply and access to refrigeration is less available, food losses tend to occur more in the production and handling/storage phases before it reaches the consumer (i.e. Africa and Southeast Asia) (Figure 2.2.1).

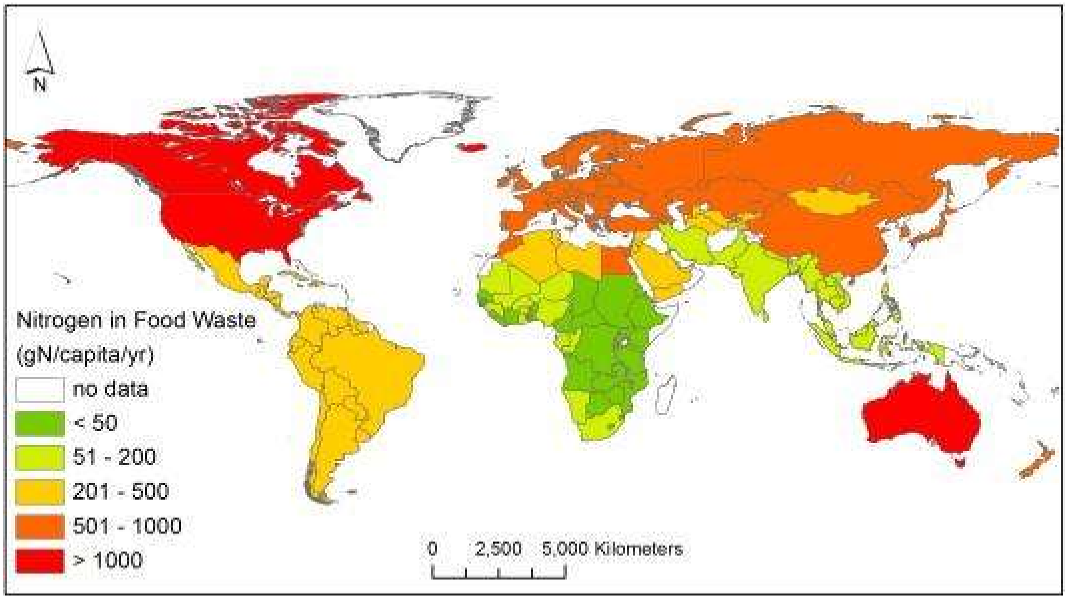

Figure 1.2.7. Nitrogen loss in food waste at consumption (gN per capita per country, 2007; Figure 1 in Grizzetti et al., 2013).

1.2.1 Methane

Roughly one-fifth of the increase in radiative forcing by human-linked greenhouse gases since 1750 is due to methane. The surface dry air mole fraction of atmospheric methane (CH_4) reached 1857 ppb in 2018 (Figure 1.2.8), approximately 2.6 times greater than its estimated pre-industrial equilibrium value in 1750. This increase is attributable in large part to increased anthropogenic emissions (Figure 1.2.9) arising primarily from agriculture (e.g., livestock production, rice cultivation, biomass burning), fossil fuel production and use, waste disposal, and alterations to natural methane fluxes due to increased atmospheric CO_2 concentrations and climate change (Ciais et al., 2013). The past three decades have seen prolonged periods of increasing atmospheric methane, but the growth rate slowed in the 1990s, and from 1999 to 2006, the methane concentrations were near constant. Yet strong growth resumed in 2007. The reasons for these observed changes remain poorly understood because of limited knowledge of what controls the global methane budget.

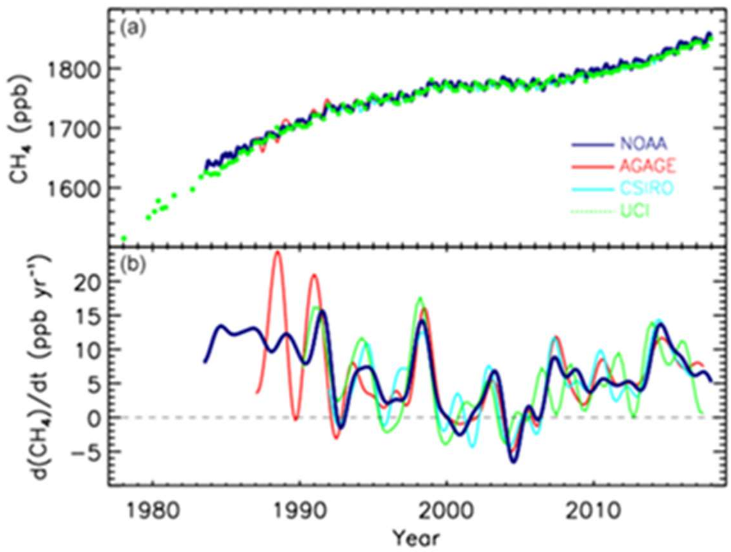

Figure 1.2.8 Globally averaged atmospheric CH_4 (ppb): (a) and its annual growth rate GATM (ppb yr^{-1}); (b) from four measurement programmes, National Oceanic and Atmospheric Administration (NOAA), Advanced Global Atmospheric Gases Experiment (AGAGE), Commonwealth Scientific and Industrial Research Organisation (CSIRO), and University of California, Irvine (UCI) (Figure 1 in Saunois et al. 2020).

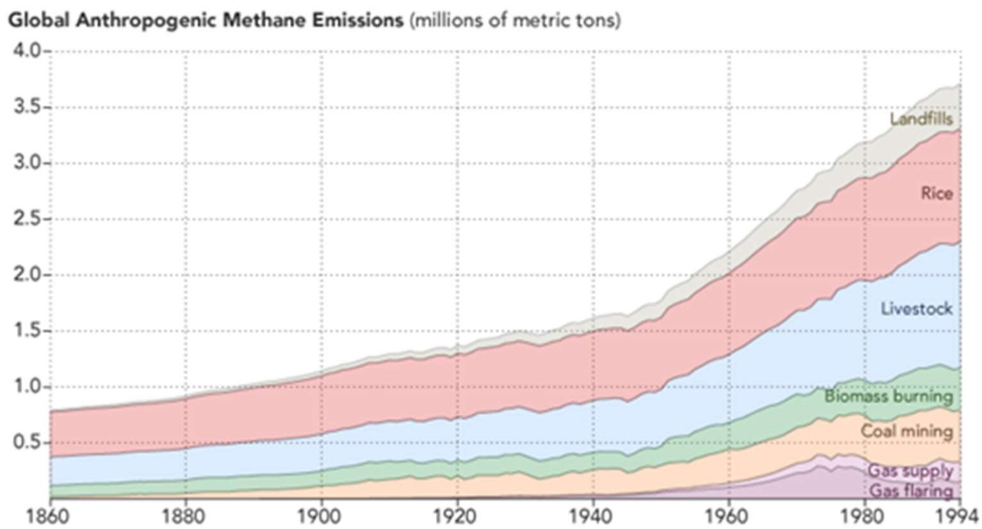

Figure 1.2.9 Methane emissions related to human activity are on the rise. (NASA Earth Observatory image by Joshua Stevens, using data from CDIAC.)

In Saunois et al. (2020), for the years between 2008 and 2017, global methane emissions are estimated by atmospheric inversions (a top-down approach) to be 576 Tg CH_4 yr^{-1} (or 737 Tg CH_4 yr^{-1} using a bottom-up approach) (Figure 1.2.10). Of this total, approximately 60 % is attributed to anthropogenic sources. The mean annual total emission for the decade (2008–

2017) was 29 Tg CH_4 yr^{-1} larger than estimates for the previous decade (2000–2009). Since 2012, global CH_4 emissions have been tracking the warmest scenarios assessed by the Intergovernmental Panel on Climate Change. The latitudinal distribution of atmospheric observation-based emissions indicates a predominance of tropical emissions (∼ 65 % of the global budget, < 30∘ N) compared to mid-latitudes (∼ 30 %, 30–60∘ N) and high northern latitudes (∼ 4 %, 60–90∘ N). The most important source of uncertainty in the methane budget is attributable to natural emissions, especially those from wetlands and other inland waters.

Overall, about two-thirds of global emissions are caused by human activities; the remaining third is from natural sources. The Global Methane Assessment (UNEP/CCAC, 2021) states that more than half of global methane emissions stem from human activities in three sectors: fossil fuels (35% of anthropogenic emissions, consisting of oil and gas extraction, processing and distribution (23%) and coal mining (12%)), waste (20% of anthropogenic emissions from landfills and wastewater) and agriculture (40% of anthropogenic emissions consisting of livestock emissions from manure and enteric fermentation (roughly 32%) and rice cultivation (8%).

Methane sources and sinks vary with latitude. At polar latitudes, methane sources include wetlands, natural gas wells and pipelines, thawing permafrost, and methane hydrate associated with decaying offshore permafrost. In the heavily populated northern mid-latitudes, the main sources are the gas and coal industries, agriculture, landfills, and biomass fires. Tropical wetlands are the world's largest natural source of methane. Emissions from equatorial and savanna wetlands, ruminants, and biomass burning are increased further by tropical anthropogenic inputs (Figure 1.2.10).

The contribution of ruminant livestock to greenhouse gas (GHG) emissions has been investigated extensively at various scales from regional to global, but the long-term trend, regional variation and drivers of methane (CH_4) emission remain unclear. Dangal et al. (2017) estimate that total CH_4 emissions in 2014 were 2.72 Gigatonnes (Gt) CO_2-eq from ruminant livestock, which accounted for 47%–54% of all non-CO_2 GHG emissions from the agricultural sector. These estimates show that CH_4 emissions from the ruminant livestock increased by 332% since the 1890s. These results further indicate that livestock sector in drylands had 36% higher emission intensity (CH_4 emissions/km^2) compared to that in non-drylands in 2014, due to the combined effect of higher rate of increase in livestock population and low feed quality. The study also finds that the contribution of developing regions (Africa, Asia and Latin America) to the total CH_4 emissions increased from 51.7% in the 1890s to 72.5% in the 2010s. These changes were driven by increases in livestock numbers (LU units) by up to 121% in developing regions, but decreases in livestock numbers and emission intensity (emission/km^2) by up to 47% and 32%, respectively, in developed regions.

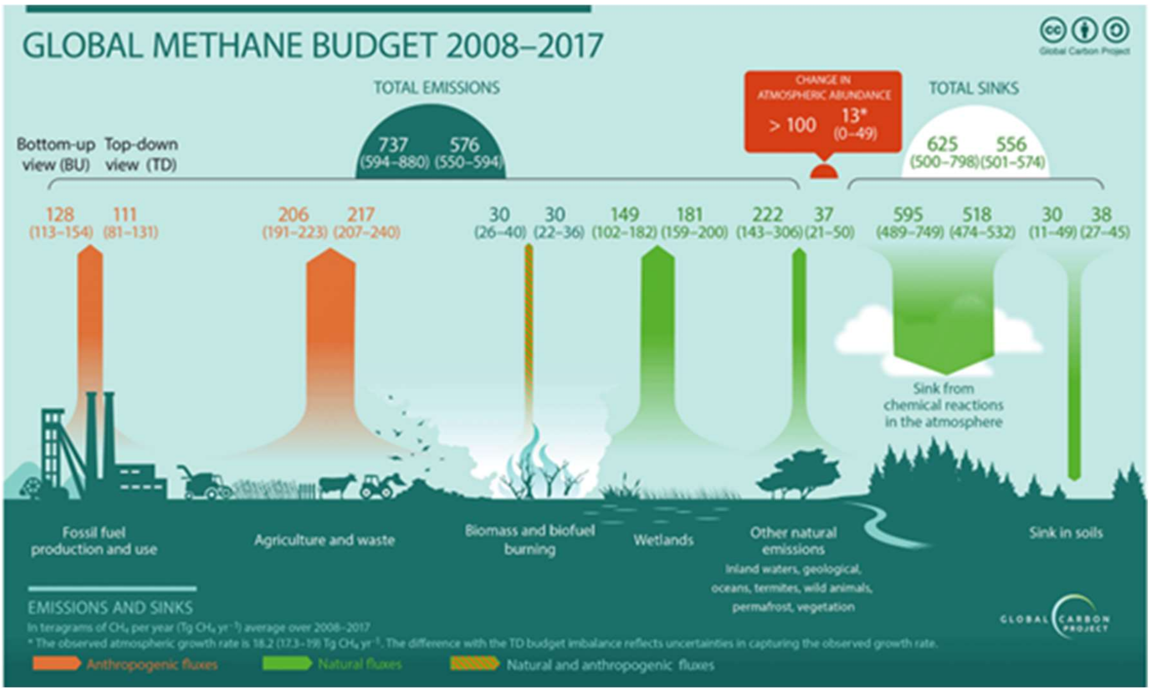

Figure 1.2.10. Global methane budget for the 2008–2017 decade. Both bottom-up (left) and top-down (right) estimates (Tg CH4 yr^{-1}) are provided for each emission and sink category, as well as for total emissions and total sinks. Biomass and biofuel burning emissions are depicted here as both natural and anthropogenic emissions (Figure 6 in Saunois et al., 2020).

Methane emissions in the UK have fallen significantly since the 1990s (a drop of over 60%). This is largely due to increasing the efficiencies of industry and combustion processes, but the most significant fall in emissions has been that of waste management (i.e., food waste and landfill) (Figure 1.2.11). While industrial and waste management emissions have largely fallen over the past four decades, emissions from livestock have remained fairly steady, primarily that of enteric fermentation (cows and sheep). In 2018 UK total methane emissions were 2060 kt CH_4 (BEIS, 2020b).

Whilst over the past few decades, the UK has seen a major reduction in the emissions of methane from landfill (Brown et al., 2021) as a result of improved landfill methane capture technology, continued efforts to reduce food waste sent to landfill will bring the co-benefit of helping to reduce the UK's carbon footprint.

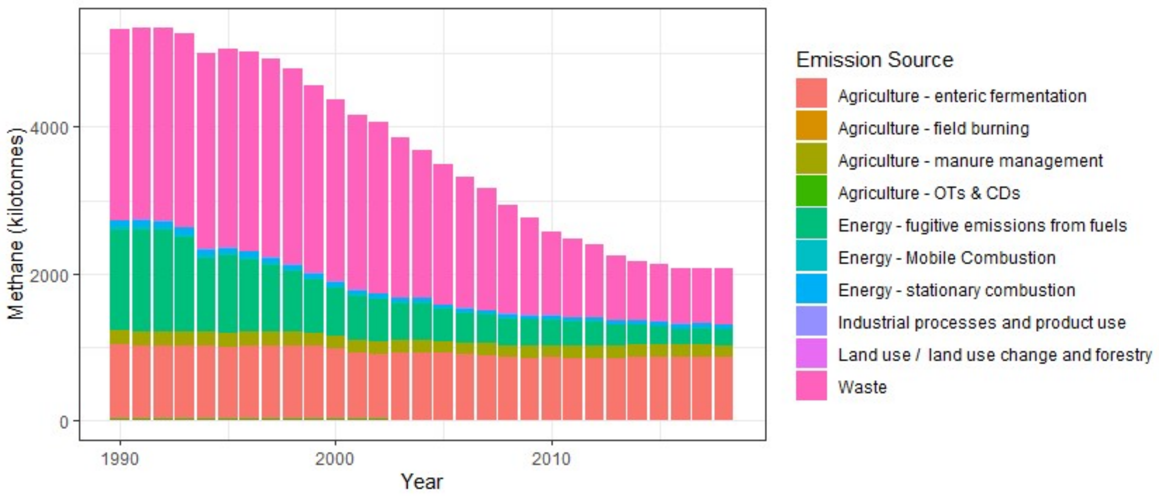

Figure 1.2.11. UK Scale emissions CH_4, from the period 1988 to 2018 as reported by the UK NAEI (https://naei.beis.gov.uk/).

1.3 Nitrogen and global trade

International trade of agricultural products (i.e. with embedded N as described below) has increased by a factor greater than 10 during the past six decades (Figure 1.3.1), and it is expected to continue to grow in the future, further increasing landuse changes, greenhouse gas (GHG) emissions, and reactive N losses (Schmitz, *et al*. 2012). The demand for more protein rich food in regions with increasing population has been a major driver for this change, together with the reduction of costs. Global meat consumption increased by 58% between 2008 to 2018, predominantly driven by population growth, but also as a result of increasing preference and income growth (Whitnall and Pitts, 2019). If this continues, the meat industry would need to increase production by an estimated 50-73% by 2050 to keep up with demand (Bonny *et al.* 2017). The increased opportunities of food and feed trading, as well as the availability of synthetic N fertilisers providing an alternative to manure, have allowed the large regions to specialise in either crop or livestock farming, often creating a disconnection between both (Naylor et al. 2005; Billen et al. 2010). Thus, the N needs of large territories can be sustained by the application of synthetic fertilizers without the need for animal manure, or areas are able to specialise in meat and milk production sustained by feed imports from overseas rather than grown locally (Lassaleta, et al., 2014). This means that the balance of nutrient cycling on a global level has become increasingly distorted and the value of manure as a fertiliser and soil conditioner has been diminished.

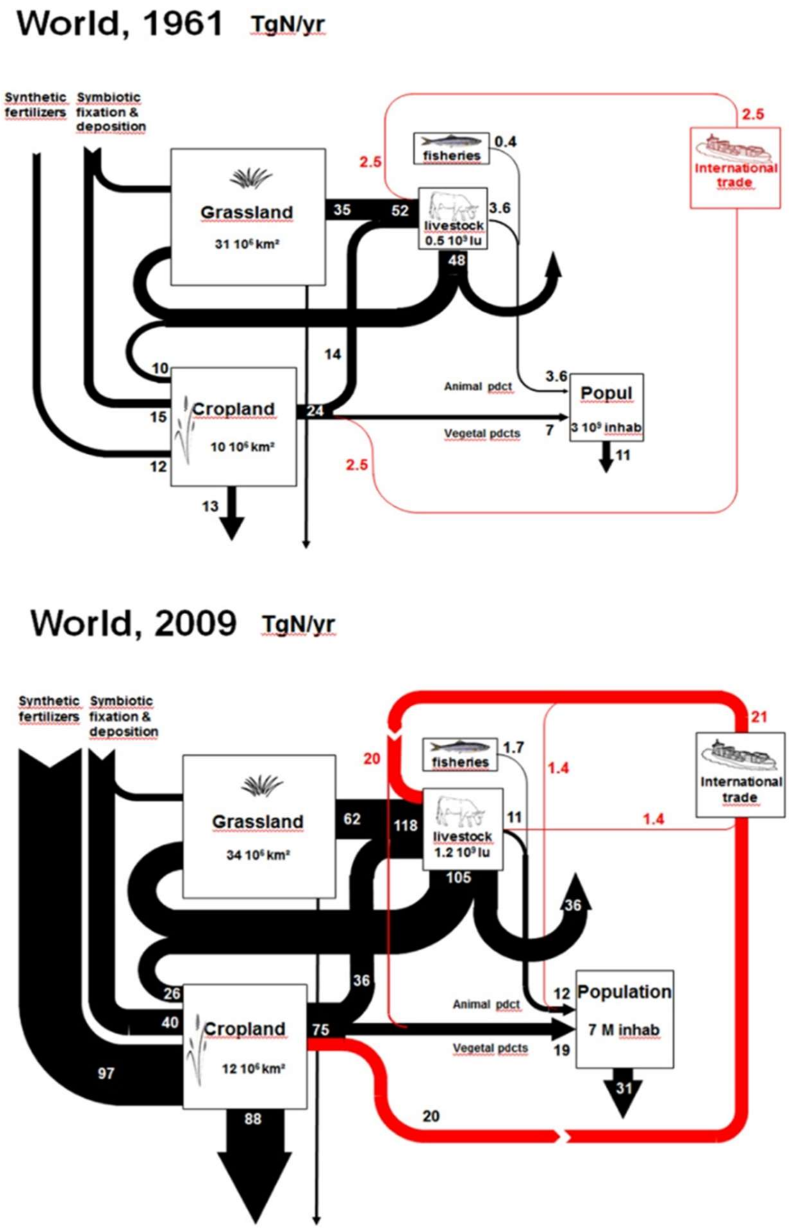

Figure 1.3.1. Generalized representation of N transfers through the world agro-food system (GRAFS) in 1961 and 2009 (Figure 1 in Lassaletta et al., 2016).

The trading of food and feed has also altered the input of new reactive N brought to a region, in a way similar to the N input associated with synthetic fertilization. As food and feed contain proteins, and proteins are composed of approximately 16% N, the trade of food and feed also results in a movement of N from one region to another (Grote et al. 2005; Burke et al. 2009; Swaney et al. 2012). To complicate matters, the influence of trading on N global fluxes is more complex than the N flows associated with import or export of food and feed. During the crop and animal production process, part of the reactive N added by fertilisers and feed, respectively, is lost to the environment, potentially impairing air and water quality, biodiversity and affecting GHG balance. This input of N required for the production of a commodity that is not embedded in the final consumed good has been defined as ''virtual nitrogen'' (Galloway et al. 2007; Burke et al. 2009; Leach et al. 2012). Virtual nitrogen includes all N that is released to the environment throughout the entire food production process but that is not contained in the final consumed food product. Most of the N losses to the environment, and therefore their negative impacts, are located where the goods are produced. So, the magnitude of the commercial exchanges of agricultural products complicates the localization of the environmental losses of reactive N linked to agriculture and food production. Trading may distance the environmental impacts far away from the place where the goods are consumed. Some world regions can receive large amounts of reactive N in the form of food and feed (Billen et al. 2010; Lassaletta et al. 2012 & 2013) while the negative consequences of the production of these imported commodities remain in the producer countries.

Countries' status as net N importer or exporter (1961 – 2010) is shown in Figure 1.3.2. With only a few exceptions (like South Africa), the behaviour of most individual countries has not changed over 50 years, but previous trends have strongly intensified: i.e. many countries have evolved from near equilibrium between nitrogen imports and exports in food and feed to a much more unbalanced situation. The number of highly net exporting countries has not increased very much, but the amount of N exported increased from 1,645 to 15,322 GgN/year between 1961 and 2010 (Lassaletta et al., 2014). For example, Argentina's net exports totaled 197 GgN in 1961 and 3261 GgN in 2010, and Brazil net exports totaled only 4 GgN in 1961 but has become the third most exporting country of the world with 2,778 GgN exported in 2010 (dominated by soy, discussed below). In contrast, the number of high net importing countries has significantly increased. Only 10 countries were net importers of more than 50 GgN in 1961, whereas in 2010 the number of such countries has increased to 47. Several Mediterranean, American and Asian countries are now very far from a balanced situation, net importing a large amount of food and feed. This analysis reveals a world with increasing specialization and interdependence, where quite a small number of highly productive countries (e.g., USA, Argentina and Brazil) are supplying protein to an increasingly large number of dependent countries (e.g., China, Japan, Mexico, Spain and Egypt) (Lassaleta, et al., 2014).

China has changed from a small net exporting country to a highly unbalanced country that imports large amounts of feed. The other regions have significantly intensified their imbalances. In addition to China, the most remarkable increases correspond to S.-E. Asia, C. & SW. America and Maghreb–M. East since they have increased their net import nine-, four- and threefold, respectively. On the other hand, S. American Soy Countries have increased their nitrogen exportation fivefold. Only Europe and Oceania have remained at a constant level of net exporting and net importing, respectively. In the UK, it is estimated that 295 to 428 kt N is imported into the country annually, predominantly in the form of food and animal feed products (Worrall et al. 2016; Fan et al., 2020; Sutton et al., 2011). The UK is a net importer of embedded nitrogen emissions, particularly from: the Netherlands, Germany, China, Ireland and France (Oita et al, 2016). Regarding the fluxes between regions (Figure 1.3.3), changes between 1986 and 2009 can be observed not only in the magnitude, but also in the origin and destination of the N trade. The main net exporters have significant agricultural, food and textile exports, and are often developing countries, whereas important net importers are almost exclusively developed economies, with the result that substantial local nitrogen pollution is driven by demand from consumers in other countries (Oita et al., 2016).

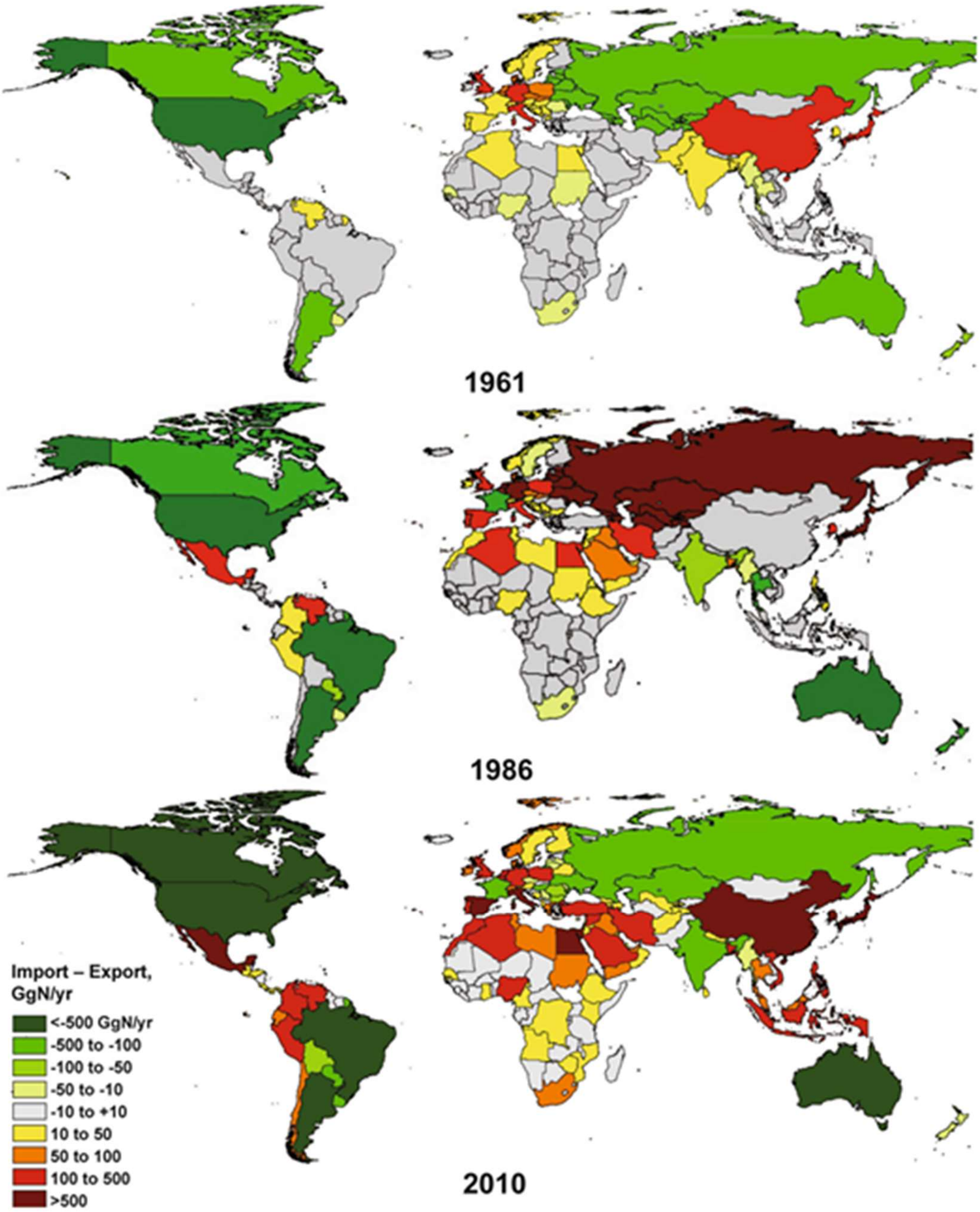

Figure 1.3.2 Net import or export of N embedded in traded commodities for each country for the years 1961, 1986 and 2010. Green countries = exportation is higher than the importation, i.e. net exporting N; yellow–red countries = countries that are net importing N; grey countries = imports and exports are balanced. Figure 2 in Lassaletta et al. (2014).

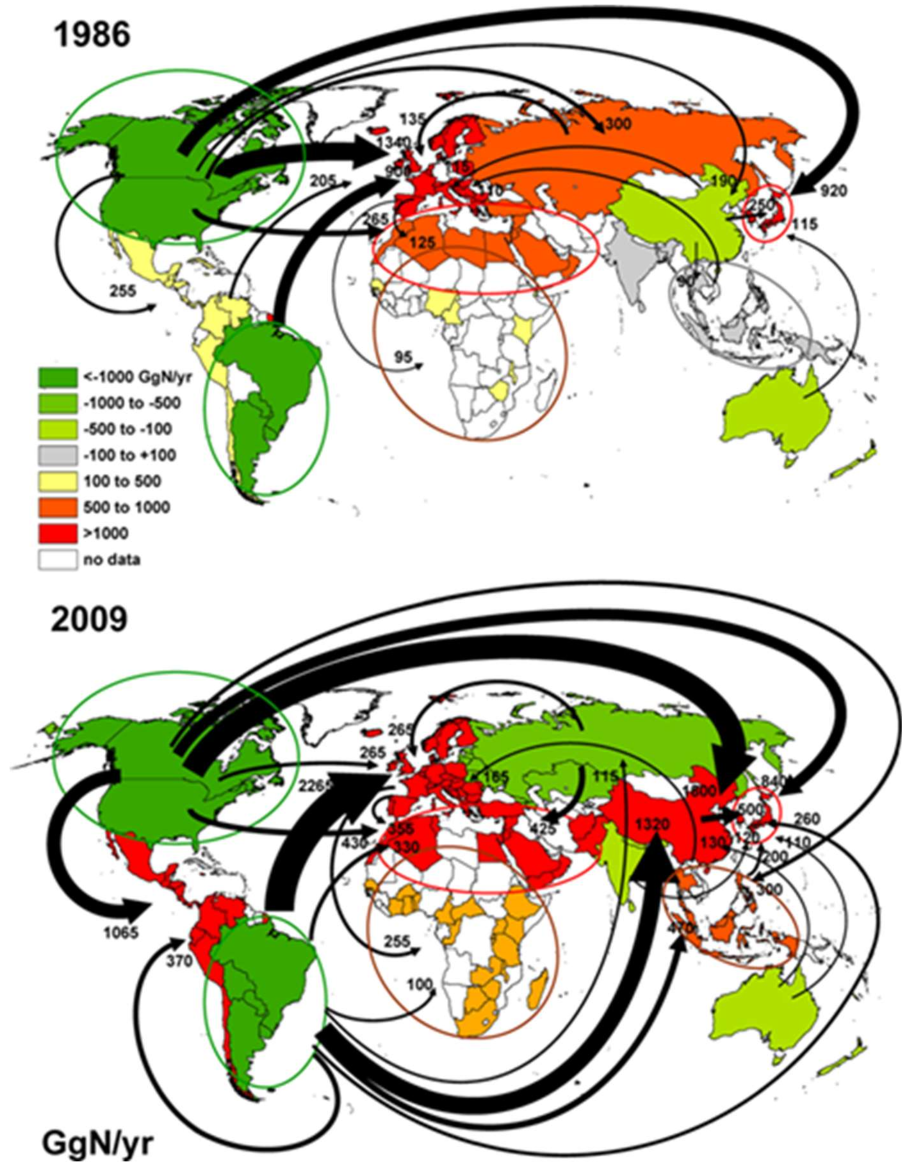

Figure 1.3.3. N fluxes from each region to the others for the years 1986 and 2009. Arrows show the fluxes between the regions (only fluxes higher than 90 GgN are represented). Figure 3 in Lassaletta et al., 2014).

As well as the quantity of international trading of N increasing throughout the past few decades, the NUE of cropping systems (i.e. the efficiency of N used) in different countries has also changed drastically. The pollution generated in different parts of the world, and the impact it has, is a complex topic, beyond the scope of this report. However, some general trends and observations are reported in literature. The overall NUE of a specific region is highly dependent on a number of factors, primarily livestock production and exports, access to N fertilisers, crop type and management practices. While the NUE of some developed countries (e.g. EU) has increased since action taken in the 1980s (Figure 1.3.4), this efficiency has dropped in less developed nations in which fertiliser use is less regulated. This is especially true for nations in which artificial fertilisers have become introduced and heavily subsidised, and are not well understood by the farming communities that apply them (i.e. China and India) (Figure 1.3.5), although there are now national and international efforts to change this situation (see Section 3.1).

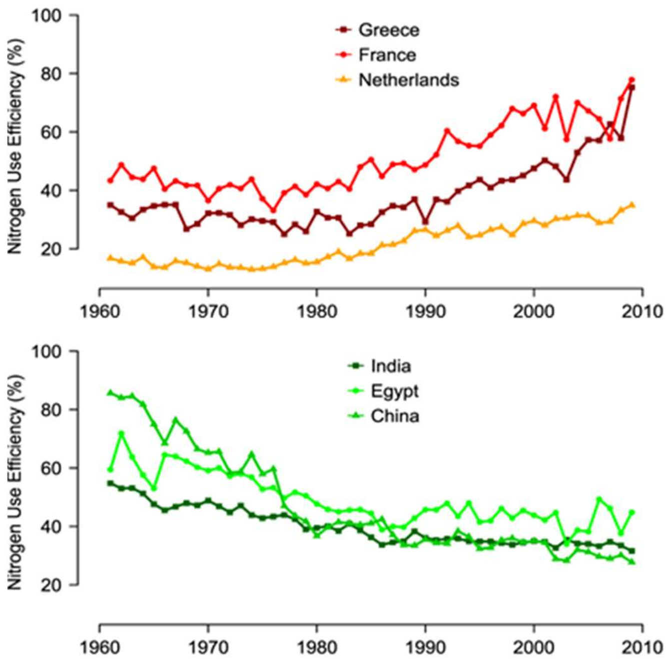

Figure 1.3.4. Fifty year trends in N use efficiency of the cropping system of a number of countries. (Edit of Figure 3 in in Lassaletta et al., 2014)

As well as fertiliser use and farming practices, crop type plays a significant role in NUE and N pollution. As described above, soy exports from countries such as US, Brazil, Argentina, etc., have grown rapidly and account for a large proportion of international N exports; however, as a legume crop, these plants are able to fix N biologically (BNF), and thus have a significantly lower N pollution burden associated with their production. In this regard, exporting N between countries may have a positive environmental effect in that alternative crops to those available locally can be grown more efficiently abroad in many cases. However, this is highly dependent on a large number of factors. In the case of examples such as soy and palm oil exports, the impacts of deforestation and loss of natural habitats as a result of converting land to grow these crops can be extreme, and may outweigh any positive impact in regard to N pollution regardless of NUE.

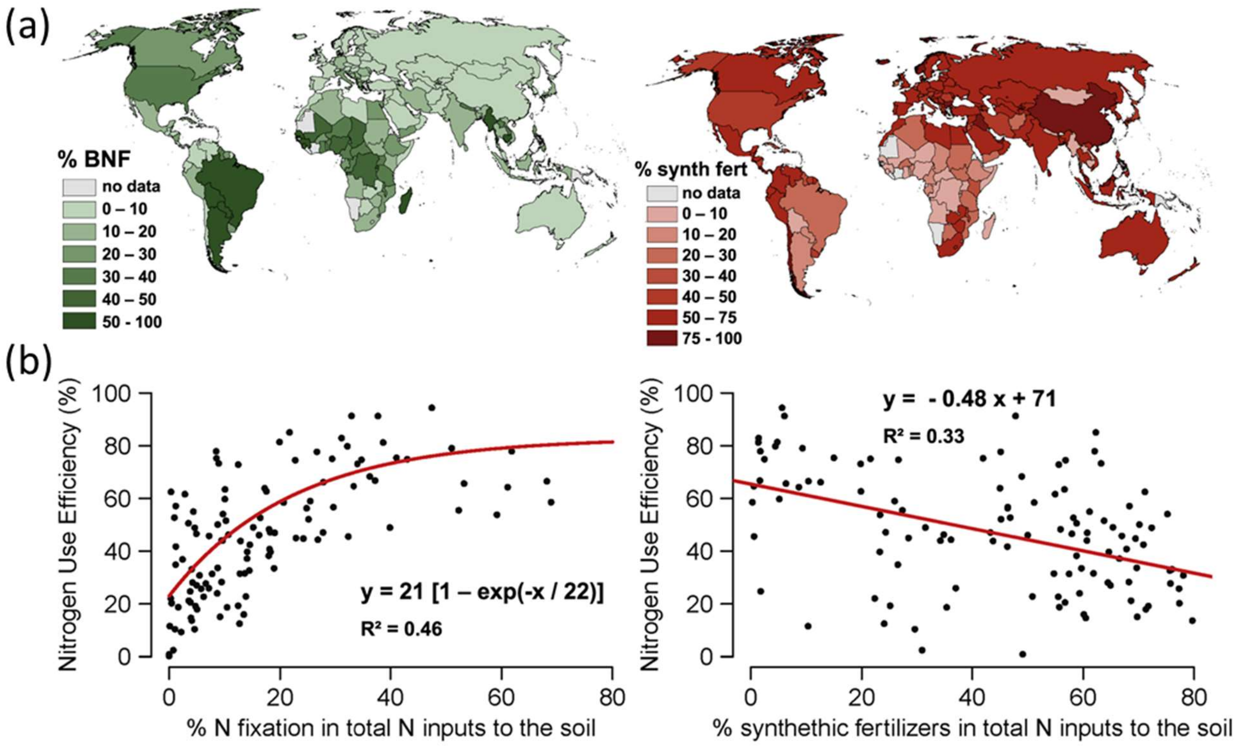

Figure 1.3.5. (a) Distribution of the share of symbiotic fixation and synthetic fertilizers in total N inputs to cropland by countries in 2000–2009. (b) Observed relationship between NUE and the proportion of symbiotic fixation, or of synthetic fertilizers in total N inputs to cropland in the period 2000–2009 (Figure 4 in Lassaletta et al., 2014)

1.4 Nitrogen in the UK

1.4.1 Fertiliser use in the UK

The UK has been using artificial fertilisers heavily since the invention of the Haber-Bosch process (i.e. industrial N fertiliser production) at the beginning of the 20th century (Figure 1.4.1). Based on data collected by the British Survey of Fertiliser Practice (BSFP, 2020), the amount of synthetic N applied as fertiliser in the UK in 2019 was 1038 kt N, with 810, 150 and 79 kt N applied in the England & Wales, Scotland and Northern Ireland regions, respectively. N fertiliser use has remained fairly steady across the UK for the past several years, falling from a peak of 1674 kt N in 1987. Since the 1990s, there has been a gradual reduction in the use of artificial fertilisers; however, the application of organic N fertilisers (i.e. animal waste, sewage and digestates) has increased in England, Northern Ireland and Wales over the past 10–20 years (Figure 1.4.2 & Table 1.4.1). While only a fraction of manures is applied directly as fertiliser via spreading (~21%, Figure 1.4.2), an equivalent amount of N as that applied as synthetic N is contained in the manure produced by farm animals in the UK (1,021 kt N in 2019; Defra, 2021), much of which is returned to the land directly during grazing (Table 1.4.1). These values are dominated by inputs from grazing animals such as cows (64%) and sheep (18%) which recycle a large amount of N back into pastures. In terms of direct spreading, beef and dairy production accounts for the largest source of organic N fertiliser in the UK, with a significant proportion also coming from the application of sewage sludge (i.e. human waste), predominantly in England. England accounts for approximately 77% of N applied in the UK, with a further 7, 10 and 7 % applied in Northern Ireland (NI), Scotland and Wales, respectively. England also applies relatively more synthetic N (79%) than the other countries in the UK (53, 60 and 68 % for NI, Scotland and Wales, respectively) (Figure 1.4.3, Misselbrook and Gilhespy, 2021).

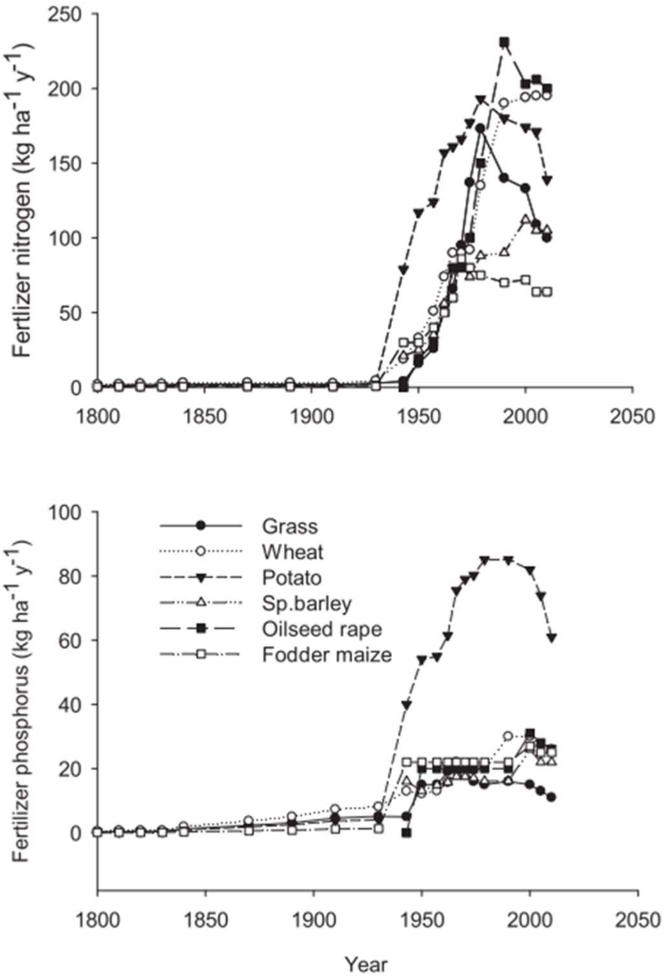

Figure 1.4.1. Historical to current rates of nitrogen and phosphorus fertilizer application rates under grass and crops (Figure 3 in Muhammed et al., 201 8).

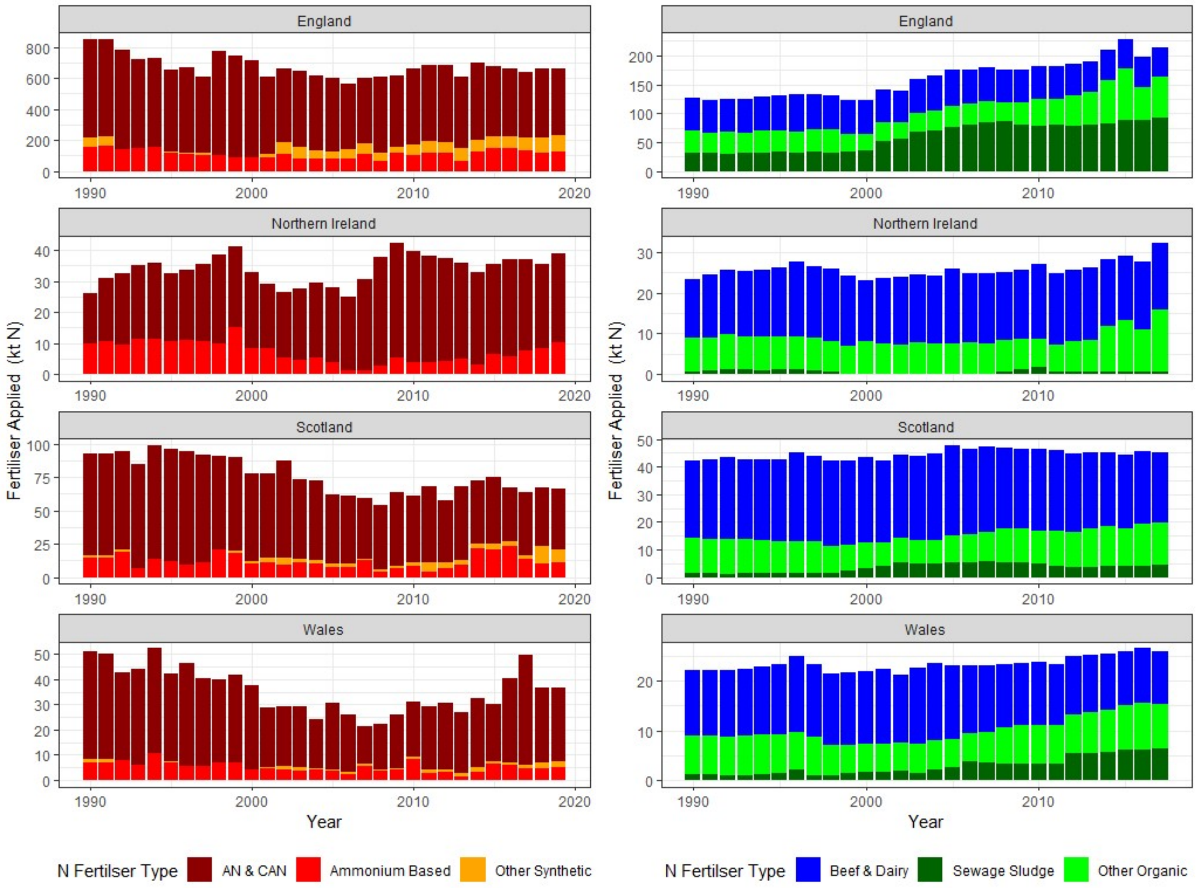

Figure 1.4.2. N fertiliser applied in the UK from 1990 to 2018. Artificial fertilisers (left) are split into ammonium nitrate and calcium ammonium nitrate (AN & CAN), ammonium based (urea and ammonium sulphate) and mix/other classifications. Organic fertilisers (right) are split into Beef and Dairy, Sewage sludge and Other (pig, sheep, digestates, etc…) categories (Misselbrook and Gilhespy, 2021).

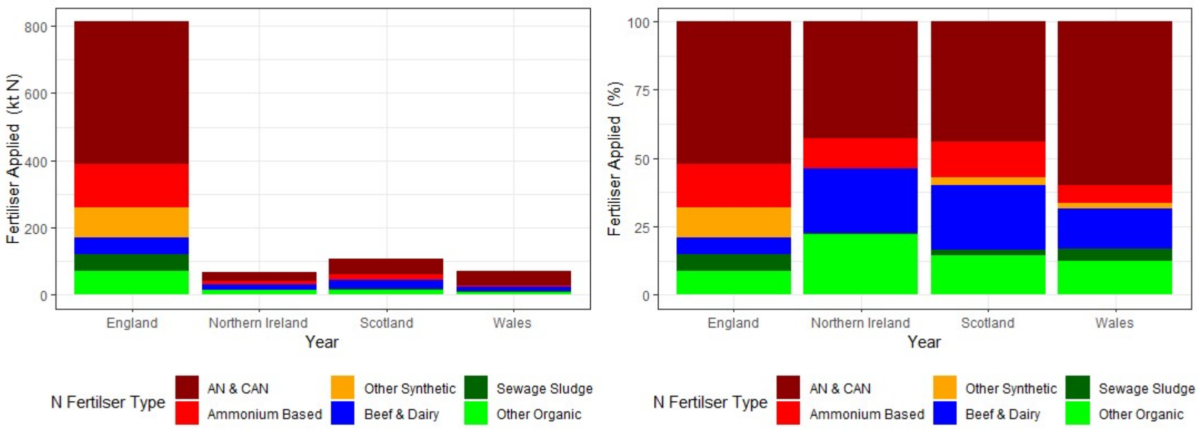

Figure 1.4.3 (Left) The total N applied to agricultural soils in the UK in both synthetic and organic forms and (Right) the proportion of each fertiliser type applied (Misselbrook and Gilhespy, 2021).

__Table 1.4.1.__ UK annual manure production in kt N (Defra, 2021).

Year	Livestock Manure Production	Cattle	Pigs	Sheep and goats	Poultry	Other Livestock
1990	1264	846	84	236	94	5
1995	1249	824	86	233	99	7
2000	1196	773	73	227	116	7
2001	1135	737	67	201	123	7
2002	1104	722	63	197	115	8
2003	1106	726	57	196	119	7
2004	1112	728	58	197	121	8
2005	1096	727	55	191	115	8
2006	1091	721	56	188	117	9
2007	1056	700	55	182	109	9
2008	1033	685	54	177	109	9
2009	1014	676	54	169	106	9
2009	1000	674	51	166	102	7
2010	1015	681	51	167	108	8
2011	1004	670	51	168	108	8
2012	1003	667	51	172	106	8
2013	1008	661	55	177	108	7
2014	1017	664	54	180	111	7
2015	1019	669	54	178	111	7
2016	1033	674	54	183	114	7
2017	1041	673	55	187	120	6
2018	1033	665	55	184	123	6
2019	1021	656	56	181	122	6
2020*	1000	646	55	174	119	6

*provisional estimate

A key feedback mechanism affecting synthetic fertiliser use in the UK is specialisation at the holding level in either crop or livestock farming, as well as a larger-scale spatial segregation with arable farming concentrated in the East and livestock farming in the West of Great Britain and in Northern Ireland (Figure 1.4.4). Regional segregation of arable and livestock farms is largely a result of the large-scale gradients in climate and soil types across the UK (i.e. rainfall), as well as positive-feedback effects of the loss of local infrastructure and organisations required (Martin et al., 2016). Historically, livestock and livestock manure had been integral parts of the arable rotation for nutrient cycling and also provision of draught power, but synthetic fertilisers (and tractors) allowed this spatial segregation to develop.

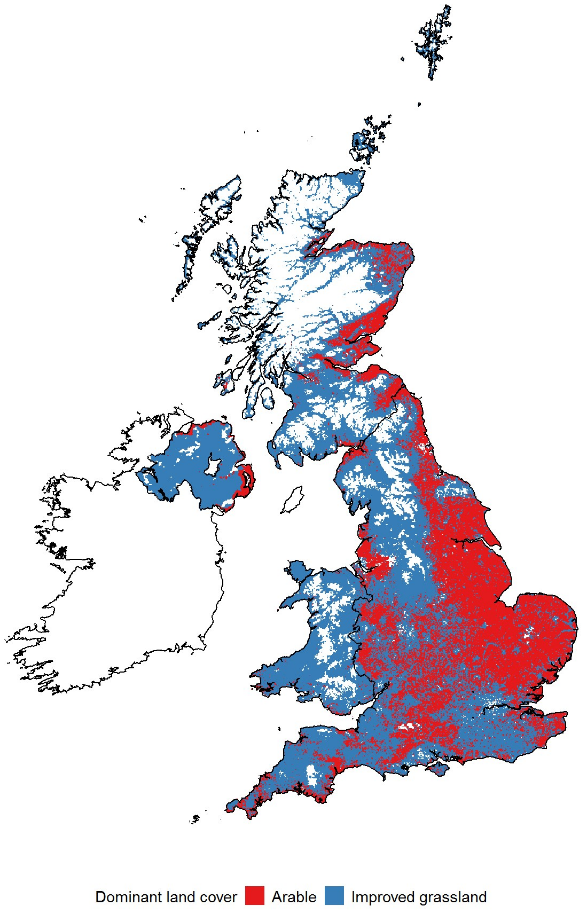

Figure 1.4.4. Dominant type of agricultural land cover for each 1x1km square in the UK, illustrating the large-scale spatial segregation of arable land (red) and improved grassland (blue). Based on data from Morton et al. (2020).

Holding specialisation and larger-scale spatial segregation of arable and livestock farming presents challenges for effectively recycling nutrients from crop-based livestock feeds back to arable land. Livestock manure has relatively low concentration of nutrients per unit mass compared with mineral fertilisers, making the economics of transport over long distances problematic. At an even larger scale, closing the nutrient cycle for animal feed imports from overseas by transport of manure in its natural form is practically impossible. Even where manure can be transported economically, there are other barriers to making full use of livestock manure on arable land, such as limitations on how and when certain kinds of manure can be applied to crops, and the needs of many modern crop varieties for the nutrient release profiles delivered by mineral fertilisers.

The impact of this is a higher level of synthetic fertiliser use on specialised arable farms, and potential for over-application of manure nitrogen (and phosphorus) on grassland and on mixed farms.

1.4.2 N emissions in the UK

The three primary forms of emissions of N pollution (to the atmosphere) are nitrogen oxides (NO_2 and NO, i.e., NO_x), ammonia (NH_3) and nitrous oxide N_2O. NO_x emissions are currently the largest source of N pollution to the atmosphere in the UK, accounting for approximately 250 kt N yr^{-1}. Large reductions in NO_x since the 1970s (Figure 1.4.5, Table 1.4.2) are due to efforts to clean up air pollution and reduce acid rain (where NOx contributes to the formation of nitric acid in the atmosphere). These savings were made by increasing efficiencies in industry/combustion processes, and filtering combustion outputs (i.e. catalytic convertors). As the UK continues to convert to electrification and renewable energies, these emissions will likely fall further in years to come, though there is still much room for improvement. Emissions of NO_x are most intense at source points such as power stations and combustion related industries, as well as cities with heavy traffic. NO_x emissions are therefore highest in large urban areas and on busy stretches of road, with emissions from aviation and shipping as well (Figure 1.4.8).

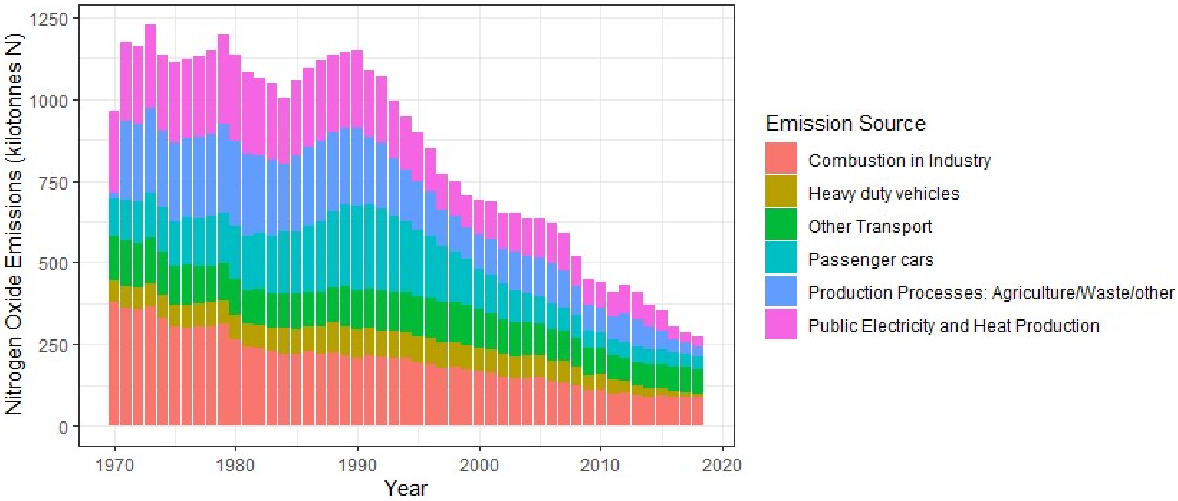

Figure 1.4.5. UK Scale emissions NOx, from the period 1988 to 2018 as reported by the UK NAEI (https://naei.beis.gov.uk/).

Emissions of NH_3 across the UK are approximately 228 kt N yr^{-1}. Unlike NO_x, emissions of NH_3 have remained fairly steady over the past 40 years, and after a slight reduction in the years around 2010, it is back on the rise (Figure 1.4.6, Table 1.4.2). NH_3 emissions are largely associated with agricultural practices and food production, and are not so easy to mitigate without financial investment or technological advancement. Emissions of NH_3 from ammonium based artificial fertilisers such as urea are sizable, as well as a large contribution from the livestock industry and the production and management of animal waste. Emissions are highest in rural areas, especially where livestock production is concentrated (Figure 1.4.8).

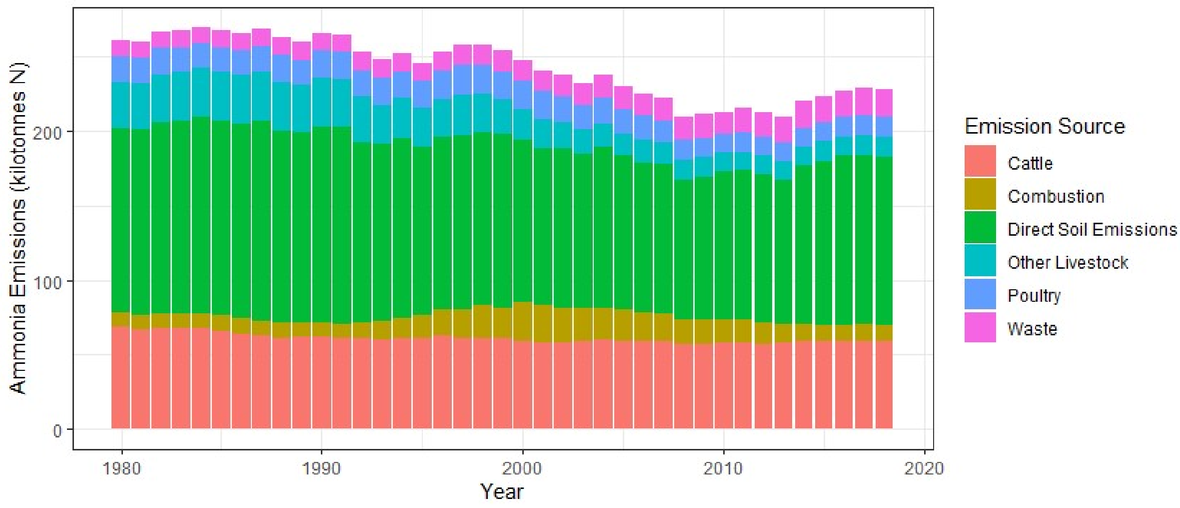

Figure 1.4.6. UK Scale emissions NH_3, from the period 1988 to 2018 as reported by the UK NAEI (https://naei.beis.gov.uk/).

The primary GHG of concern regarding N pollution is that of nitrous oxide (N_2O). With a global warming potential (GWP) 265 times that of an equal volume of CO_2 (IPCC, 2014), and an atmospheric lifetime of approximately 100 years, N_2O contributes significantly to GHG inventories (contributing 4% of the UK's greenhouse gas emissions in 2014 and 5% in 2019 (BEIS, 2021b)). Historically in the UK, N_2O was primarily released via industrial processes (combustion and manufacturing) and agricultural sources (N fertiliser use and animal waste). UK emissions of N_2O have more than halved since 1990 (163 kt N in 1990 to 70 kt N in 2018, Figure 1.4.7, Table 1.4.2), primarily due to a significant reduction in the industrial sector (a fall of 96%). However, emissions from agricultural sources have changed little during this time, and now dominate UK emissions (69% of all N_2O emissions) (UK NAEI, Figure 1.4.8). N_2O emissions are more dispersed across the UK than those described above, with highest emissions coming from areas with intensive agriculture or combustion activities.

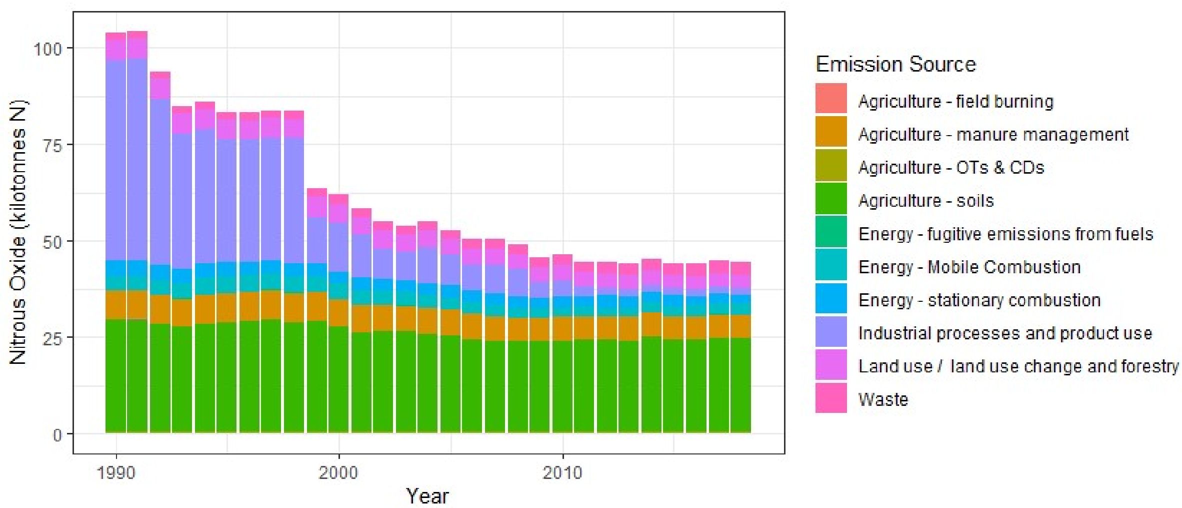

Figure 1.4.7. UK Scale emissions N_2O, from the period 1988 to 2018 as reported by the UK NAEI (https://naei.beis.gov.uk/).

Table 1.4.2. UK annual national emission estimates of NOx, NH_3 and N_2O for the year 2018 in kilotonnes of N (kt N). Data extracted from the UK NAEI (https://naei.beis.gov.uk/).

Sources	Flux N yr^{-1} (kt N)
Nitrogen Oxides (NOx)	
Combustion in Industry	86.7
Heavy duty vehicles	9.7
Other Transport	73.7
Passenger cars	40.7
Production Processes: Agriculture/Waste/other	9.6
Public Electricity and Heat Production	30.0
Total	**250.5**
Ammonia (NH_3)	
Cattle	58.5
Combustion & Production Processes	11.4
Direct Soil Emissions	113.0
Other Livestock	13.0
Poultry	13.6
Waste	18.0
Total	**227.6**
Nitrous Oxide N_2O	
Agriculture - OTs & CDs	0.3
Agriculture - soils	24.3
Agriculture - field burning	0.0
Agriculture - manure management	6.0
Energy - fugitive emissions from fuels	0.1
Energy - Mobile Combustion	2.9
Energy - stationary combustion	2.4
Industrial processes and product use	1.9
Land use / land use change and forestry	3.4
Waste	3.2
Total	**44.4**

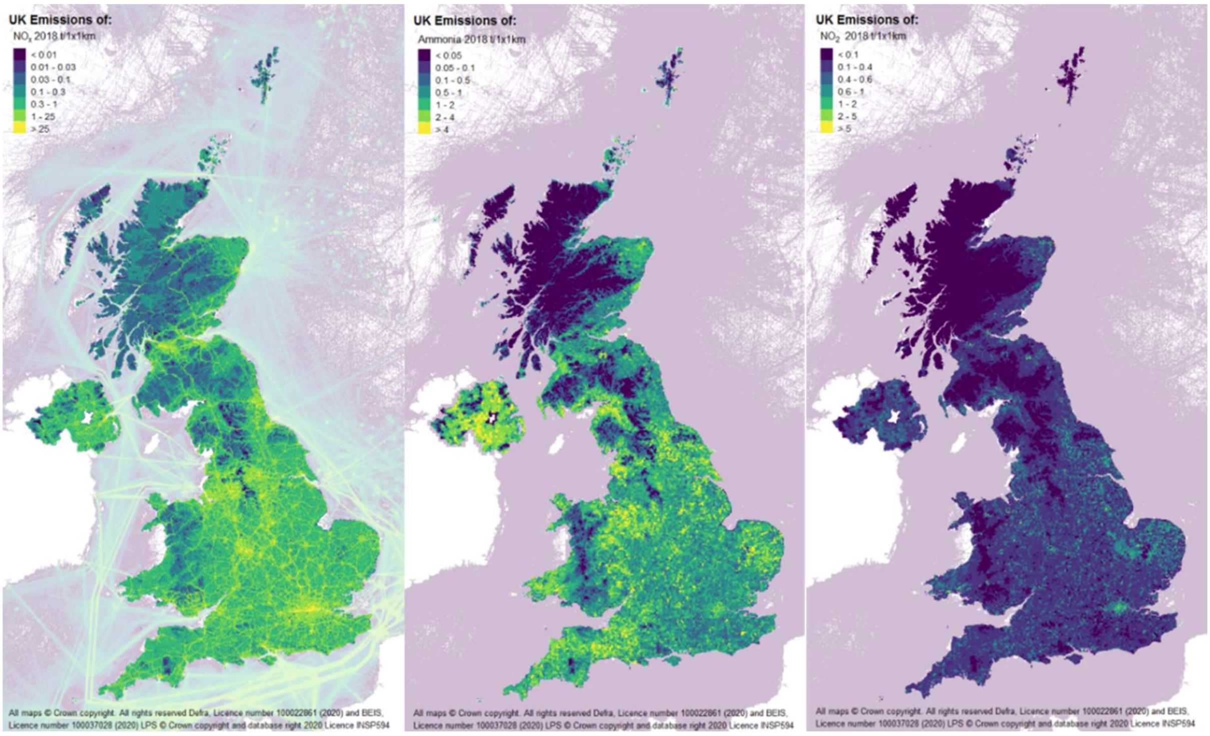

Figure 1.4.8 *Modelled emission maps of (a) NOx and (b) NH_3 and (c) N_2O across the UK for the year 2018 (Figures from NAEI). (Note: typo in Figure on right. NO_2 should read N_2O)*

1.4.3 UK Nitrogen Budget and Footprint

Nitrogen budgets are difficult to establish, and often come with high uncertainties. This is partly due to the complexity of the N cycle, and the cascading effect of Nr in the environment. Nr released due to human activities will quickly integrate with natural Nr pools and can influence natural processes for days to centuries in multiple forms and across vast scales (Figure 1.4.9). The same atom of Nr can move through the environment causing multiple effects in the atmosphere, in terrestrial ecosystems, in freshwater and marine systems, and on human health; a phenomenon that is known as the 'Nitrogen Cascade' (Galloway et al. 2003). The direct influence of human activity can thus become difficult to disentangle or attribute N impacts to. With current data and models, N budgets can be estimated, but uncertainties can be of the same order of magnitude as many of the components of the budget (as high as 200%), so caution is required.

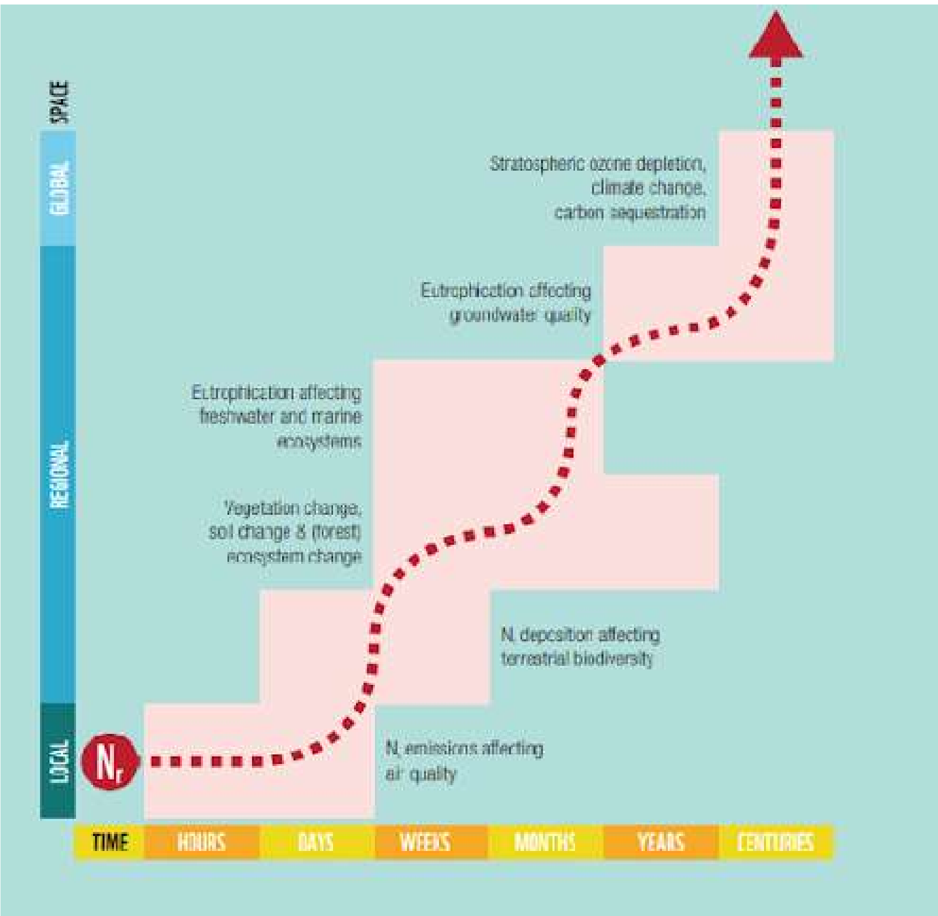

Figure 1.4.9. Nitrogen cascade: This figure shows how the nitrogen cascade amplifies Nr effects through both time and space (taken from Erisman et al., 2015).

In terms of domestic N pollution, studies carried out by Worral et al. (2016) compiled UK scale N inputs and outputs for the years 2012, and estimated a budget for 2020 based on trends for both natural and anthropogenic sources and sinks (Table 1.4.3). This was later built upon by including a detailed spatial map of the UK (Fan et al, 2020). These values are compared with those provided by the European N Assessment that show the total terrestrial to surface water N flux (Sutton et al, 2011) (Figure 1.4.10), a value much smaller than the Worral et al. (2016) or Fan et al. (2020) estimates but one that has been validated in the UK (see below). National Atmospheric Emissions Inventory (UK NAEI) for anthropogenic emissions for 2018 provide details on sources of NH_3, NOx and N_2O emissions at the UK scale (Table 1.4.3). These sources identify that a significant proportion of N waste in the UK is as a result of losses to aquatic systems (i.e. fluvial losses or losses to groundwater, surface water and oceans), which represent N lost in both mineral forms (i.e. nitrates and ammonium) and organic forms (i.e. dissolved organic matter). Losses to aquatic systems account for sources such as sewage, soil erosion and nitrogen fertilisers. Greene et al. (2015) produced a national modelling framework for simulating N and P fluxes to UK coastal waters under the NERC Environmental Virtual Observatory Programme (Emmet et al., 2014), using an aquatic fluxes model, that is the only model for the UK that has been calibrated and validated against nutrient flux measurements. The model predicts Total N) flux from land to water of ~712 kt N yr^{-1}, including all N fractions, and shows for 2010 that there was an N loading to groundwater of 291 kt N (~41%, made up of 288 kt N diffuse emissions into groundwater plus 2.7 kt N from septic tanks in groundwater), ~267 kt N (~37%) from diffuse sources in agriculture to surface waters, ~6 kt

N (~1%) from direct Nr deposition, ~135 kt N from public sewage (~19%) and 22 kt N (~3%) from industrial discharge (the model also estimates total P). Of this total ~712 kt N N yr^{-1} flux into watersheds, ~605 kt N yr^{-1} goes to coastal and marine ecosystems after denitrification. These figures are close to those estimated for the flux to surface waters in Figure 1.4.10.

The release of N_2 via industrial combustion and terrestrial denitrification (see Davidson et al., 2000 for microbial mechanisms) makes up a substantial amount of N loss. Although this represents a large source of nitrogen inefficiency, N_2 gas is inert and ultimately harmless to the environment and human health. However, denitrification to N_2 is nevertheless a significant loss of Nr resources that contributes to lower nitrogen use efficiency and the total amount of Nr that is wasted. Ultimately, if this N_2 loss can be reduced, it means that less fresh Nr input is needed to sustain human needs, increasing economy-wide NUE while reducing Nr pollution at the same time. For this reason, it is desirable to reduce all losses of N, including denitrification to N_2.

Table 1.4.3. Summary of annual N fluxes for the UK, based on data extracted from Worrall et al. (2016), Fan et al. (2020) and ENA (Sutton et al, 2011) with estimates of anthropogenic emissions of NOx, NH_3 and N_2O for the year 2018 as reported by the UK NAEI (https://naei.beis.gov.uk/). The values presented from Worral et al. (2016) represent the range of estimates from 2012 to 2020 and come with an estimated 80% range of uncertainty.

Source	Worral et al. (kt N)	Fan et al. (kt N)	ENA (kt N)	UK NAEI (kt N)
Estimate year	2012-2020	2015	1995-2008	2018
Input				
Atmospheric deposition	393-421	306	216	
Biological nitrogen fixation	402-413	505	108	
Food & feed transfers (Imports)	297-428	295	312	
Inorganic fertilizers	783-978	1650	1076	
Output				
Atmospheric Emissions (Nr)*	659-872	845		
N_2O (Anthropogenic)			21	44.4
NH_3 (Anthropogenic)			257	228
NO_x (Anthropogenic)			226	251
Industrial emissions of N_2	294-348	261		
Terrestrial denitrification to N_2*	205-209	173		
Fluvial losses at source*	1709-2220	1823		
Total N flux from terrestrial to surface water^{+}			714 (712^{+})	
Groundwater*	0-15	15		
Direct waste losses (Including Sewage)	49-55	58		

*Natural and anthropogenic processes including sediment and soil export; $^{+}$Derived using methodology shown in Greene et al. (2015)

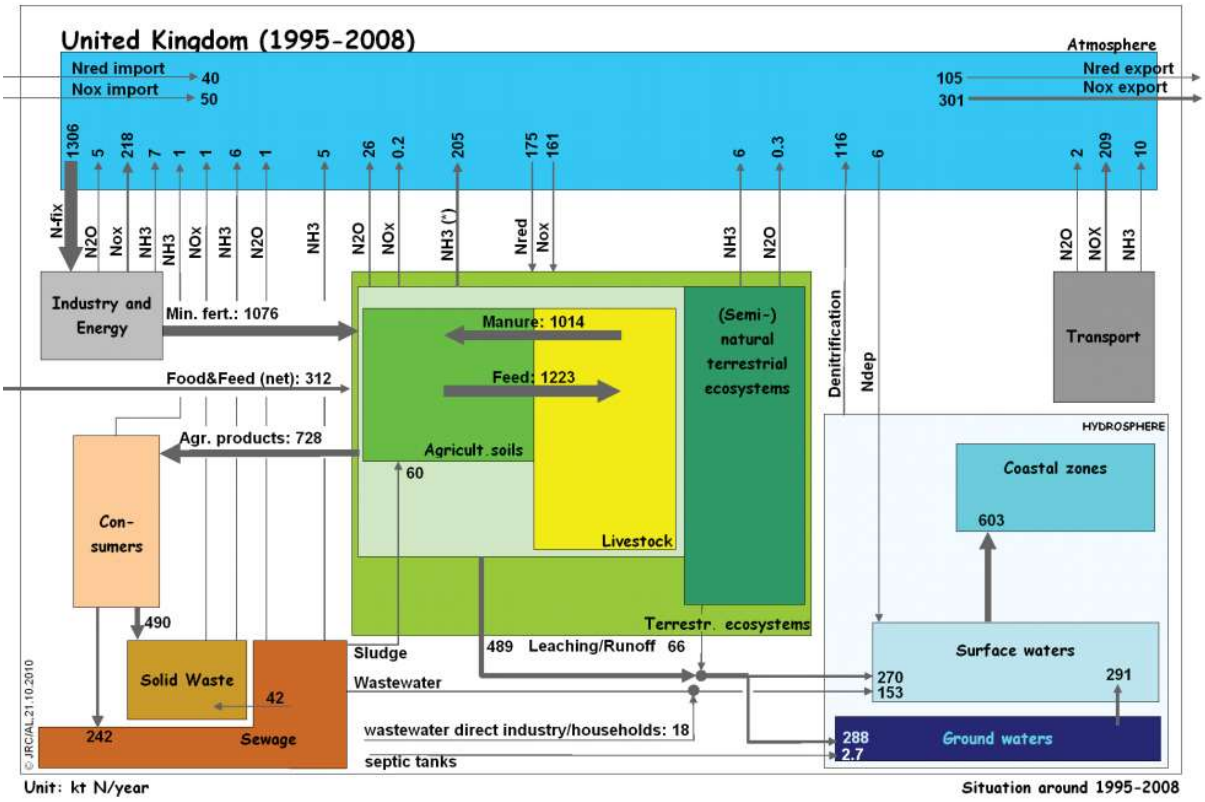

Figure 1.4.10 The integrated nitrogen budget of the United Kingdom builds on the UK TAPAS modelling for agriculture (Defra), providing data on mineral fertilizer application rates and manure management and on the national food and feed balance modelling, national N flux modelling to freshwaters and the coastal zone, the UK national emissions inventory and atmospheric transport and deposition modelling and a range of literature sources for UK waters. The data sets are not, however, co-incident in time, and the present budget represents the period from 1995–2005 (extracted from Figure 16.16 in ENA, Sutton et al., 2011).

In Stevens et al. (2014) the per capita N footprint (production and consumption) in the UK is estimated at 27.1 kg N per capita per year with food production constituting the largest proportion of the footprint (18.0 kg N per capita per year). In Stevens et al. (2014), a N footprint calculated for 1971 (26.0 kg N per capita per year) demonstrates that per capita N footprints have increased slightly. In Galloway et al. (2014) a comparison is made with several nations for which the information was available to carry out a more detailed N footprint (Table 1.4.4). Here, it is estimated that developed countries (such as the UK) consume (not just food) about the same amount of N (20–30 kg N capita^{-1} yr^{-1}), which is more than the example of a less-developed country (Tanzania; 15 kg N capita^{-1} yr^{-1}). The amount of N consumed in food varies between countries, with significantly higher quantities consumed in Portugal, the USA, the UK and Japan (3-6 kg N capita^{-1} yr^{-1}), while significantly less is consumed in Germany, Australia and Tanzania (1-2 N capita^{-1} yr^{-1}). Although these differences may be partly attributable to uncertainty in the studies carried out, there is a clear trend in that some nations consume significantly more N than others, predominantly as a result of a high protein diet (i.e., fish, meat and dairy).

One large difference between the developed nations was that some have substantially diminished the discharge of that N to the environment through advanced wastewater treatment that converts Nr to N_2 (e.g. The Netherlands, Germany, Austria). In the UK, it is estimated that only 2% of Nr in sewage waste is treated to denitrify it. Given that the level of sewage treatment is out of the consumers' direct control, the only way they have to decrease the food consumption portion of their N footprint is to decrease their consumption of protein to recommended levels. At the level of society, however, investment in efficient sewage systems is a realistic option to decrease losses of Nr to the environment, either by increasing denitrification rates, or by increasing the quality of sewage sludge produced and using this re-captured Nr for agricultural production.

Table 1.4.4. Annual per-capita nitrogen footprints for eight countries (kg N capita^{-1} yr^{-1}) calculated using the N-Calculator. 'Food consumption' refers to the nitrogen actually consumed and subsequently excreted, whereas 'food consumption, released' refers to the nitrogen released to the environment after sewage treatment (extracted from Table 1 in Galloway et al., 2014).

	UK	**US**	**NL**	**Germany**	**Japan**	**Austria**	**Portugal**	**Tanzania**
Food consumption	4.9	5	1.1	1.6	3.4	1.1	6	2
Food production	18	22	20	18	26	16	18	12
Housing	2	3	0.8	1.6	0.8	0.8	0.7	0.2
Transport	1.1	6	1.1	1.8	0.7	1.6	3.5	0.8
Goods and services	1.1	2.5	0.5	0.7	1	0.6	0.5	0.2
Total	**27**	**39**	**23**	**24**	**32**	**20**	**29**	**15**
Sewage, % denitrification	2%	5%	78%	67%	33%	79%	0%	0%
Food consumption, released	5	5.3	5	4.9	5.1	5.2	6	2

Another footprint methodology focusing on consumption-based allocation of N fertilizer applied to cropland only, as compared to N foot-printing where N losses along the supply chain are assessed (e.g. Stevens et al., 2014), has been conducted by Jennings et al. (2021). In this method, the Planetary Boundary for N of 62 Tg N y^{-1} (Steffen et al., 2015) was divided by world population to arrive at a per capita PB of 7.9 kg N y^{-1} (after O'Neill et al., 2018). National N footprint data were then taken from data published by O'Neill et al (2018), which was based on figures obtained from the Eora MRIO database (Lenzen et al., 2012, 2013), to represent the consumption-based allocation of N fertilizer applied to cropland. The underlying N fertilizer data was then compiled (e.g. Potter et al., 2011) and the N data scaled to match the current global anthropogenic N fixation (150 Tg N y^{-1}) as reported by Steffen et al. (2015). This analysis yields a UK per capita footprint of 72.9 kg N yr^{-1} (See: https://goodlife.leeds.ac.uk/countries/ and underlying data/analysis). This was compared by Jennings et al. (2021) to a global per capita PB of 7.9 to give a required reduction in the UK per capita footprint of 89%. As national N footprint data were obtained from the Eora MRIO database, and represent the consumption-based allocation of N fertilizer applied to cropland, this approach assigns responsibility for embodied resource use to final consumers, and therefore includes the influence of international trade on nutrient use. However, improved traceability of imports to a producer level is required to more fully understand nutrient use overseas associated with the production of goods for UK consumption. Research is also needed to understand air and water pollution embodied in UK imports.

1.5 Nitrogen, agriculture and Net-Zero in the UK

Agricultural GHG emissions in the UK were 45.6 MtCO_2e in 2017 (Estimated by BEIS, Figure 1.5.1). This represents 10% of UK GHG emissions in 2017 compared to 7% in 1990 (this is actually a lower estimate than the 54.6 $MtCO_2e$ estimated for 2018 in the sixth carbon budget (CCC, 2020a)). This increase from 7 to 10% reflects both the slow rate of progress in reducing the sector's emissions, and the faster pace of decarbonisation elsewhere in the economy (CCC, 2020a). Agricultural emissions have declined by 16% since 1990. This is mainly due to successive reform of the Common Agricultural Policy (CAP) in the 1990s and early 2000s, which reduced livestock numbers, coupled with changes in farming practices due to EU environmental legislation to address non-GHG pollutants (e.g. Nitrates Directives). There has been little change in emissions since 2008. However, the 6th Carbon Budget follows the proposed adoption of the new Global Warming Potential (GWP) values in 2024, in line with the IPCC Fifth Assessment Report (AR5) which includes climate-carbon feedbacks. There are two methodologies, and it is not yet clear which will be used, but both are different from the values used in the current emissions inventory and will lead to an increase in the estimate of UK emissions. A 'high' estimate of the GWPs include climate-carbon feedbacks and the size of the existing inventory would increase by around 19 $MtCO_{2e}$ while the 1990 baseline would increase by nearly 47 $MtCO_2e$. This is almost entirely due to a 36% increase in the estimated global warming impact of methane (CH_4) emissions, and is the basis upon which targets in the 6th Carbon Budget report are recommended (CCC, 2020a). The 'low' estimate of GWPs does not include climate-carbon feedbacks, and leads to a smaller increase in the size of the UK emissions inventory (around 5 $MtCO_2e$ while the 1990 baseline would increase by 10 $MtCO_2e$). Under this 'low' estimate CH_4 emissions have a 12% higher warming impact than the current estimate, while the warming impact of N_2O emissions is 11% lower. The two changes overlap because peatlands are a source of both CH_4 and N_2O emissions. The range for the total combined impact of the peatland and GWP changes is around an additional 27-70 $MtCO_2e$ in 1990 and 23-42 $MtCO_2e$ in 2019 compared to the current inventory (CCC, 2020a).

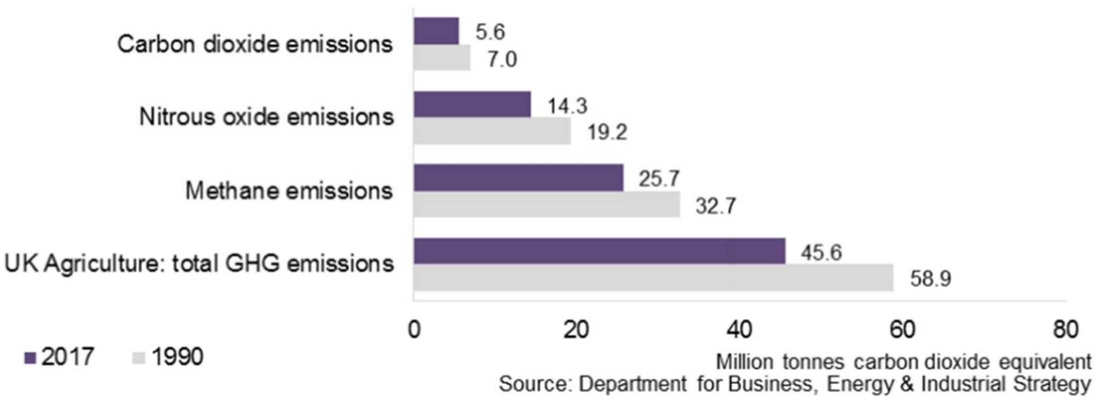

Figure 1.5.1 A comparison of GHG emissions from agriculture in the UK, 1990 and 2017 (Defra, 2019b).

Comparatively, the UK is relatively high emitting in terms of gross agricultural production (Figure 1.5.2). Comparisons of domestic agricultural GHG emissions across countries are difficult, not only because of data availability but also due to the differing types of agriculture undertaken in each country. Malta, Italy and Greece have some of the lowest levels of emissions per unit of gross agricultural production. This reflects the production of high value crops with low emissions (for example, olives and grapes) in these countries. Countries such as New Zealand and Ireland have some of the highest levels of emissions per unit of gross agricultural production reflecting the dominance of livestock farming in those countries. The diverse farming systems found in the UK leads to a lower level of emissions per unit of gross agricultural production. However, the preponderance of grassland, the largest population of sheep in Europe and a large population of suckler cows (which produce methane and are produced largely at very low or negative profit margins even though they may be comparatively efficient in production terms) place the UK amongst the upper half of UNFCCC countries when considering domestic emissions in this way (Figure 1.5.2).

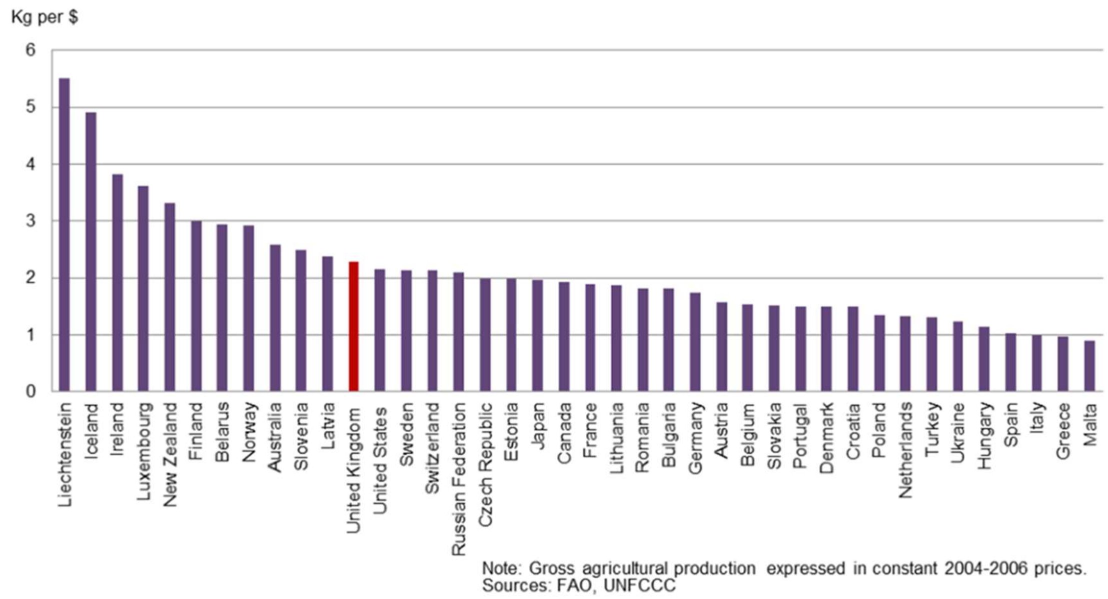

Figure 1.5.2. Agricultural emissions in CO_2e per unit of gross agricultural production (UNFCCC, 2016).

At the core of the 6th Carbon Budget (see: Climate Change Committee (CCC), 2020b) are multiple scenarios exploring the actions required in each abatement category and every year in order to reduce UK emissions to Net Zero by 2050 at the latest (Table 1.5.1). The detailed scenarios explore uncertainties, particularly over how far people will change their behaviours, how quickly technology will develop and the balance between options where credible alternatives exist (Figure 1.5.3). All the scenarios are ambitious while bounded by realistic assumptions over the speed at which low-carbon technologies can be developed and rolled out, allowing time for supply chains, markets and infrastructure to scale up. They are self-consistent and recognise other priorities – for example, the energy analysis maintains security of supply, the housing analysis considers the need for flood protection and to avoid overheating , the land analysis supports the natural environment. The 'Balanced Net Zero Pathway' is designed to drive progress through the 2020s, while creating options in a way that seeks to keep the exploratory scenarios open. The CCC also constructed a further exploratory scenario ('Tailwinds') that assumes considerable success on both innovation and societal /behavioural change and goes beyond the Balanced Pathway to achieve Net Zero before 2050.

Headwinds scenario - assume that policies only manage to bring forward societal/behavioural change and innovation at the lesser end of the scale, similar to levels assumed in the 2019 Further Ambition scenario. People change their behaviour and new technologies develop, but do not see widespread behavioural shifts or innovations that significantly reduce the cost of green technologies ahead of our current projections. This scenario is more reliant on the use of large hydrogen and carbon capture and storage (CCS) infrastructure to achieve Net Zero.

Widespread Innovation scenario - assumes greater success in reducing costs of low-carbon technologies. This allows more widespread electrification, a more resource- and energy-efficient economy, and more cost-effective technologies to remove CO_2 from the atmosphere. Assumed societal/behavioural changes are similar to Headwinds.

Widespread Engagement scenario - assumes higher levels of societal and behavioural changes. People and businesses are willing to make more changes to their behaviour. This reduces demand for the most high-carbon activities and increases the uptake of some climate mitigation measures. Assumptions on cost reductions are similar to Headwinds.

Based on the insights of these scenarios, the CCC has developed a Balanced Pathway as the basis for their recommended Sixth Carbon Budget for consideration by the UK Government's Department for Business, Energy & Industrial Strategy (BEIS) for inclusion in the UK's Nationally Determined Contribution (NDC). The Balanced Pathway makes moderate assumptions on behavioral change and innovation and takes actions in the coming decade to develop multiple options for later roll-out (e.g. use of hydrogen and/or electrification for heavy goods vehicles and buildings). While it is not a prescriptive path that must be followed exactly, it provides a good indication of what should be done over the coming years to inform the UK Government's strategy.

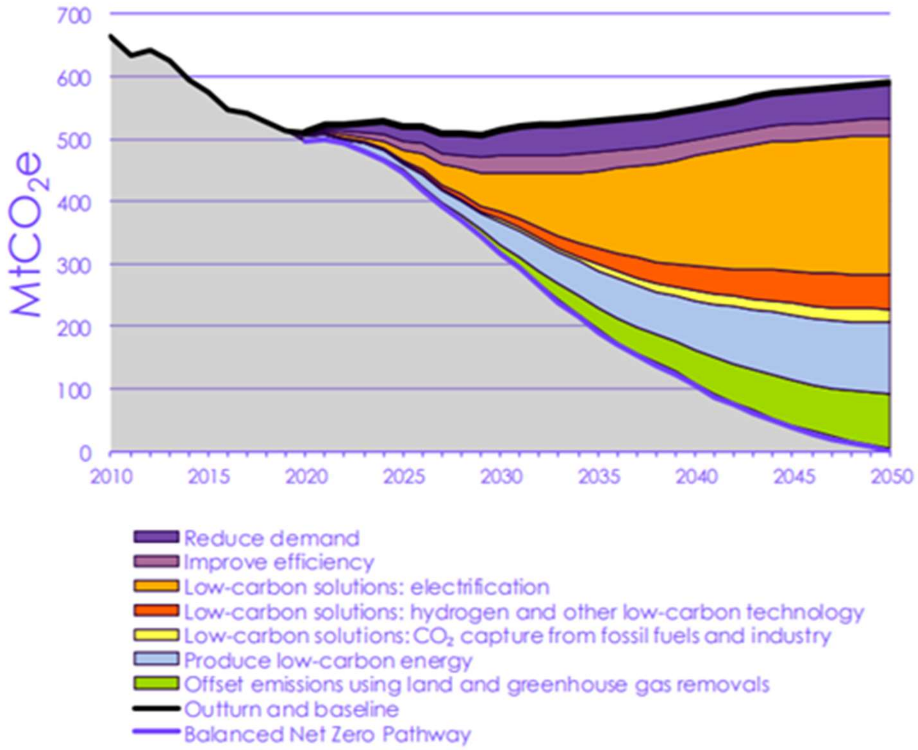

Figure 1.5.3. Types of abatement in the balanced net-zero pathway (CCC, 2020b)

Table 1.5.1. Key metrics for actions in the Balanced Pathway to meet the Sixth Carbon Budget.

		2019	2025	2030	2035	2050	Trend
UK greenhouse gas emissions	UK greenhouse gas emissions ($MtCO_2e$)	522	445	316	191	0	
	UK greenhouse gas emissions per person (tCO_2e/capita)	7.8	6.5	4.5	2.7	0	
Demand reduction	Weekly meat consumption (g) (includes fresh and processed meat)	960	880	770	730	630	
	Weekly dairy consumption (g)	2,020	1,840	1,620	1,620	1,620	
	Plane-km per person	11,700	11,000	11,000	11,400	13,700	
	Car-km per driver	12,900	12,600	12,400	12,200	11,700	
	Remaining waste per person, after prevention & recycling (kg)	490	400	310	280	300	
Efficiency	Carbon-intensity of a new HGV (gCO_2/km)	680	580	420	20	0	
	Increase in longevity of electronics	0%	30%	80%	120%	120%	
Electrification, hydrogen and carbon capture and storage	Carbon intensity of UK electricity (gCO_2e/kWhe)	220	125	45	10	2	
	Offshore wind (GWe)	10	25	40	50	95	
	Share of BEVs in new car sales	2%	48%	97%	100%	100%	
	Heat pump installations (thousand per year)	26	415	1,070	1,430	1,480	
	Manufacturing energy use from electricity or hydrogen	27%	27%	37%	52%	76%	
	Low-carbon hydrogen (TWh)	<1	1	30	105	225	
	CCS in manufacturing ($MtCO_2$)	0	0.2	2	5	8	
	CCS in rest of the economy ($MtCO_2$)	0	0.1	20	48	96	
Land	UK woodland area	13%	14%	14%	15%	18%	
	Energy crops (kha)	10	23	115	266	720	
	Peat area restored	25%	36%	47%	58%	79%	
	Land-based carbon sinks ($MtCO_2$)	18	18	20	23	39	
Removals	Greenhouse gas removals ($MtCO_2$)	0	<1	5	23	58	

It is recognised that fully decarbonising the agricultural sector is not possible (on current understanding) due to the uncertainties associated with the inherent biological and chemical processes involved in crop and livestock production. The options that there are to reduce GHG emissions cover behaviour change, productivity improvements and the take-up of low-carbon farming practices (Figures 1.5.4). Particularly important in the scenarios are an accelerated shift in diets away from meat and dairy products. For the categories of agriculture and land use, land use change and forestry, the combined GHG emissions were 67 $MtCO_2e$ in 2018, which could fall to 40 $MtCO_2e$ by 2035 in the Balanced Net Zero Pathway (CCC, 2020b).

Annual savings total 25 $MtCO_2e$ when compared to emissions in the Business as Usual scenario in 2035. By 2050 residual emissions reach 16 $MtCO_2e$ under the Balanced Pathway (approx. 27% of current emissions) but fall to Net Zero by 2047 in the Wider Innovation and Tailwinds scenarios.

Delivering this transition requires a transformation in the use of land. Around 9% of agricultural land will be needed for actions to reduce emissions and sequester carbon by 2035, with 21% needed by 2050 (this rises to 11% and 23% when including land for settlement growth). Under the balanced pathway route, by 2035 the CCC scenarios involve planting of 440,000 hectares of mixed woodland to remove CO_2 from the atmosphere as they grow, with a further 260,000 hectares of agricultural land shifting to bioenergy production (including short rotation forestry). This would see UK woodland cover growing from 13% now to 15% by 2035. Peatlands must be restored widely and managed sustainably. Low-carbon farming practices must be adopted widely, while raising farm productivity. Alongside the nature-based carbon removals, by 2035 the UK should be using bioenergy (largely grown in the UK) with carbon capture and (CCS) to deliver engineered removals of CO_2 at scale (see Figure 1.5.4).

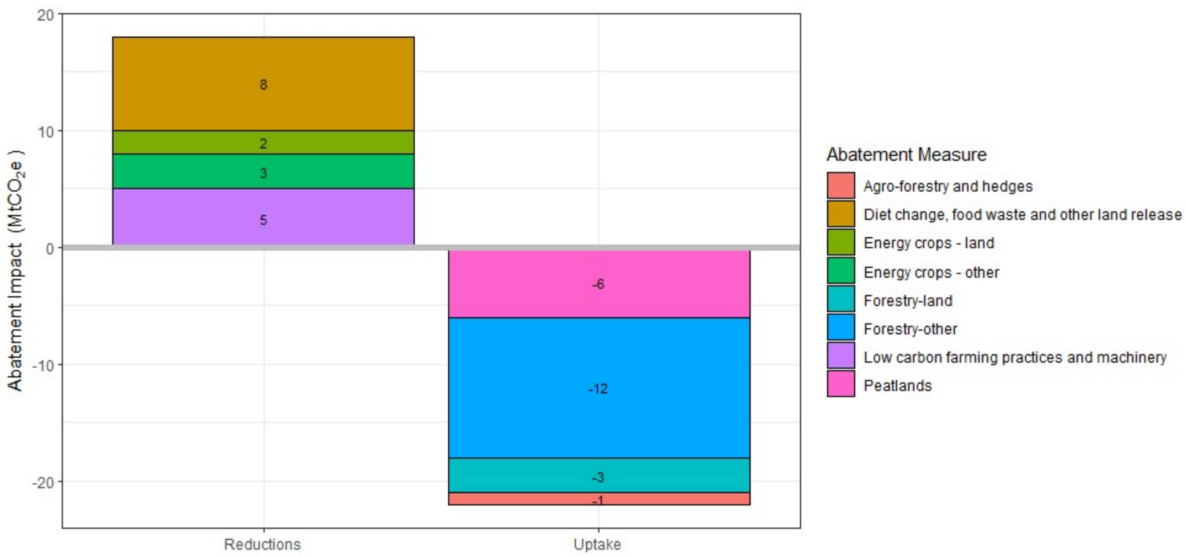

Figure 1.5.4 GHG savings from measures to reduce agriculture and land use emissions, 2035 (MtCO2e). Source: BEIS Provisional UK greenhouse gas emissions national statistics 2019; Centre for Ecology and Hydrology (2020); CCC analysis. Data are split between measures where reductions in all GHGs are considered, and where uptake of CO_2 is considered.

Actions in the balanced pathway include:

- The take-up of measures associated with changing farming practices (e.g. planting cover crops, livestock health measures and feeding cattle a high starch diet), and take-up of more innovative options (e.g. 3NOP additives, GM cattle, and breeding).
- Improvements in agricultural machinery.
- Releasing agricultural land by moving diets away from the most carbon-intensive foods delivers the highest emissions savings. (i.e. a shift away from meat and dairy and a greater willingness to act on food waste).
- Developing technology to bring lab-grown meat to the market.

- Crop breeding (e.g. development of new cultivars /traits) leads to higher yields (e.g. to 13 tonnes/hectare for wheat by 2050).
- Higher livestock stocking densities on permanent grassland releases around 0.8 million more hectares of land out of agricultural production.
- Scaling up afforestation rates to 30,000 hectares a year by 2025 in line with the UK Government's commitment, rising to 50,000 hectares annually by 2035.
- Full restoration of upland peat by 2045 (or stabilisation if degradation is too severe to restore) and re-wetting and sustainable management of 60% of lowland peat by 2050.
- Planting perennial energy crops (e.g. miscanthus and short rotation coppice) alongside short rotation forestry needs to accelerate quickly to at least 30,000 hectares a year by 2035, so that 700,000 hectares are planted by 2050.
- The use of Carbon Capture and Storage (CCS) technologies.
- Increasing on-farm diversification with the integration of trees on 10% of farmland and extending the length of hedgerows by 40% by 2050.

Delivering emissions reductions from agriculture should not be at the expense of increasing food imports that risk 'carbon leakage'. Therefore, both production and consumption of the highest carbon foods need to fall. The balanced pathway analysis assumes that the same proportion of UK food demand is met by UK food production in 2050 as is the case currently (taking account of the nutritional composition of different food after diet change). The carbon footprint of the UK's imported food would also fall, with the change in diets reflected in reduced imports of meat and dairy products. Policy will need to be carefully designed to ensure that risks of carbon leakage are avoided (see the CCC's accompanying Policy Report: Policies for the Sixth Carbon Budget & Net Zero (CCC, 2020e).

Deep emissions reduction in agriculture and land cannot be achieved without changes in the way land is used in the UK. Changes in consumer and farmer behaviour can release land from agriculture while maintaining a strong food production sector. CCC (2020b) considered five measures that could release land covering societal changes and improvements in agricultural productivity. The analysis implies that these five measures could reduce annual agricultural GHG emissions by 8 MtCO_2e by 2035, rising to just over 11 MtCO2e by 2050, **with diet change the most significant**: (i) Diet change; (ii) Food waste; (iii) Improving crop yields; (iv) Stocking rates for livestock; and (v) Moving horticulture indoors.

The Balanced Pathway involves a 20% shift away from meat and dairy products by 2030, with a further 15% reduction of meat products by 2050. These are substituted with plant-based options. This is within range of the Climate Assembly's recommendations for a 20-40% reduction in meat and dairy consumption by 2050. The pathway results in a reduction in livestock numbers and grassland area, delivering annual abatement of 7 MtCO_2e by 2035, rising to nearly 10 MtCO_2e by 2050. The CCC assumes food waste is halved across the supply chain by 2030 in line with the Waste and Resources Action Programme's (WRAP) UK Food Waste Reduction Roadmap. This would reduce UK emissions by almost 1 MtCO_{2e} in 2035.

The sixth carbon budget (CCC, 2020a) considers that improving crop yields without the need for additional inputs such as fertiliser and pesticides can be achieved through improved agronomic practices, technology and innovation while taking account of climate impacts. Crop breeding and selection could lead to higher yields through development of new cultivars/traits that allow the next generation of wheat and other crops to be more sustainably productive and resilient to disease in a warmer climate. It is assumed that policy will enable technological developments to be transferred to farmers (e.g. through information, skills and other incentives) to ensure the take-up of climate-resilient varieties that are most suitable to local conditions. The Sixth Carbon Budget states that it should be possible to sustainably increase crop yields in the future but also consider that if climate risks dominate then yields could fall. The scenarios assume average crop yields rise from 8.2 tonnes/hectare for wheat (the average over the past four years) to between 11 and 13 tonnes/hectare by 2050 (and equivalent increases for other crops) (CCC,2020a).

Shifting 10% of horticulture production indoors under a controlled environment reduces the carbon, nutrient, land and water footprint. Indoor horticulture, such as vertical farming where crops are grown in stacks in a controlled environment, can raise productivity while reducing nutrient, land and water footprints (CCC, 2020a). Horticultural products such as fruit, vegetables and salad crops are grown on 163,000 hectares, or 3% of cropland in the UK. Indoor horticulture in the UK is mainly for high value salad crops and is currently small scale. Some systems are based on hydroponic and vertical production systems using LEDs. These systems could be applied to 10–50% of current horticultural production. However, given the small area of land currently used for horticulture in the UK, moving production indoors has a limited impact on land area and carbon impacts. More significant emissions savings would come from moving horticultural production from lowland peat. Greater benefits could accrue from shifting arable crop production indoors. The controlled environment could allow for quicker and multiple harvests each year. Estimates suggest that combined with a ten-tier stacking system, yields could be 220 to 600 times higher than the current global average annual wheat yield of 3.2 tonnes/hectare. However, this production method is still at the experimental stage, with trials on-going at Rothamsted Research, while the costs of energy (e.g. LED lighting) would also have to reduce to make this a cost-effective option. Indoor wheat production is not included in the CCC (2020a) scenarios.

The CCC commissioned SRUC (e.g. Eory et al. 2020; Eory et al. 2015) to assess the abatement potential from measures to reduce emissions from soils (e.g. grass leys and cover crops), livestock (e.g. diets and breeding) and waste and manure management (e.g. anaerobic digestion). These reduce agricultural emissions by 4 $MtCO_2e$ in 2035. This takes account of the interaction with other actions, notably diet change, which reduces the abatement potential of these measures overall (See Section 2 for details).

Currently 18 TWh of fossil fuels are used in agricultural vehicles, buildings and machinery, resulting in emissions of 4.6 $MtCO_2e$. Options to decarbonise fossil fuel use are similar to those in surface transport, off-road machinery in industry and commercial buildings. These cover electrification, biofuels, hydrogen and hybrid vehicles. The Balanced Pathway assumes biofuels and electrification options are taken-up from the mid-2020s and hydrogen from 2030, reducing emissions to 2 $MtCO_2e$ in 2035.

1.5.1 Where does Nitrogen fit in?

The actions outlined in the above balanced pathway are focused largely on changing how land is used, what we consume and improving the efficiencies of current farm management practices. In terms of land use change, carbon capture and mechanical efficiencies, much of these actions are focused on the reduction of CO_2 emissions (or creating sinks). Where livestock and general efficiency savings are concerned, this will reduce emissions of all three of the main GHGs; CO_2, CH_4 and N_2O. In terms of magnitude of agricultural measures, 18 $MtCO_2e$ of savings are targeted by reducing all GHG emissions, while 22 $MtCO_2e$ is assumed as a result of increasing carbon sinks (CO_2 only).

The largest source of N_2O emissions in the UK that will need to be reduced is that of direct emissions from agricultural soils which account for 55% of the UK's N_2O emissions (i.e. the microbial emission of N_2O after N fertiliser/manure is applied (Davidson et al., 2000)). This is a topic of high uncertainty and limited options regarding mitigation efforts without negative economic consequences or a significant fall in production. Essentially, where fertiliser is applied to fields in artificial or organic forms, approximately 0.03 – 3% of the N applied will be converted to N_2O by microbes in soils and aquatic systems (when leaching/fluvial losses occur) (IPCC, 2014). In terms of mitigating microbial emissions as a result of direct fertiliser use, options are limited beyond efficiency savings (Rees et al., 2013).

In regards to N emissions pollution that is not N_2O, there is a fertilization impact that will increase photosynthesis, and thus increase CO_2 uptake in plants that receive N deposition. Although the impact of N pollution on human health and ecosystems is sizable and important to mitigate, air pollution by N also generates social benefits for climate by present cooling effects of N containing aerosol and C-sequestration driven by N deposition, amounting to an estimated net benefit in the EU of about €5 billion/yr (Van Grinsven et al., 2013)(Figure 1.5.5).

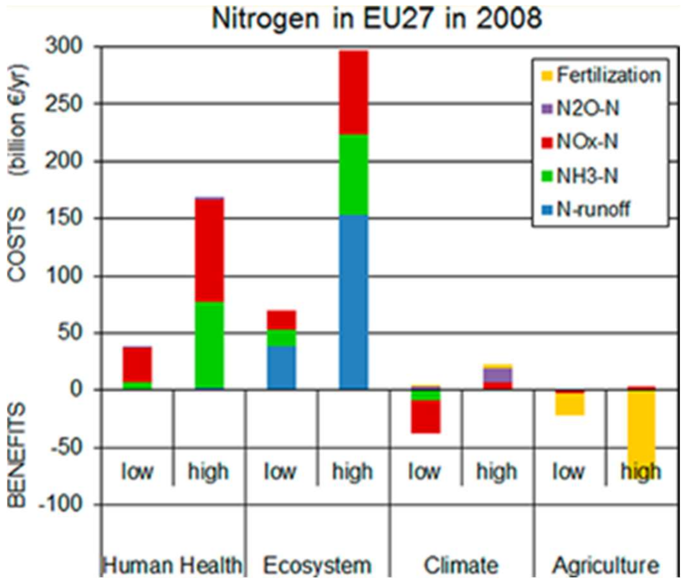

Figure 1.5.5 Costs and benefits of N pollution for the EU27 in 2008, as reported in Van Grinsven et al. (2013).

Agriculture currently accounts for 10% of GHG emissions in the UK, and N_2O accounts for approximately 32% of agricultural emissions. Thus, agricultural emissions of N_2O account for approximately 3% of the UK total GHG budget (in terms of CO_2e). As reductions in GHGs are made in the UK in the coming years (potentially following the balanced pathway approach), these emissions will likely decrease to some extent; however, due to specific difficulties in reducing N_2O emissions from soils, they will likely contribute to a larger proportion of the remaining emissions in the UK. However, this is also dependent on how GHGs are classified, and the adoption of the new Global Warming Potential (GWP) values in 2024, in line with IPCC guidance, will increase methane emissions by 36%, thus the impact of N_2O on overall emissions will lessen.

As the approaches set out to achieve net-zero in the UK allow for some residual emissions from agriculture (i.e. agriculture is offset in the balanced pathways approach), the threat of future N_2O emissions to achieving net-zero is largely tied to the success of actions aimed at creating carbon sinks, as well as reductions in N_2O itself. As explained above, reductions in N pollution may also result in the reduction of photosynthesis and atmospheric particulates which currently act beneficially to prevent global warming. Therefore, any efforts to reduce N pollution in the UK (i.e., NO_x and NH_3) will also likely offset some of the savings made by a reduction in N_2O emissions. Based on the relatively small contribution to total UK emissions as a result of N_2O emissions from agriculture, and the numerous different pathways outlined to achieve net zero (and associated uncertainties), it is unlikely that N_2O emissions from agriculture are at risk of preventing net-zero being achieved. However, this is entirely dependent on the ability to sequester carbon elsewhere to offset those from agricultural sources. Should the UK not manage to become a net-sink of CO_2e in all sectors, then emissions of N_2O will remain, and will prevent the country from reaching its net-zero goals.

Although the impact of N emissions plays only a small role in the overall GHG budget in the UK, there are still dangers associated with N_2O emissions that will have lasting effects on climate change. As a long-lasting gas species (with a lifetime longer than 100 years), N_2O will have a warming impact for more than a century after its release. The planting of forests and regeneration of carbon sinks in the UK (or globally) will not result in falling N_2O concentrations in the atmosphere. As plans for net-zero are focused on CO_2e, and use carbon sinks to offset N_2O emissions, concentrations of N_2O will continue to rise even if the most intense reductions in GHG emissions are achieved. If GWP values are revised (as suggested by the CCC) from 100 to 25-year lifetime impact, then CH_4 will take on a greater role in GHG budgets, and N_2O will become further marginalised. Here there is a real danger that the long-lived and difficult to mitigate emissions of N_2O will remain a low priority in future Net-Zero targets and will continue unabated.

The most significant sink of N_2O in the atmosphere is via the reaction with ozone in the stratosphere, which converts N_2O into N_2. As a result, the increasing concentrations of N_2O in the atmosphere (rising at approximately 1 ppb yr^{-1}, see https://www.n2olevels.org/) are having a larger impact on ozone depletion. As a result of the banning of Chlorofluorocarbons (CFCs) in the 1980s, N_2O has become the single largest contributing species to ozone depletion (Figure 1.5.6). So long as agricultural activities result in the release of N_2O emissions beyond that which can be naturally destroyed, concentrations will continue to rise globally and the threat to the ozone layer will increase.

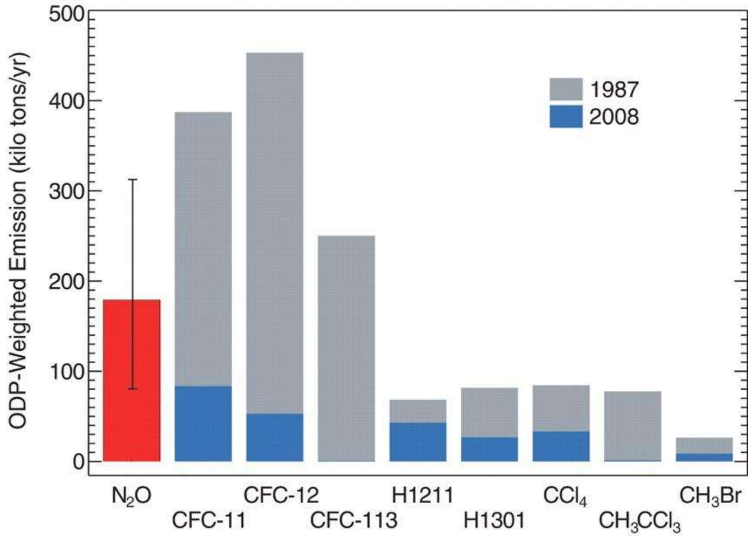

Figure 1.5.6 Comparison of annual N_2O ODP-weighted emissions from the 1990s [IPCC, 2007] with emissions of other ozone-depleting substances in 1987, when the emissions of chlorine- and bromine-containing ODSs were near their highest amount. (Figure 1 in Ravishankara et al., 2009).

1.6 The Costs and Impacts of Nitrogen in the UK

The costs of N use can be classified in multiple ways. The simplest of these is the actual cost of N application, and the efficiency of the N that is converted to a useful product (i.e., food), compared with what is wasted. The NUE of N application varies significantly, depending upon crop type, environmental conditions and management practices. Where N fertilisers are applied intensively and regularly, N losses can be expected to be higher than where N is lacking in supply. An NUE above 100% would suggest soil N depletion, which would be unsustainable. In terms of other EU nations with a high degree of intensive agriculture, the UK is comparable in terms of NUE, converting approximately 55% of applied N to a form of crop (arable, grass, etc.) (Figure 1.6.1).

NUE at the farm level is dependent upon a number of factors but estimates of overall N losses in the food chain are greater than 80% if food waste, fertiliser production and transportation are all considered (Sutton, 2013). When crops are grown for direct human consumption, NUE can be as high as 40 to 80%; however, losses in the livestock sector are considerably higher. Livestock NUE indicates that only 6% of the N consumed by livestock globally reaches human food (prior to food waste), as compared with 19% in the European estimates (Sutton, 2011). This is due to N losses in the rearing of livestock (metabolism, animal waste, animal feed waste, human food waste, etc.)

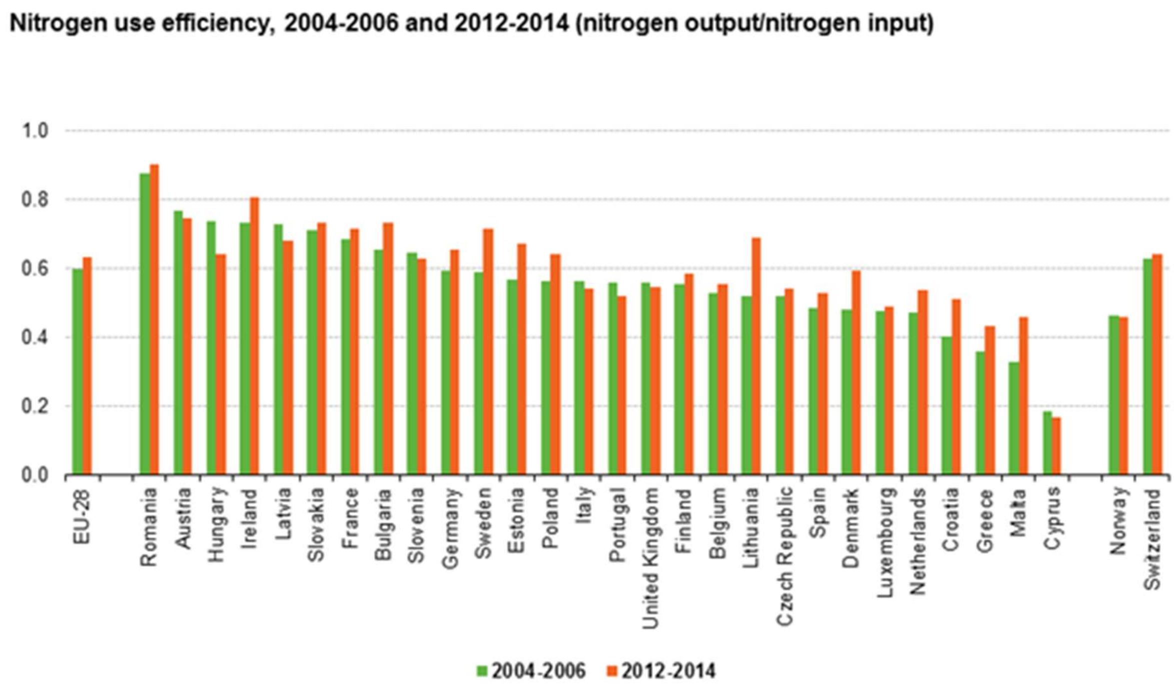

Figure 1.6.1 Nitrogen use efficiency in the EU, 2004-2006 and 2012-2014. Source: Eurostat (2021)

In the UK, artificial fertilisers cost approximately £0.70 – 0.99 kg N (AHDB, 2021), and the average N use (including manures) is approximately 137 and 54 kg N/ha for tillage crops and grasslands, respectively (BSFP, 2019). Nitrogen application therefore incurs a cost (using average price of £0.85) of approximately £116 ha^{-1} and £46 ha^{-1} annually for tillage crops and grassland soils (excluding the use of manures) respectively. In the UK, animal manures make up just over half of the applied N, thus annual fertiliser costs are likely to be significantly smaller where a supply of manure is readily available. To put this in perspective, a typical wheat harvest is expected to make in the region of £450-750 ha^{-1} and beef production is expected to make £600-1000 ha^{-1} in profit margins for a farmer in the UK (depending on weather, costs, global economics etc.) (AHDB.org.uk).

Although not trivial, the relatively low cost of N application and wide scale availability of animal manures has historically resulted in inefficient use of available N resources in the UK. Investments in farming have typically been targeted elsewhere, such as land improvement (i.e., drainage, pH, soil quality, etc.) and farm machinery (i.e., reduced labour and energy costs) to increase profit margins and overall efficiencies. Based on the above estimates, N waste is costing UK farmers approximately £21 - £52 ha^{-1} yr^{-1} in fertiliser costs annually for tillage crops and grassland soils respectively. Assuming an application of 1038 kt N in artificial fertilisers, a cost of approximately £0.85 per kg N and an NUE of 55%, it can be estimated that N losses from artificial fertiliser alone costs UK farmers approximately £397 million every year (artificial N only). A gross value of £9.4 billion was generated by farming in the UK in 2020, of which farmers were able to keep £4.1 billion as profit (Defra, 2021) (Figure 1.6.2). In terms of total profits, the cost of N waste at a UK scale is similar in magnitude to the year-to-year variability experienced by farmers due to weather conditions (Figure 1.6.2).

Based on the minimum and maximum data (and references) presented in Table 1.4.3 it is estimated that 1802 – 3695 kt N is lost from the UK annually. Based on a fertiliser cost (farmer prices) of approximately £0.70 – 0.99 kg N (AHDB, 2021), it can be estimated that between

£1.3 to 3.7 billion worth of N is lost to the environment in the UK annually. As there is uncertainty over the magnitude of losses to aquatic systems (see section 1.4.3), here we take an estimate of total nitrogen waste (from all sources) at a UK scale as approximately £2.5 billion per year as an illustration. Then if combustion NO_x emissions are subtracted and only N wasted in the agri-food chain is considered, this becomes £2.3 billion a year, which represents 56% of the £4.1 billion profit.

Although 100% NUE is not a realistic target due to the inevitable losses of N via microbial transformations and other pathways, improving NUE via a reduction in N use, or increasing yield could have a significant impact on profitability of farming in the UK (approximately 10% increase if 100% NUE were theoretically achieved). Experience from climate mitigation technologies highlights the need for investment to catalyse innovation in the most cost-effective nitrogen technologies, including mobilizing 'NitroFinance' in order to accelerate the development of the most profitable systems.

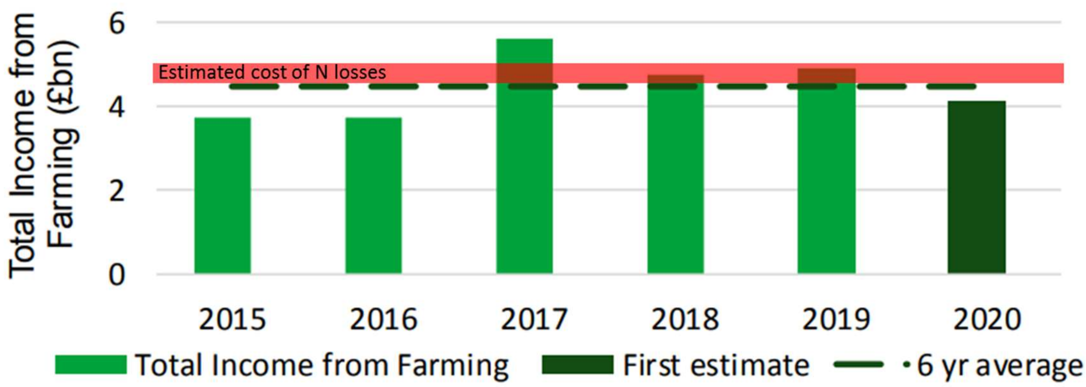

Figure 1.6.2. Total Income from Farming (TIFF) for the United Kingdom: 2015 to 2020 at Current Prices (not adjusted for inflation). Scale of profitability (£397,000) loss due to N losses included as red bar. (Defra, 2021).

Also worth considering are the societal costs of the pollution generated as a result of N losses. This is not an easy task, though estimates have been assessed in a number of studies. It is abundantly clear that production of food is the primary concern regarding N use; however, there is an important balance to be made in terms of the resulting impacts. In terms of N application, there is an offset between what is optimal for the farmer in terms of yield response and income, and what is optimal for society (economically) (example in Figure 1.6.3). There comes a point at which the non-linear yield response to N fertiliser slows, and the cost of the impacts of N pollution exceed that of the additional agricultural production (varies by fertiliser type and crop type). As these costs are uncoupled (i.e. farmers have little incentive to reduce N application below farm optimum), it is unlikely that fertiliser use will decrease without further action, although this action can come in several forms. Most desirable, would be to decrease the rate at which N needs to be applied to reach farm optimal conditions (i.e. increase NUE). Bringing optimal levels of fertilisation in line (or below) societal costs would not have a negative impact on food production or farming profitability, thus would not face the same resistance as other methods.

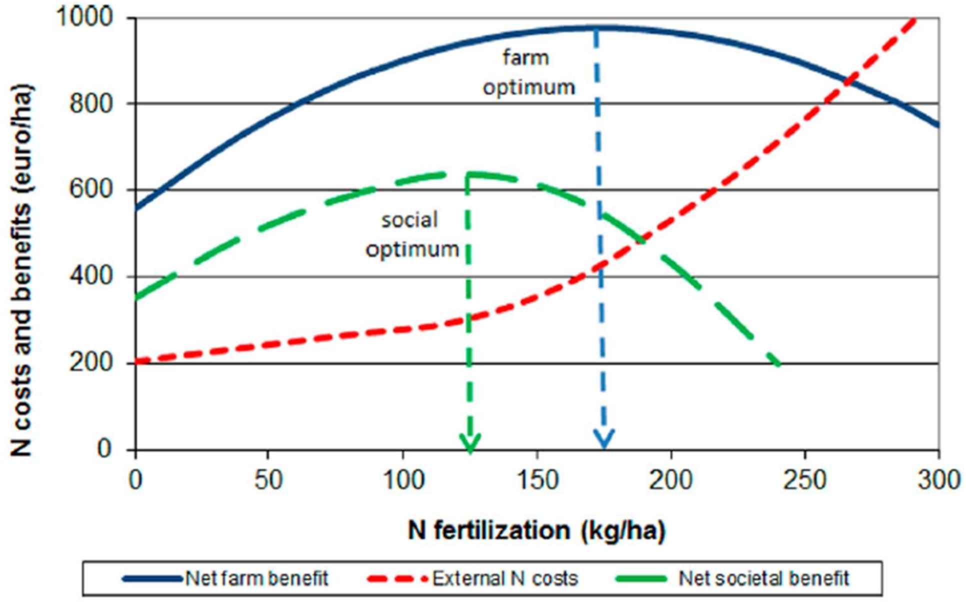

Figure 1.6.3. Benefits and costs of nitrogen fertilization (CAN) on winter wheat. (N response based on Henke et al. (2007) which is representative for German conditions) (Figure 5 in Van Grinsven et al. 2013).

Considering that the impact of pollution will vary geographically, the cost to human health will depend on population exposure and the impact on the environment will depend upon many factors, the complexities of these calculations are beyond the scope of this report. In Van Grinsven et al. (2013) the costs of N pollution are set out, per species and in terms of the multiple societal impacts that each has in the EU (Table 1.6.1). Combining these estimates with those set out in Section 1.4 (using the European Nitrogen Assessment to fill gaps, see Figure 1.4.10), provides an estimated cost of approximately £10.9 billion per year of societal costs due to N pollution in the UK (assuming an exchange rate of £0.86 per €)(Table 1.6.2). Of these costs, approximately 60% are attributed to the impact on human health, predominantly that of NO_x and NH_3 emissions. Defra (2019) in their Clean Air Strategy report that Public Health England has estimated that the health and social care costs of air pollution ($PM_{2.5}$ and NO_2) in England could reach £5.3 billion by 2035. This is a cumulative cost for diseases which have a strong association with air pollution: coronary heart disease; stroke; lung cancer; and childhood asthma. When diseases with weaker evidence of association are also added, including chronic obstructive pulmonary disease; diabetes, low birth weight, lung cancer, and dementia, the costs could reach £18.6 billion by 2035. When all diseases are included, air pollution is expected to cause 2.4 million new cases of disease in England between now and 2035. $PM_{2.5}$ alone could be responsible for around 350,000 cases of coronary heart disease and 44,000 cases of lung cancer in England over that time. Even small changes can make a big difference, just a 1µg/m^3 reduction in $PM_{2.5}$ concentrations this year could prevent 50,000 new cases of coronary heart disease and 9,000 new cases of asthma by 2035.

Table 1.6.1 Marginal costs between 1995 and 2005 of different Nr-Threats in the EU (Data from Van Grinsven et al. (2013), Table 1).

Effect	N Form	Loss to	Estimated Cost (€ kgN^{-1})
Human health (particulate matter, NO_2 and O_3)	NO_x	Air	10–30 (18)
Crop damage (ozone)	NO_x	Air	1–2
Ecosystems (eutrophication, biodiversity)	N_r (nitrate) N_r deposition	Surface water	5 to 20 (12)
Human health (particulate matter)	NH_3	Air	2–20 (12)
Climate (greenhouse gas balance)	N_2O	Air	4–17 (10)
Climate**	NO_x	Air	−9 to 2 (−3)
Climate**	NH_3	Air	−3 to 0 (−1)
Ecosystems (eutrophication, biodiversity)	NH_3 and NO_x	Air	2–10 (2)
Human health (drinking water) N_r (nitrate)	N_r (nitrate)	Groundwater	0–4 (1)
Human health (increased Ultraviolet radiation from ozone depletion)	N_2O	Air	1–3 (2)
Climate (N-fertilizer production)	N_2O, CO_2	Air	0.03–0.3

Table 1.6.2 Estimated societal costs of N pollution in the UK, based on values provided by Van Grinsven et al. (2013).

Effect	N Form	ktN	Estimated Cost £ Million	Min Cost £ Million	Max Cost £ Million
Human health (particulate matter, NO_2 and O_3)	NO_x	251	3885	2159	7530
Crop damage (ozone)	NO_x	251	324	216	502
Ecosystems (eutrophication, biodiversity)	N_r (nitrate) N_r deposition	342	3529	1471	6840
Human health (particulate matter)	NH_3	228	2353	392	4560
Climate (greenhouse gas balance)	N_2O	44.4	382	153	755
Climate**	NO_x	251	-648	-1943	502
Climate**	NH_3	228	-196	-588	0
Ecosystems (eutrophication, biodiversity)	NH_3 and NO_x	479	824	824	4790
Human health (drinking water) N_r (nitrate)	N_r (nitrate)	291	250	0	1164
Human health (increased Ultraviolet radiation from ozone depletion)	N_2O	44.4	76	38	133
Climate (N-fertilizer production)	N_2O, CO_2	1038	134	27	311
Total (£ Billion)			**10.9**	**2.7**	**27.1**

The Environmental Audit Committee (2018) reported that in 2015 Defra estimated that in England, businesses, the third sector and public sector jointly spent about £5 billion a year to protect the water environment (to prevent deterioration) and protect public health and wellbeing. This included:

- water industry operating costs to collect and treat sewage of approximately £3billion;
- industry and businesses investment of around £1 billion to mitigate their potential impact on the water environment and meet basic regulatory requirements;
- £450 million by agriculture to meet basic regulatory requirements and further reduce impacts on the water environment, including payments under the Common Agricultural Policy and voluntary industry initiatives;
- expenditure by government and the voluntary sectors to mitigate historic damage and provide water related benefits for people and wildlife.

The true cost of N pollution will depend largely on where it is experienced and what forms it takes. The societal costs of elevated particulate matter concentrations in the middle of oceans will have significantly smaller effect on human health than particulate matter in busy cities, though will likely impact marine life more when it deposits. Costing the impacts of N pollution is difficult and depends heavily on how we value human life and nature in terms of monetary worth. Some impacts of N pollution are highly localised (such as health impacts of NH_3 emissions), whereas some have more of a global impact (such GWP of N_2O emissions). In terms of N pollution from the UK, approximately 37.5% of reduced N (NH_3, NH_4, amines etc…) remains in the UK after emission, while only 16% of oxidised N (NOx, N_2O, NO_3, etc…) does. The majority of N pollution released in the UK ends up deposited in the Atlantic or North Sea, with a small percentage of reduced and oxidised N species reaching neighbouring states such as France, Germany and others. N pollution from other nations also contribute to pollution experienced in the UK, which is largely dependent on wind direction and environmental conditions (i.e. temperature and rainfall) (Table 1.6.3). There can also be a significant flux in the opposite direction, for example, it has been estimated that about 50% of the particulate NH_4 related PM in the UK may originate from gases emitted elsewhere in Europe (see Section 1.1.2).

Table 1.6.3. Fate of Oxidised and Reduced Nitrogen released from the UK for the year 2108, as modelled by the EMEP model (UKCEH). More information about calculations can be found in chapter 5 of EMEP Status Report 1/2018 and definitions of compounds in chapter 1.2 of latest EMEP Status Report (https://www.emep.int/publ/common_publications.html).

GB dep on:	**Reduced N (kilotonne N yr^{-1})**	**rel. share**	**Oxidised N (kilotonne N yr^{-1})**	**rel. share**
Great Britain	85.1	*37.5%*	38.8	*16.0%*
Atlantic	52.3	*23.1%*	72.3	*29.9%*
North Sea	48.9	*21.5%*	57.9	*23.9%*
France	5.9	*2.6%*	9.8	*4.1%*
Germany	5.5	*2.4%*	10.3	*4.3%*
Baltic Sea	3.6	*1.6%*	6.8	*2.8%*
Norway	3.2	*1.4%*	5.4	*2.2%*
Russia	3.1	*1.4%*	6.6	*2.7%*
Sweden	3.1	*1.3%*	5.4	*2.2%*
Ireland	2.6	*1.2%*	2.5	*1.0%*
Netherlands	2.0	*0.9%*	2.9	*1.2%*
Poland	1.5	*0.7%*	3.7	*1.5%*
Denmark	1.4	*0.6%*	2.3	*0.9%*
Belgium	1.1	*0.5%*	1.6	*0.7%*
Mediterraenan	1.1	*0.5%*	2.2	*0.9%*
Spain	1.0	*0.4%*	1.6	*0.6%*
Finland	0.9	*0.4%*	1.6	*0.7%*
Ukraine			1.2	*0.5%*
Belarus			1.0	*0.4%*

2. Identifying the key interventions

As set out in Section 1, there are many factors driving N over-use, poor NUE and high emissions of reactive N. UK farmers have a key role to play in actively implementing measures to reduce loss of N and N pollution, but they are part of a wider UK and global agri-food system of consumers, suppliers, retail, food and waste processors, government and scientists who all have a role in providing financial support, market signals, social norms, regulations that influence farmers' decisions about what to produce, how much to produce, and how to go about it (UNECE, 2021; principle 2). UK consumption also has an impact elsewhere in the world. In addition, significant quantities of N pollution occur from other sectors, resulting from fuel combustion and waste processing.

This section outlines key interventions – technical measures and approaches, or specific changes that need to be made – that will have a substantial impact in reducing N pollution and more generally waste of N. Section 3 then discusses the types of policy frameworks – regulation, financial incentive structures, education/outreach, decision support – which can drive the uptake of these interventions. Several different themes are considered separately (although they are clearly linked in many ways):

- Dietary change
- Reduction in food waste
- Imports of food and feed through global supply chains
- Farm-level and farming system measures
- Non-agriculture emissions
- Wastewater treatment measures

For each theme, key interventions for the UK are identified based on the literature, considering the potential impact, marginal abatement cost, co-benefits / trade-offs (e.g. with GHG emissions, water quality or biodiversity), current level of UK implementation, examples of implementation elsewhere (where relevant) and any barriers to adoption.

Whilst we have highlighted a "long list" of what we believe to be the most promising measures for circumstances in the UK, there is likely no "one-size-fits-all" solution and in any given case there may be practical considerations which mean another measure is more appropriate. Also, the marginal abatement cost may vary from place to place.

Therefore, this list should not be seen as exclusive or prescriptive and, to a certain extent, policies (see Section 3) should remain as neutral as possible with regard to which measures are favoured. A greater overall impact can likely be achieved by employing a variety of approaches in a context-specific way. For this reason, we have also grouped together similar types of measure to avoid being too specific. In addition, we will highlight any interventions (whether "key" or not) which would likely have serious trade-offs, to provide context.

Importantly, interventions considered here include both those which reduce N pollution *per se*, and those which reduce N loss including as N_2 through denitrification. Around 30% of N lost from agricultural soils in the EU28 (in 2014) was in the form of N_2, so this is a key flow. We try to make this distinction throughout, as terminology such as "nitrogen waste" is sometimes used in an ambiguous way in the literature and we wish to be more precise here.

2.1. Dietary change

Westhoek et al. (2015) provide a comprehensive review of N emissions associated with current food consumption and alternative diets at the EU level. Here, we bring out highlights from this report considering the N footprint of different diets, but also bring evidence from elsewhere to consider other diet scenarios, interactions with GHG mitigation, alternative protein sources, and policy options for influencing diet choice.

2.1.1 N (and GHG) footprint of different foods

Current average diets in the UK (and other developed countries) are associated with high emissions of reactive N. The reason for this is twofold: i) on average we eat more protein than we need, and ii) over half of the protein we consume comes from animal products, which in general have a higher N_r footprint than plant or fungal protein (Westhoek et al., 2015).

2.1.1.1 N_r Emissions footprints of different protein sources

Figure 2.1.1 illustrates the stark difference in Nr emissions footprint between animal and plant products respectively, and also differences among these.

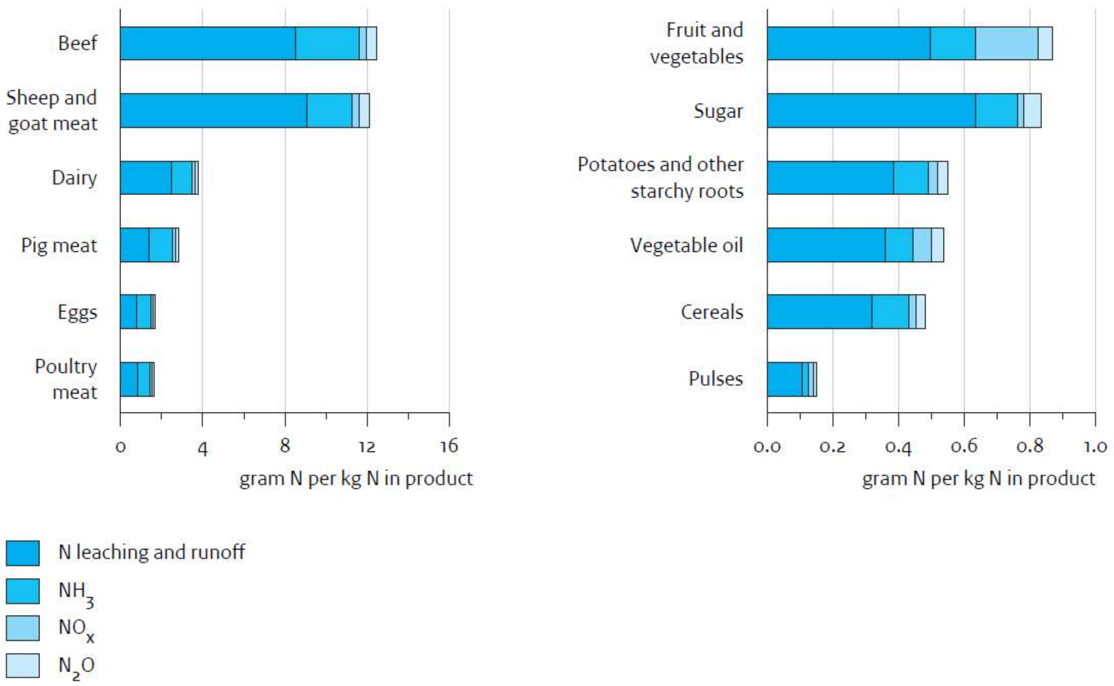

Figure 2.1.1 Nr emissions intensity of different N species, per gram of N in product, for animal products (left) and plant products (right). Source: Westhoek et al. (2015), based on the CAPRI model. Where sheep are housed outdoors all year round, much smaller NH_3 losses are expected. Note: the different scales on the x axis of each graph.

According to the Life Cycle Analysis (LCA) conducted, beef and sheep and goat meat has almost 100 times the associated N_r emissions per kg of N in the product than do pulses, and the best-performing animal product (poultry meat and eggs) still has over 2 times the footprint of the worst-performing plant product (fresh fruit and vegetables).

The basic reason for this difference is that when plant food is converted to animal protein, there is always a loss of energy and materials in the process – in other words, animals are “leaky”. Losses also occur between applying fertiliser to land and protein available in crops, but N use efficiency (NUE) of crops in terms of N outputs versus N inputs to soil is higher than that of animals in terms of animal protein out vs. feed protein in. NUE in the EU in 2014 was around 45% - 76% depending on the crop (just over 60% for cereals), and between 6% and 37% for animal products (Westhoek et al., 2015). For livestock fed with crop-based feeds, there are two stages of N loss – once when the crop is grown, and once when the feed is fed to the animal (although use of animal manure N to fertilise feed crops makes the overall system losses lower than would be expected). The NUE varies significantly across animal product types, varying largely with the feed conversion ratio (FCR). Estimates vary, but according to Westhoek et al. (2015), conversion of feed protein to human-edible protein is around 5-6 times more efficient in poultry meat and eggs compared with beef, lamb and goat meat, with pork and dairy products somewhere in between. This difference in feed conversion ratio and NUE relates to the more rapid growth and/or higher productivity of some livestock types than others, which means a greater proportion of feed calories and protein can be allocated to productive growth and milk/egg production, rather than body maintenance. There are also differences between production systems for ruminants, with (in general) higher NUE and lower Nr emissions per unit product for ruminants in low N input (e.g., extensive grazing) systems (Oenema, 2006; M. Sutton, pers. comm).

Within crops, pulses have an especially low Nr emissions footprint, largely because they fix a large proportion of their own N in root nodules filled with symbiotic bacteria, which is associated with lower Nr losses through all routes than applied fertiliser. For the same reason, Nr emissions associated with unfertilised grasslands which depend on biological fixation are much lower than for fertilised grasslands.

The same underlying differences in feed conversion ratio also give rise to similar patterns observed for GHG emissions intensity of different protein sources (Figure 2.1.2).

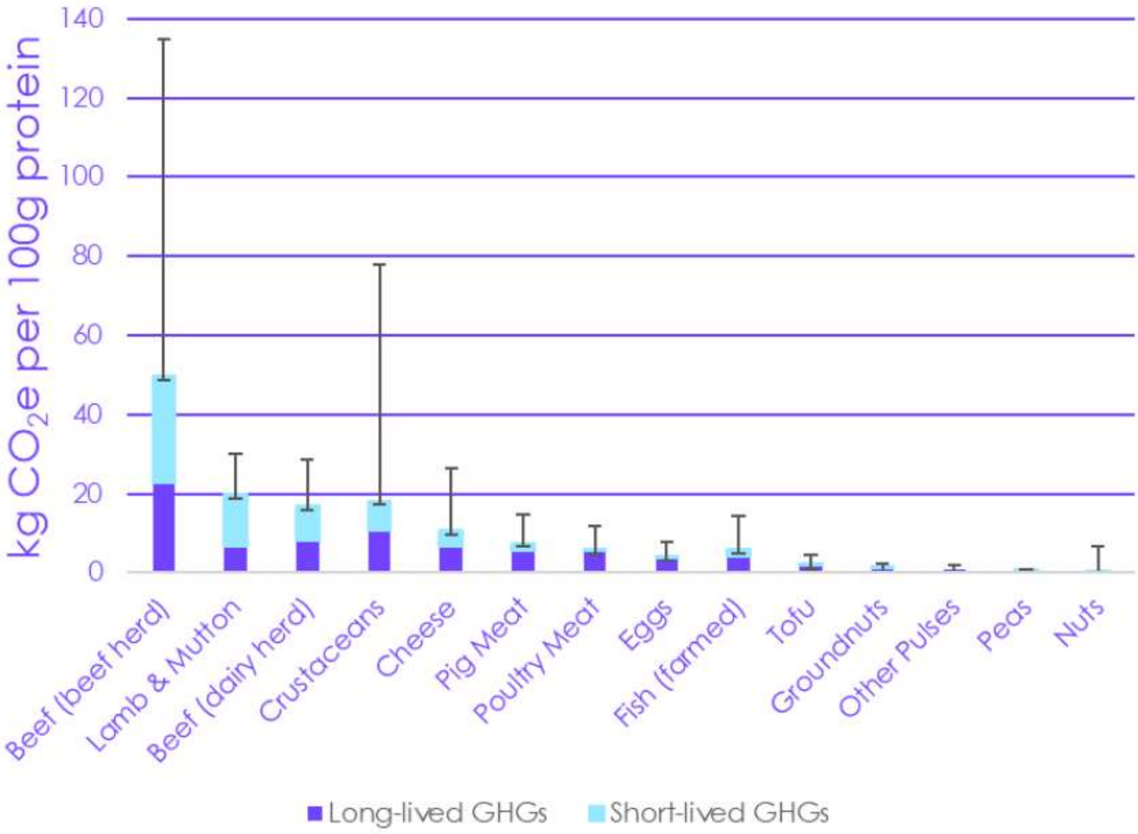

Figure 2.1.2 Global average GHG emissions intensity of protein consumption, for different animal and plant sources of protein. Lifecycle emissions are expressed on the basis of equal protein content. Error bars indicate the 5th and 95th percentile of studies within the database. GHG emissions are aggregated using the GWP100 metric. Long-lived GHGs refer to CO_2 and N_2O and short-lived GHGs to methane. Source: Committee on Climate Change (2020a), originally from Poore, J. & Nemecek, T. (2018).

For GHG emissions, an additional factor on top of lower feed conversion ratio of ruminants in comparison to pigs and poultry, is that ruminants produce methane through enteric fermentation, which adds considerably to their GHG footprint. Estimates of the GHG footprint of different foods through LCAs do vary widely, however, often related to the way in which land use change (LUC) is handled. Searchinger et al. (2018) estimate the global average GHG footprint of beef production at 3-4 times as high as other LCAs at 188 kg CO_2e per kg fresh weight, of which over 75% results from the "carbon opportunity cost" of historically cleared land, which would provide carbon storage benefits if it were not in production. Considering only current LUC for UK diets gives a different picture; the feed and pasture footprint of UK cattle and sheep mostly comes from within the UK and EU where LUC is historic and well-regulated currently, whereas soy makes up a larger proportion of pig and poultry diets and is imported from regions vulnerable to deforestation or other land clearing (WWF, 2020). Note that the values depicted in Figure 2.1.2 are global averages, and UK-specific values may differ.

Despite these variations between livestock types and crop types, in general it is most "efficient" in terms of N_r and GHG emissions per gram of protein eaten by a human to do so directly from plants (although more care must be taken to ensure a balanced supply of amino acids; Weindl et al., 2020).

When considering comparisons between animal types, it is important to recognize several nuances. For example, while the NUE for animals (N feed conversion efficiency) is higher for poultry and pig than for cattle and sheep, the former are typically fed with plant materials resulting from more intensive agricultural activities (depending on prime farmland), while cattle and sheep may graze low quality extensive land. This also means that cattle and sheep can be associated with lower spatial intensity of losses. Where livestock are housed, this also increases some N losses, such as ammonia, for example associated with housing, manure storage and spreading; in general, higher NUE and lower Nr emissions per unit product occur for ruminants in low N input (e.g., extensive grazing) systems (Oenema, 2006; M. Sutton, pers. comm). While poultry is associated with higher feed conversion efficiency than cattle and sheep, it is also often associated with large 'industrial farming' activities. Such 'super farms' can represent major point sources of pollution, with substantial impacts on local nature issues (Dragosits et al., 2002), and can even be seen from space (van Damme et al., 2018).

2.1.1.2 Current protein consumption patterns

In the UK in 2018/19, on average we consumed 76g of protein per person per day (CCC, 2020a), around 50% more than is recommended by WHO guidelines of 50g per person per day. Of this protein, 60% is consumed as animal products, made up of meat, dairy, fish and eggs in decreasing order of importance (CCC, 2020a). This is about average for the EU as a whole, but there is significant variation across countries (Figure 2.1.3).

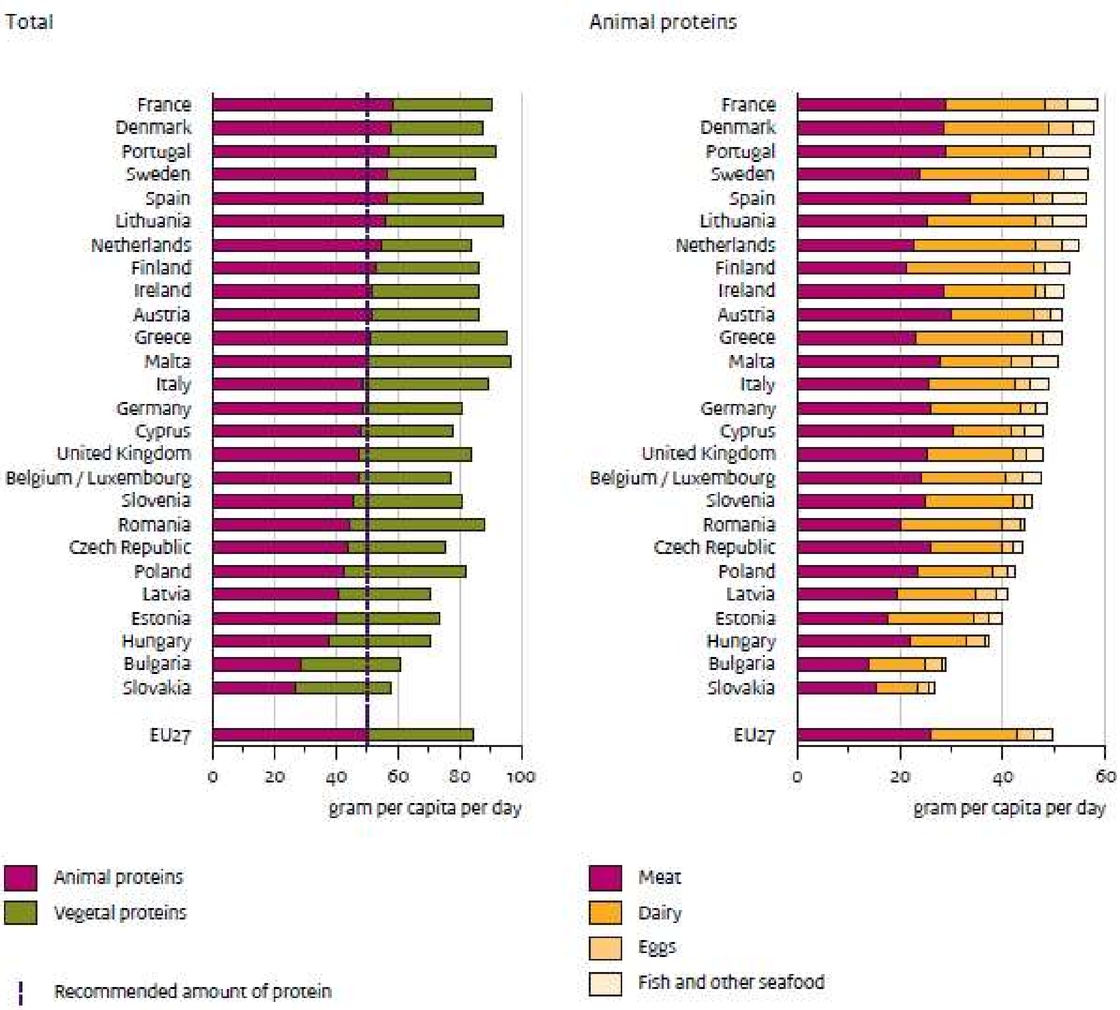

Figure 2.1.3. Protein intake and breakdown of protein sources in the EU by country in 2007. Source: Westhoek et al. (2015)

Associated with this high proportion of animal-based protein in the diet, consumption of red meat and saturated fats are also well over WHO recommendations, increasing the risk of obesity, cardiovascular disease and other chronic health conditions such as colorectal cancer.

There is some evidence from Public Health England's National Diet and Nutrition Survey[1] that meat and dairy consumption in the UK has fallen in recent years, down 6% and 16% respectively between 2008 and 2018. This has been paralleled by an increase in the vegan or vegetarian share of the population, from 1.6% in 2009/10 to 2.5% in 2015/16 (CCC, 2020a). Perhaps more important in terms of overall impact, from a 2020 survey by the Eating Better Alliance (not necessarily as representative as the PHE statistics), 65% of respondents indicated they were willing to eat less meat in future, if they had more knowledge on planning and cooking meat-free dishes.

There is therefore clearly further scope and potential for people in the UK to both reduce their overall protein consumption, and their intake of meat and dairy products in particular. In

[1] https://www.gov.uk/government/statistics/ndns-time-trend-and-income-analyses-for-years-1-to-9

theory, we could reduce meat and dairy consumption without having to substitute this with protein from other sources, although some compensation related to calories might occur. This would have a considerable impact on Nr emissions. Across the EU, production of animal products accounts for over 80% of all Nr emissions from agriculture, and beef and dairy specifically for 56% (Westhoek et al., 2015). The high intake of animal products is also reflected in how agricultural land is used. In 2010, 85% of agricultural land in the UK was used directly or indirectly for livestock production, with 22% being used for livestock feed crops and 63% being grassland. This supplies about 32% of total calorie supply and 48% of total protein supply (CCC, 2020a).

2.1.2 Impacts of shifting diets

As outlined above, reducing consumption of animal protein will reduce the flow of N through the system, and thus will tend to reduce all types of N_r emissions (see Section 2.3.1), avoiding pollutant swapping and other trade-offs. Here, we summarise the results of selected studies which have attempted to quantify the benefits of dietary shift through life-cycle analysis (LCA).

2.1.2.1 Diet scenarios

There is an increasing body of literature considering the environmental benefits of dietary shift. Westhoek et al. (2015) provide a detailed analysis of the N_r emissions reductions associated with alternative diets which is summarized below, including "demitarian" scenarios halving of meat and dairy intake, as well as scenarios of 25% reduction. These are presented alongside a selection of other diet scenarios focusing on GHG emissions and other impacts from the Committee on Climate Change's 6th carbon budget (CCC, 2020a) and elsewhere.

There is significant variation in the level of ambition of the scenarios, the focus of changes, and other assumptions (e.g., parallel reductions in food waste, substitution of animal products with other protein sources), so a selection of scenarios with their assumptions are examined. Unfortunately, quantification of N_r and/or GHG emissions from all scenarios of different studies was not available. Table 2.1.1 below provides an overview of the scenarios considered as well as a quantification of the impact on N_r and GHG emissions where available.

Table 2.1.1 Selection of alternative diet scenarios from the literature, with assumptions made and quantification of impacts on Nr and GHG emissions where available.

Source	Diet scenario	Assumptions	Total Nr loss	NH_3	N leaching and runoff	N_2O	GHG impact
Westhoek et al. (2015)	-50% pork and poultry in the EU (Greening scenario)	No change in food waste, calories maintained by increase in cereal intake	NA	c. -16% (EU)	c. -7% (EU)	c.-5% (EU)	c. -5% (EU)
Westhoek et al. (2015)	-50% beef and dairy (Greening scenario)	No change in food waste, calories maintained by increase in cereal intake	c. -26% (EU)	-26% (EU)	-27% (EU)	-26% (EU)	c. -35% (EU)

Westhoek et al. (2015)	-50% all meat and dairy (Greening scenario)	No change in food waste, calories maintained	-42% (EU)	-43% (EU)	-35% (EU)	-31% (EU)	-42% (EU)
Committee on Climate Change (CCC) (2020a)	Balanced pathway to 2050	-35% meat and -20% dairy, substituted by plants -60% food waste Production emissions only accounted for	NA	NA	NA	NA	c. - 19%
Committee on Climate Change (CCC) (2020a)	Widespread engagement to 2050	-50% meat and dairy, substituted by plants -70% food waste Production emissions only accounted for	NA	NA	NA	NA	c. -37%
EAT (2021)	Planetary health diet – Full waste and no production practice changes	For UK, roughly[1]: -81% red meat -57% dairy 7-fold increase in legumes	-4% nitrogen application(Global)	NA	NA	NA	c.-60% (UK)
EAT (2021)	UK National dietary guideline diet	For UK, roughly: -25% red meat 4-fold increase in legumes	NA	NA	NA	NA	c.-15% (UK)
IDDRI (Poux & Aubert, 2018); FFCC (2021)	Average sustainable European diet in 2050 compatible with agroecology (plus a UK regionalised version from FFCC); -50% animal protein	-6% total calories (EU) -10% food waste (UK) -25% beef consumption (UK); little change to beef production (UK) -60% pork and -66% poultry (UK) Increase in plant-based protein, especially legumes. Includes wholesale shift to organic farming	Increase in crop NUE from 63% -> 92% (Europe)	NA	NA	NA	-38% (UK)

Eshel et al. (2010)	Purely plant-based diet (vegan)	Synthetic modelled diets for a typical US citizen, based on typical fertilization requirements of different crops	-71% decrease in N_r inputs (USA)	NA	NA	NA	NA
National Food Strategy (2021)	National Food Strategy initial diet recommendations by 2032	-30% meat -25% high fat,salt and sugar foods +50% fibre +30% fruit and veg +15% productivity -50% food waste	NA	NA	NA	NA	NA

1. Tabulated data for the UK in terms of grams of protein is not available, but visual estimates have been made based on charts showing changes required in terms of grams of product for Great Britain. NA = quantification not available

2.1.2.2 Impacts on N_r emissions

The scenarios from Westhoek et al. (2015) provide the most comprehensive analysis of impacts of diet shift on N_r emissions. Under the demitarian scenario (–50% all meat and dairy) of Westhoek et al. (2015) (applying to the whole of Europe), the reduction in Nr emissions translates into a 40-50% reduction in N deposition across most of the UK. Synthetic fertilizer application is 23% lower, and there is reduced overall demand for cereals, even with the assumption of some replacement of meat and dairy protein with cereal-based protein (this is due to the large energy and N losses avoided by cutting out a trophic level). Reducing beef and dairy generally has a larger impact than pork and poultry, due to superior feed conversion efficiency of the latter and because beef and dairy represent larger sectors overall. However, as already noted, indoor intensive pig and poultry rearing is associated with greater point-source issues for sensitive areas, whereas beef and sheep can be reared in extensive systems where N inputs are lower and emissions more diffuse. These scenarios all assume that calorie consumption is maintained by substituting with additional cereal intake, which given that calorie intake is around 10% higher than needed means that if this assumption were removed the reductions in N_r emissions would be even greater. Overall, the “Nitrogen on the Table” demitarian scenario of Westhoek et al. (2015) roughly doubled food-chain NUE from 21% to 47% (high prices scenario) or 41% (greening scenario).

The IDDRI diet is based on the $\text{TYFA}_{\text{regio}}$ model (FFCC, 2021) and is part of a wider sustainable agroecological farming system in the UK. Analysis of the N cycle impacts of the agroecological scenario are only available at the European scale currently, though restricted to an input/output balance. These predict an increase in average crop NUE from 63 to 92%, largely related to the reliance on biological N-fixation for inputs of N into the system, and a large increase in consumption of N-fixing leguminous crops. In this system, beef production is maintained to fulfil its role in maintaining fertility and biodiversity in the wider system. Poultry, pork and dairy production are strongly reduced, as they are reliant on grain and so compete more directly for arable land.

The EAT planetary health diet (EAT, 2021) is based around the concept of planetary boundaries and involves even more extreme recommendations for UK diets. The focus of changes differs from the IDDRI diet, with little decrease in poultry consumption (for the G20 as a whole, and assumes the UK is consistent), but an 85% reduction in red meat consumption and 57% reduction in dairy consumption. However, as with the IDDRI diet, legume, fruit and vegetable consumption is increased markedly. The impacts on N at the UK scale are not quantified, but globally diet change was estimated to result in around 4% decrease in N application, which includes increases in parts of the world currently operating within planetary boundaries.

Although based on US diets and using a different modelling framework, work by Eshel et al. (2010) suggests that a shift to vegan diets would result in even greater reductions in Nr inputs (71% reduction), and therefore Nr emissions. However, this result contrasts with a review by van Zanten et al. (2018) which concludes that around 25% less arable land (and corresponding Nr inputs) is required when a small number of livestock are kept, fed only on waste, crop residues and grassland.

2.1.2.3 Impacts on GHG emissions

For greenhouse gas emissions, the CCC's 6th carbon budget analysis provides the best reference for the UK. There are several scenarios depending on the level of innovation, behaviour change and other factors (see Figure 3.5.2), but the two picked out here are i) the "Balanced Pathway" scenario which represents the "central" pathway which forms the focus of UK government policy attention and ii) the "Widespread Engagement" scenario, which is the basis of much of WWF's thinking.

The Balanced Pathway and Widespread Engagement scenarios forecast a 19% and 37% reduction in agricultural GHG emissions respectively due to dietary and food waste reduction combined (with diet shift accounting for the majority of this). The CCC reports do not specify different reductions in different types of meat, so it is assumed that the same percentage reductions are applied to all meat types. Reduction in enteric fermentation is the largest driver, so reducing lamb, beef and dairy consumption is likely the most important factor in their modelling, as ruminants generally have considerably higher GHG footprints per unit of production. However, in current systems, imported soy meal is an important part of pig and poultry diets, and the carbon emissions or removals from reduction / reversion of land use change have not been taken into account in the CCC analysis for the 6th Carbon Budget. Soy import could fall by 76%, if waste-derived and grassland resources are still maximally utilised as is assumed in the scenario.

The IDDRI diet predicts a 38% GHG emissions reduction from both dietary changes and changes in production practices. These largely arise from reduced emissions from manure management, synthetic fertilizer manufacture and N_2O from soils, but enteric fermentation emissions do not decrease much due to the maintenance of beef and lamb in the diet.

The EAT planetary health diet would produce around a 60% decrease in all food-consumption related GHG emissions in the UK (EAT, 2021). The biggest driver is large reductions in enteric fermentation emissions from beef cattle and sheep.

Current UK consumption patterns fail to meet UK dietary guidelines, with higher red meat and lower legume consumption than recommended. If the UK adhered to its own guidelines, there would be a 15% reduction in GHG emissions, largely related to a 25% reduction in red meat consumption (EAT, 2021). The recently published National Food Strategy advocates a 30% reduction in meat consumption in the UK by 2032 based on healthy diet and alignment with the 5th Carbon Budget recommendations, and although it does not quantify GHG emissions or Nr savings, it estimates that around one third of agricultural land could be saved.

2.1.2.4 Health impacts

All of the dietary scenarios outlined above are likely to bring health co-benefits. These are chiefly related to lower risk of colorectal cancer from reduced red meat consumption, and reduced risk of cardiovascular disease and stroke from lower saturated fat intake (Westhoek et al., 2015). In the UK, increasing consumption of legumes to match dietary guidelines would also add considerable amounts of beneficial fibre to the diet.

As a source of protein, meat, fish and dairy products are very well balanced compared with many plant and fungal sources. Reduced meat consumption therefore brings with it an increased need for awareness and planning of protein (and micronutrients) sources in the diet, to ensure that nutritional requirements are met. Nevertheless, a review of the scientific literature by Costa Leite et al. (2020) concluded that even in their most restrictive forms (e.g., vegan), low environmental footprint diets can be compatible with health goals, although solid trustworthy information needs to be given to those wishing to follow them.

To sum up, there is general agreement across different scenarios that a reduction in animal protein consumption will result in considerable reductions in Nr and GHG emissions, and health benefits. However, depending on the pathway taken, it will emphasise some benefits over others. Reducing red (ruminant) meat consumption in particular (as advocated by the EAT planetary boundary diet) would tend to provide the largest health benefits and reduction in Nr and GHG emissions overall. Reducing poultry, pig and dairy consumption would likely provide fewer health benefits, and smaller Nr and GHG emissions reductions overall (in particular, smaller reductions in methane emissions), but would be effective in reducing point source Nr emissions, reduce food-feed competition and imports of feed from environmentally sensitive locations, and facilitate sustainable (from a land-use and biodiversity perspective) agroecological systems, with associated benefits for biodiversity.

2.1.2.5 Differences in the assumptions and ambition of different diet scenarios

There are a number of interesting assumptions made in the scenarios above and the rationales behind the level of ambition.

Firstly, the scenario variants presented from Westhoek et al. (2015) and the CCC's 6th Carbon Budget assume that reduction in local consumption will result directly in reductions in local production, as opposed to increases in exports. In contrast the IDDRI scenario for the UK takes into account an increase in beef exports from the UK, so that production is almost unchanged despite a 25% reduction in UK beef consumption, based on the fact that UK beef has relatively good environmental and animal welfare credentials so it can be exported as a premium product. Clearly, modelling the responses of farmers in the UK to changes in domestic demand is complicated, and depends also on changes in demand in other countries, so as a first pass the assumption of UK production levels falling in step with UK consumption seems reasonable. However, it should be noted that this assumption could make the estimated reductions on the optimistic side.

Secondly, it is interesting to note the relationship between the dietary changes assumed and concepts of what is sustainable. The diet scenarios presented by the CCC and Westhoek et al.

(2015), although fairly ambitious, do not seem to have a particular theoretical basis – they are illustrative of what could be achieved. In contrast, the IDDRI diet takes into account available land, productivity of different systems and crop rotations, nutrient (especially N) balances to design a diet which sits within these constraints, and the EAT-Lancet diets use the concept of planetary boundaries against a variety of criteria to define acceptable consumption ranges. Van Zanten et al. (2018) review land-use modelling studies, and define the amount of animal protein that can be consumed from livestock fed purely on leftovers, at 9-23g per person globally. The Widespread Engagement scenarios proposed by the CCC and the -50% all meat and dairy diet from Westhoek et al. (2015) broadly seem to match the ambition of the diets quantitatively based on sustainability constraints or planetary boundaries – for example, 23g of protein from animal sources is a 50% reduction from the 46g per day currently consumed in the UK.

Thirdly, the importance of legumes / pulses as a substitute protein source for reduced meat and dairy consumption differs among studies. In the scenarios of Westhoek et al. (2015), it is assumed that reduction in meat and dairy consumption is compensated for (on a calorie-matched basis) by increased cereal consumption only, with consumption of pulses held constant. In contrast, the IDDRI diet (Poux & Aubert, 2018) and EAT planetary health diet (EAT, 2021) assume that increased consumption of pulses plays an important role, whilst cereals increase to a lesser extent. In the IDDRI study, growing pulses serves to increase biological N fixation into the system, but also reduces Nr emissions compared with growing other crops. As mentioned in Section 2.1.1, pulses have a far lower Nr emissions footprint per gram of protein delivered, than cereals or other plant products. Therefore, the total Nr savings estimated by the Westhoek et al. (2015) scenarios may be conservative. However, considering pulses also as a source of carbohydrate rather than just protein, in the UK peas (and field beans for animal feed) tend to be lower yielding than cereals per area, so would require more land to deliver the same number of calories.

Finally, another interesting key message from EAT (2021) relating to the UK is that even if UK dietary guidelines were met this would not be sufficient (according to EAT) to align with the planetary health diet. According to EAT, dietary guidelines need to specify more vegetables, legumes and nuts, as well as 81% less red meat, and 57% less dairy in order to achieve the planetary boundaries. The National Food Strategy recommendations (NFS, 2021) - which could possibly be reflected in future UK dietary guidelines – are more ambitious with respect to meat consumption (at 30% reduction of all meat) than current UK guidelines as they incorporate climate and nature commitments. However, they still fall short of the EAT planetary health diet.

2.1.2.6 Opportunities and implications for "freed up land" through diet shifts.

Reduction in livestock numbers, all else being equal, leads to a large area of land being freed up. Van Grinsven et al (2015) estimated that reducing all animal products by 50% cuts the land demand of European consumption by more than 100 million hectares. In the CCC's balanced pathway scenario, diet change alone is estimated to free up around 4.4 million hectares of land in the UK (c.18% of all "used" land in 2019) (Committee on Climate Change, 2020a; Figure M.7.7). There are choices in how that freed up land can be used, with different implications for the ultimate impact on N_r and GHG emissions, as well as other impacts.

Westhoek et al. (2015) present two different variants of each scenario which both assume that land is still used for agriculture: i) the "greening" variant (presented in Table 2.1.1) where spared land on productive arable land is used for biomass production for energy and spared grassland is used for extensive grazing; and ii) the "high prices" variant, where the spared land in all cases is used for additional cereal production for export. The greening variant of the -50% all meat and dairy scenario sees N_r losses reduced by 42%, compared with 37% in the high prices scenario.

The assumption by Westhoek et al. (2015) that the land spared by reducing meat and dairy consumption is still used for agriculture is quite a conservative one, and may mean that the possible reductions in Nr emissions are underestimated in that study, compared with if land were used for other purposes. The IDDRI diet also implicitly assumes that land spared by reductions in meat and dairy consumption is used for more extensive, lower-productivity organic farming, although the change in farming system could be simultaneous with dietary change.

In the 6th carbon budget scenarios, the CCC assume that most of the land spared through diet changes, reduction in food waste and productivity improvements combined is used to sequester carbon, through afforestation/reforestation, farmland trees and peatland restoration, with a reduction in permanent grassland and rough grazing, and some bioenergy crops.

This is not the right place to enter into a lengthy discussion of land use scenarios, but it must be remembered that types of dietary change, the use for any spared land, and the particular environmental impact and geographic scope in question are all very interlinked into broader socio-ecological system scenarios. The focus of the CCC's scenarios is only the UK GHG emissions and removals balance, so does not consider what might be best for global GHG emissions, for minimising N_r emissions or for local or global biodiversity. It focuses primarily on freeing up UK land, rather than reducing imports (although it does achieve this). The focus of the IDDRI scenario is to create a functional agroecological system which does not rely on imports of vegetable protein or synthetic fertiliser, restores soils, reduces N and phosphorus emissions, and restores local, agriculture-adapted biodiversity in Europe. This requires sufficient demand for ruminant meat or milk to maintain the extensive grassland systems delivering many of these benefits. But one could take other perspectives: for example, from a global biodiversity conservation point of view (in terms of minimising species extinctions, at least), could it be even better for the UK to use spared land to export more food, feed or energy crops, to (in a globalised market) spare further deforestation for oil palm or soy?

2.1.3 The impact of pet food

The discussion above focuses on human diets, but a recent study by Okin (2017) estimated that the animal product consumption of pet cats and dogs in the USA makes up around 33% of the total for humans and pets combined. To some extent cat and dog food makes use of by-products which contribute proportionally less to demand for additional livestock production, but use of human-edible meat products for cats and dogs is on the rise[2]. Okin (2017) estimated that cats and dogs are responsible for around 25% of the GHG emissions related to animal product consumption in the USA. In the UK, the number of dogs and cats per capita is lower than in the USA (around 0.14 vs 0.24 dogs and 0.16 vs. 0.27 cats per capita in the UK and USA respectively (Okin, 2017)). Therefore, the equivalent figures in the UK are

[2] https://www.theguardian.com/global/2018/jun/26/pet-food-is-an-environmental-disaster-are-vegan-dogs-the-answer

likely lower than those cited above for the USA, although it is difficult to estimate due to there being several relevant components where data is not easily available (e.g., differences in average size and diet composition of pets). Nevertheless, reducing meat consumption by cats and dogs would undoubtedly also make a sizeable impact on the Nr footprint of livestock production in the UK.

Possible options to do so include:

- Reducing the number of cats and dogs *per se*. However, cat and dog ownership are important aspects of a person's identity, so it is not obvious how this could be achieved.
- Replacing meat with plant or fungal protein.
    - For dogs which are omnivorous, it is theoretically possible to provide a fully vegetarian diet[2], but extreme care would need to be taken to ensure the right balance of nutrients is consumed as well as to maintain digestive health. Smaller reductions in meat content should be quite feasible.
    - Cats are thought of as "obligate carnivores", as they require certain compounds (including carnitine and taurine) usually only present in meat. This makes it much more difficult to adjust cats' diets. Non-animal-based replacements for meat could be feasible if they include supplements of these essential compounds, but these have not yet been tested[2].
- Replacing traditional meat and fish with insect protein (see below).

Carcass-balance needs to be considered as a factor linking consumption of animal products by pets and humans, given that certain parts of animal carcasses are not permitted in the human food chain, or have low demand. For example, if dogs reduce animal product consumption but humans do not, this could result in a greater quantity of animal by-products being wasted.

2.1.4 Alternative protein sources

Another dimension to dietary change, in addition to switching from animal-based to plant (or fungal) protein sources, is the adoption of novel protein sources in the diet. Here we briefly describe the impacts of 3 alternative protein sources as reviewed by Eory et al. (2019), from an N_r emissions angle.

2.1.4.1 Synthetic / lab-grown meat

Under the CCC's "Widespread innovation" scenario, around one-third of the reduction in meat and dairy consumption is substituted by "lab-grown" meat rather than plant or fungal protein.

Lab-grown meat involves creating a cell culture in a closed-system to grow muscle tissue, by providing the appropriate nutrients and other stimulation required. Inputs can be derived from cyanobacteria and plant extracts. The technology is still very immature and not commercially viable, but cradle to factory gate assessment estimated that if cyanobacteria are used as feedstock, the production of cultured meat would involve 7–45% less energy, 78–96% lower GHG emission, 82–96% lower use of water and 99% lower land use than conventional meat production. Due to the closed system, N_r emissions would likely be very low or non-existent. However, for the moment the energy consumption of cell culture is still very high.

2.1.4.2 Insect protein

Insects such as blowfly larvae and mealworms have extremely high feed-conversion efficiency compared with conventional livestock, because they require little energy for maintenance, grow quickly and have a high proportion of edible body parts. This factor alone would make them favourable from an Nr emissions perspective compared with conventional livestock products. However, compared with plant protein, to become comparable in GHG (and likely in Nr terms also), waste needs to be used as a feedstock as opposed to crops. This is feasible for some kinds of insects such as blowfly larvae, and livestock manure can even be used, which would offer an alternative route to adding value to livestock manure than anaerobic digestion.

Insect protein can be used for livestock feed (for example in aquaculture), but the benefits would likely be greater if consumed directly by humans (or pets).

However, there are behavioural and regulatory barriers to be overcome in the UK to scale up use of insect protein in food or livestock feed. Currently, insects can only be fed to ruminants, pigs or poultry as processed lipids and hydrolysed protein, or in live form, but not as simple insect meal (though this can be included in pet food) (WWF, 2021b). In addition, the rules around feedstocks for insect farming currently do not allow certain kinds of food waste (especially those containing animal-derived products) and livestock manure, which limits the opportunities to make use of these more sustainable waste streams.

To overcome behavioural barriers to direct human consumption, substitution of conventional livestock protein with insect protein powder in processed foods is likely to be successful, whereas consumption of whole insects is more of a challenge.

2.1.4.3 Microbial protein

Microbial protein is derived from cell culture of algae, bacteria or fungi on an industrial scale. Foods such as Quorn and Marmite are current forms of microbial protein. As with insects and lab-grown meat the comparative impact of microbial protein depends on the source of nutrients and energy. One kind of microbial protein system recently developed is “agriculture free”, in that it derives its energy for growth autotrophically from hydrogen, rather than organic molecules. This process has a high energy demand (electricity required to produce hydrogen through electrolysis), but has an almost zero land footprint and likely very low N_r emissions due to a closed-system process.

2.1.5 Key interventions to bring about dietary change

Despite nuances with regard to which types of animal products have higher or lower impact against different metrics, the overall message that reduction in animal product consumption will bring multiple benefits is a simple one. However, shifting people’s diets is not a straightforward path, but one that requires behavioural change. Food choices are influenced by a variety of interacting factors, including food prices, gender, health, income, geography, social identity and networks, exposure to marketing and media, and ease of access to supermarkets and other food outlets. Some choices are based on a reflective decisional system (rational choice) – for example those based on labelling and price, while others are based on automatic behaviour (Godfray et al., 2018) such as responses to the choice architecture presented in a supermarket or restaurant menu. Strategies should aim to address both of these mechanisms.

There are calls for more government intervention to bring about dietary change, but it is important to recognize that changes will require the buy-in of all actors in the food chain, including consumers, retailers, processors and primary food producers.

2.1.5.1 Types of intervention

There are a range of different types of intervention possible, ranging from strong interventions such as banning certain products, through to disincentives such as taxes and weaker interventions such as working with stakeholders to enable consumers to make better choices, labelling regulations, and education campaigns.

There are also different stakeholders that can be targeted by interventions. Some interventions will target consumers directly, whilst others target food producers, processors or retailers, and the impact could differ considerably. For example, taxes directed at producers may lead to lower domestic production but increased imports (Ranganathan et al., 2016). Taxes aimed at food processors related to inclusion of particular ingredients in a product may tend to incentivise reformulation of a product, as opposed to causing an increase in prices for consumers. Interventions can be arranged on a ladder from “hard” to “soft” as illustrated in Table 2.1.2 below.

Table 2.1.2 Review of effectiveness of different kinds of intervention for dietary change from a variety of studies. Source: Latka et al. (2021).

Freedom of consumer choice

Freedom of supply chain actor choice

Intervention	Brambila-Macias et al. 2011	Capacci et al. 2012	Garnett et al. 2015	Hyseni et al. 2017	Mazzocchi 2017	Mozaffarian et al. 2018	Sassi et al. 2009	Modelling instruments
Information campaigns/ dietary guidelines	Absent for short-lived interventions, awareness raised	Suggestive, small	Unclear long-term effects, awareness increase	Small effect size, uncertain long-term effects	Strongly effective	Limited overall direct effectiveness	+18.4g V&F	Preference shifters
Compulsory information on products (e.g. labelling)	Uncertain, more promising for simple labels, contributing to informed choice	Mixed	Inconsistent consumer responses	Effective, but interpretation difficulties	Suggestive, slightly effective	Mixed, effectiveness depending on knowledge and attention	+9.9g V&F -0.4% fat%E	
Food advertising regulations	Weakly effective	Suggestive, uncertain long-term	Significant	Appears effective	Suggestive, short-term, effective if comprehensive	Sustained, effective if implemented across formats	+0.4% fat%E	
Ensuring choice availability (e.g. school food programs)	Effective, limited to target group	Suggestive	Positive impacts on diets in intervention setting	Modest to small effect size, uncertain long-term effects	Suggestive, strongly effective in intervention setting	Sustained, effective	+38g V&F -1.6% fat%E	
Financial (dis-) incentives through taxes/ subsidies	Effective, but intrusive and potentially regressive	Suggestive, mixed, uncertain regarding distributional impacts	Combinations of taxes and subsidies effective	Consistently effective, diet change price dependent, substitutions can offset improvements	Suggestive, strongly effective	Effective, most promising as combination of incentives and disincentives	+8.6g V&F -0.8% fat%E	Taxes/ subsidies
Restricting/ eliminating choice	Seems effective, limited evidence	Suggestive	Positive impacts on diets in intervention setting	Appears powerful, but neglected	Suggestive, mixed effects	Promising, but neglected	-	Trade/ production quota

Evidence for the effectiveness of policy interventions on dietary change suggests that in general harder interventions are more effective, but the situation is mixed and some softer ones such as ensuring choice can be very effective in the target setting (Table 2.1.2). Harder interventions may also generate more political push-back, though pursuing parallel strategies to inform and educate and promote alternatives may ameliorate backlash (Wellesley et al., 2015).

Ranganathan et al. (2016) explore the concept of a “shift wheel”, which encompasses multiple – potentially parallel- strategies to achieve change (Figure 2.1.4), and consider some useful examples of previous shifts in behaviour that have been accomplished through government intervention, that can also be instructive for reductions in meat and dairy consumption.

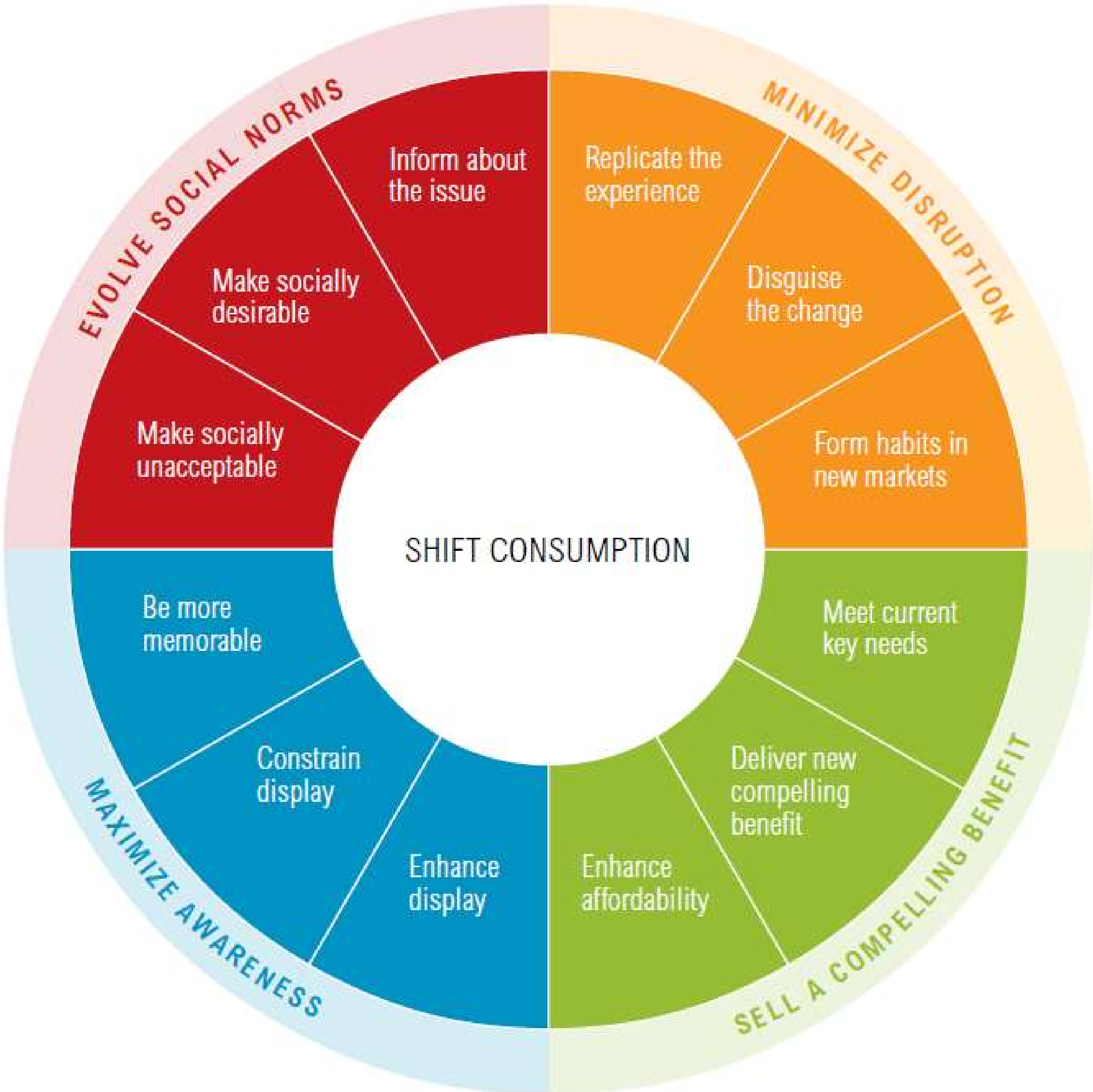

Figure 2.1.4. The “Shift Wheel” of strategies to shift consumer behaviour, from Ranganathan et al (2016).

Minimizing disruption includes food industry marketing and product formulation strategies which create meat-replacement products (e.g. Beyond Meat or Quorn), or reformulate products to replace animal products with vegetable ones.

Selling a compelling benefit also largely relates to marketing to appeal to consumers' conscious choices. Products may be sold on the basis of benefits to health or quality. Health benefits are certainly a key point of leverage for plant-based foods over animal-based products.

Financial incentives and disincentives are also included under this heading, as lower price is a key motivator of conscious choice. Taxes are a key tool for governments to manipulate price, and there is some evidence that they work to change consumer habits, but that care must be taken in design to avoid unintended consequences (Ranganathan et al., 2016), for example:

- Taxes on domestic producers have to be matched by taxes on imported products, to avoid consumers buying more imported meat, exporting environmental impacts;
- Taxes should be designed to disincentivize all similar products which could be substituted for one another, to avoid e.g. a tax on beef causing an increase in poultry consumption, rather than plant-based protein;
- Taxes should be implemented over a broad enough area to prevent people crossing borders to avoid them (as occurred during the short-lived Danish fat tax in 2011-2012).

Maximising awareness refers to regulations that manipulate the food environment, which can encompass outright bans (in the case of trans-fatty acids in many countries), restricting marketing or placing of certain products, or actively placing certain foods. Strategies could include not separating out vegetarian options on menus or putting them first to make them "opt out" rather than "opt-in"; not putting vegan food in a separate aisle in shops, to enforce normality, and decreasing portion size of meat in restaurants.

Behavioural mechanisms for voluntary change focus on making sustainable food:

- Easier – more available and more convenient. This could include providing foods in an easily cooked form (e.g. canned rather than dried chickpeas), as well as simple recipes;
- More appealing – nice tasting, with a wealth of good recipes;
- More normalised - removing stigma around vegetarianism and veganism and emphasizing positive messages. Currently, meat consumption is viewed as "normal", so less an active choice than an assumption about centre-point of meals (Godfray et al., 2018).

Evolving social norms can include:

- Information campaigns, which may or may not be government led. Government public health messages on drinking, smoking and road safety are prominent in the media, so there is no obvious reason why the government could not also lead on the issue of meat and dairy consumption;
- Regulations around labelling and certification. This helps to ensure plant-based alternatives can be compared on a like-for-like basis with meat products. Standardised labelling such as traffic-lights has been implemented for health measures, but sustainability labelling (perhaps due to the complexity of issues) has mainly relied upon certification schemes such as MSC seafood, Rainforest Alliance coffee etc., rather than the health-like traffic light approach. There is in general limited evidence for the effectiveness of sustainability labelling (Godfray et al., 2018);
- Campaigns by well-known figures and NGOs to highlight issues (such as shark-fin soup and battery hens);
- Provision of recipes for non-meat-based dishes to make consumption more normal.

A key recommendation from Ranganathan et al. (2016) is that governments should first set a target, then consider what elements of the shift wheel can be applied. How the shift wheel could be applied in the UK to reduce meat and dairy consumption depends on the landscape of animal and plant-based food consumption, including who the consumers are, and why, when and where consumption is happening.

The National Food Strategy (NFS) (2021) recently undertook panel discussions and public polls on different policy options. There was overwhelming support for restrictions on sales and marketing of "junk food", but also high levels of resistance to a meat tax (Figure 2.1.5). Fifty percent of people supported the idea of the government setting targets for meat consumption reduction, and 48% of people supported taxes on "processed" meat, whereas 58% of people either somewhat or strongly oppose a tax on fresh meat.

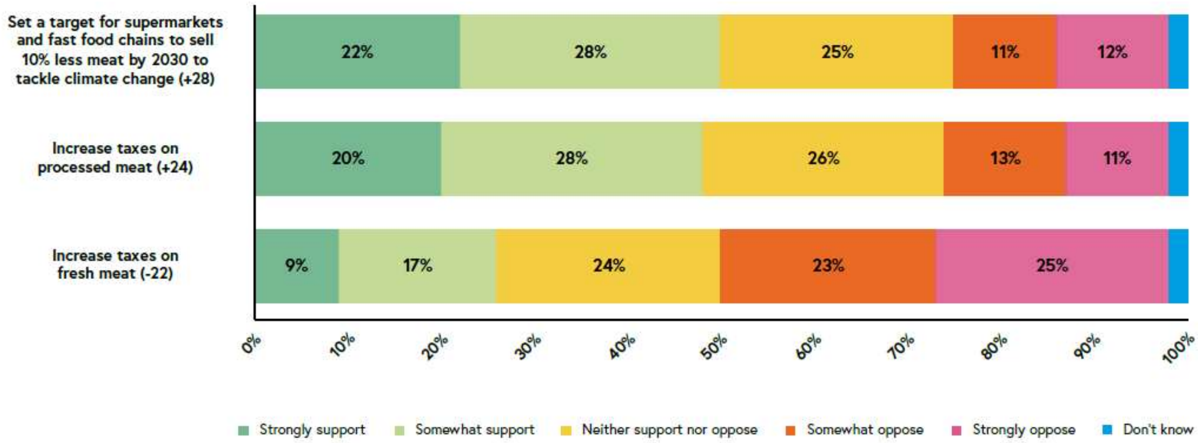

Figure 2.1.5. Results of a poll to understand attitudes to government interventions to reduce meat consumption. Source: National Food Strategy (2021)

An actual ban on meat products was not discussed, but can be assumed to be potentially even more unpalatable than a meat tax. Wellesley et al. (2015) surveyed public attitudes in 12 countries (including the UK), as well as focus and stakeholder groups, to understand what would enable them to change their diets. They concluded that:

- Focus groups all indicated they thought it was the role of government to lead, but policymakers overestimate the extent of backlash so are slow to act. The NFS also found higher tolerance for government intervention than expected;
- Awareness raising among consumers is not the whole solution, but is a first and vital step. Education and labelling will help reduce backlash against stronger government intervention, as currently public understanding is low (as the issues are complex) compared to other issues such as energy;
- There is quite a lot of misinformation and polarised arguments in the mainstream media, related to narrow consideration of different impacts, or based on ideology rather than evidence. This leaves the argument open to pressure groups. The issue is complex, but messages must be simple - less meat and dairy will always be good, so should be the main message;
- Broaden the message – emphasise co-benefits like health and cost, rather than just appealing to environmental conscience;

- Support innovation.

A final point made by Wellesley et al. (2015) is that although there are significant behavioural, social and knowledge barriers to dietary shift, compared with other changes we need to make in society to solve environmental problems (e.g. retrofitting the UK's ageing housing stock), there are no large technological or financial barriers to change. In the UK, given the right motivation, environment and choice there is no reason why individuals and families could not reduce meat consumption from one day to the next, likely saving money in the process.

As discussed above, interventions to change diets to reduce meat and dairy consumption can be one of the most effective tools in reducing N pollution both in the UK and abroad. This is because meat and dairy products have the highest N footprint. However, it is less clear what impact "on-farm" interventions might have on meat consumption.

The most obvious potential link between on-farm interventions and meat consumption is via prices. Whilst some measures can potentially save farmers money (reducing surplus fertiliser inputs, compliance with regulation to avoid financial penalties) others may increase the cost of meat. This may be due to:

- High up-front investment or running costs (for example purchasing low-emission manure spreading equipment) increasing production costs; or
- Lower levels of production, meaning that a larger margin needs to be made per kg of meat sold in order to maintain farm viability.

If these additional costs are passed onto consumers and they do not have access to cheaper meat options, then in theory meat consumption could fall (at least within a subset of consumers). More generally, there is a "chicken and egg" situation in relation to meat consumption and production – will dietary shifts drive changes in production, or will it be the other way around?

In the scenario of proactive reduction in production levels by farmers (e.g. through extensification), this would also reduce meat supply, which if not fully offset by increased imports could also further push up prices. However, in both of these cases farmers are vulnerable to the risks of taking action, as they hold little negotiating power in food supply chains and risk becoming uncompetitive against other farmers who have not taken action, either within the UK, or abroad via imports (UNECE, 2021). It is therefore vital that all actors in the food chain (particularly supermarkets in the UK) accept responsibility and help spread the cost of implementing expensive measures across the food chain. Given the relentless competition between supermarkets to push down the price of milk, it is not easy to see how an increase in meat prices across the board could occur without government intervention, such as setting minimum standards and ensuring imports do not undercut domestic consumption. Currently, whilst all supermarkets sell premium meat lines (e.g. organic) with a lower N footprint, this is alongside cheaper "conventional" meat, so is unlikely to reduce overall meat consumption.

Retailers would likely be very unwilling to deliberately fall short of consumer demand, so in the case of proactively reduced UK production there would be pressure to increase imports. This leads to the realisation that in a globalised food system, the UK agriculture sector taking proactive action alone may not be effective in reducing meat consumption in the UK via prices, unless global action is also taken.

More expensive meat in general may result in a shift for some consumers away from more expensive (per kg) beef and lamb to cheaper pork and poultry, without reducing the total quantity of meat consumed.

2.1.5.2 UK recommendations from the National Food strategy

The NFS makes a number of recommendations related to dietary change. Due to the high levels of public resistance, the UK government has already stated that it has no plans to introduce a meat tax. Therefore, the NFS acknowledges that although public acceptance of a meat tax may grow over time, in the meantime it recommends other strategies. Its key recommendations related to diet shift are:

- **Introduce a "reformulation" tax on salt and sugar sold for use in processed food or food service.** This is not a tax on meat, but illustrates the useful principle that targeting taxes toward food processors rather than consumers incentivises reformulation where possible, without causing price rises for consumers for many foods. This concept could also usefully be applied to animal products;

- **Introduce mandatory reporting for large companies** to allow scrutiny, accountability and better environmental decision making by consumers;

- **Launch a new "Eat and learn" initiative for schools** to build solid cookery skills, to give lifelong confidence in cooking a wide range of healthy foods. This can facilitate switches from some foods to others;

- **Trial a "Community Eatwell" Programme, supporting those on low incomes to improve their diets**. This would allow GPs to prescribe healthier diets and food education to patients suffering from effects of poor diets;

- **Invest £1 billion in innovation to create a better food system.** For diet shift, this could support reformulation of processed foods, piloting behaviour change approaches, local healthy eating initiatives and also development of alternative protein sources (such as microbial protein);

- **Create a National Food System Data programme** to link data on land use and farming practices with food processing and retail, to allow a better understanding of the environmental impacts of the foods we eat for all consumers;

- **Set clear targets and bring in legislation for long-term change.** The NFS envisages an expanded role for the Food Standards Agency (FSA) to take on the role of monitoring, reporting, creating a "reference diet" in line with the NFS recommendations, and creating a consistent food labelling system for environmental sustainability.

2.2 Reducing food waste

The proportion of food that ends up as waste is a key factor in determining the N requirements of growing a sufficient harvest. When food waste is high, farmers must raise levels of production, which in turn increases the amount of N required. In addition, if a large proportion of supplied food is lost from the supply chain to the environment, then the excess N in the wasted food can lead to a number of negative impacts on the ecosystem and human health, whilst also having considerable economic impacts for the food industry itself.

It is not only the N implications of food waste that are of concern: recent estimates of the contribution of food waste to total anthropogenic greenhouse gas emissions range from between 4 and 6% (OurWorldInData.org, 2020; WWF, 2021a). As food decomposes it produces methane, a potent greenhouse gas. Whilst over the past few decades, the UK has seen a major reduction in the emissions of methane from landfill as a result of improved landfill methane capture technology (Brown *et al.*, 2021), continued efforts to reduce food waste sent to landfill will bring the co-benefit of helping to reduce the UK's carbon footprint.

Despite these impacts, a large proportion of food continues to be wasted. Typically, food waste is thought of in terms of waste by the consumer. Indeed, Grizzetti et al. (2013) estimate that the food lost at the consumption stage equates to 9% of total global food consumption. However, food is wasted at all stages of the supply chain, across its growth, harvest, storage, retail, and at final consumption. For example, a recent WWF study (WWF, 2021) estimates that 15% of all food produced globally is wasted at the farm stage. In particular, waste at the production and handling and storage phases can be significant contributors to overall losses. As the figure below illustrates (Figure 2.2.1), the trends vary by region, with 33% and 10% of food waste being attributable to production and handling and storage in Europe respectively. Indeed, it could also be argued that the over-consumption of protein in present-day diets represent a form of food waste, as excess protein would be excreted from the body.

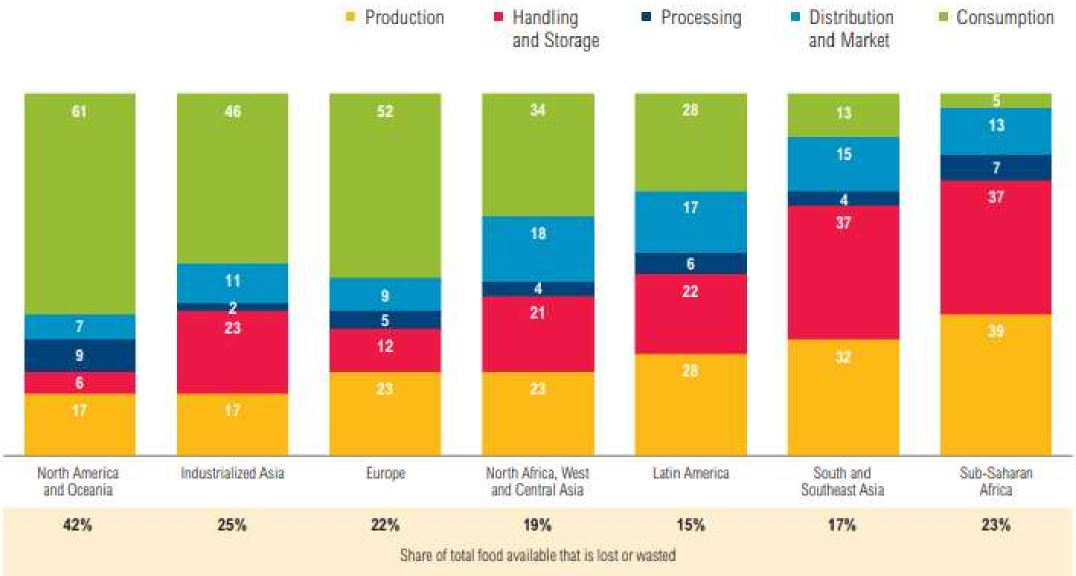

Figure 2.2.1. Food Lost or Wasted by Region and Stage in Value Chain, 2009 (Percent of kcal lost and wasted). Note: Number may not sum to 100 due to rounding. Source: World Resource Institute (WRI, 2013), Analysis based on FAO. 2011. Global food losses and food waste—extent, causes and prevention. Rome: UN FAO.

As food is wasted at every point along the supply chain, the measures that can reduce food waste are varied and need to be combined as a package of actions each with different targets, scales, and responsible entities. The table below (Table 2.2.1), from Jeswani *et al*. (2021) estimates food lost in different parts of the supply chain by food groups in the UK. It illustrates that whilst consumption remains the largest source of food waste, the contribution of primary production, processing and manufacturing, and distribution are all significant contributors, particularly on an individual food-group basis. For example, primary production and manufacturing are dominant factors in food waste for potatoes. In addition, the waste of ingredients like pulses, although not included Table 2.2.1 are likely to be higher on the production side given the ease and longevity of storage methods in households.

Table 2.2.1 - Quantities of food consumed and wasted in the UK by food group and stage of waste (Jeswani et al., 2021). All values are in kilotonnes

Food group	Food consumption	Food waste				
		Primary production	Processing and manufacturing	Distribution	Consumption	Total
Cereals	*8227*	*1245*	*761*	*161*	*1913*	*4080*
Wheat (and products)	5001	925	470	98	1163	2656
Barley (and products)	1557	242	138	30	362	773
Others	1668	77	153	33	388	651
Sugar and sweeteners	*2793*	399	*123*	*27*	*104*	*653*
Sugar	2227	399	98	22	83	601
Honey and sweeteners	567	-	25	5	21	51
Oil crops and vegetable oils	*833*	*21*	*50*	*8*	*53*	*131*
Rapeseed	457	20	24	4	21	69
Soya beans	129	-	11	0	10	22
Others	299	0	14	4	21	40
Vegetables and starchy roots	*11,249*	*1342*	*607*	*420*	*1251*	*3620*
Potatoes	5895	830	549	199	618	2195
Tomatoes	908	8	7	38	108	160
Others	4447	505	51	183	525	1265
Fruits	*5636*	*117*	*43*	*235*	*669*	*1064*
Apples	1014	88	8	42	120	258
Oranges	1459	-	11	61	173	245
Others	3163	29	24	132	375	561
Alcoholic beverages	*6219*	-	*102*	*30*	*289*	*421*
Beer	4677	-	76	23	217	316
Others	1542	-	25	7	72	104
Meat and fish	*6437*	*203*	*254*	*237*	*585*	*1278*
Bovine meat	1096	60	47	44	112	263
Pig meat	1522	25	66	61	156	307
Poultry meat	2061	66	89	82	211	448
Fish	1194	9	30	31	56	127
Others	564	43	21	19	50	132
Dairy and eggs	*15,684*	*230*	*148*	*86*	*1028*	*1492*
Milk	14,710	201	143	71	958	1373
Eggs	733	29	3	14	55	100
Others	242	-	2	1	16	19
Others	*1554*	*65*	*82*	*22*	*200*	*368*
Total	**58,686**	**3622**	**2169**	**1226**	**6091**	**13,107**

Work is ongoing to reduce the amount of food that is wasted, and there are a number of relevant targets in place that mandate reductions in waste over the coming decade, including:

- The UK's commitment to the targets set out in UN Sustainable Development Goal 12.3 of a 50% decline in edible food waste per capita by 2030 against the 2007 baseline;
- The Welsh Government is aiming to meet these UN targets five years earlier, by 2025 (Welsh Government, 2019);
- The Scottish Government announced a mandatory target in 2018 to cut food waste by a third by 2025 against a 2013 baseline (Zero Waste Scotland, 2019);
- Many of the major retailers (representing 93% of the grocery sector; WRAP, 2020) in the UK are signatories to WRAP's voluntary Courtauld Agreement. Amongst the targets of the Courtauld Agreement are a 20% reduction in waste food and drink post farm-gate against a 2015 baseline by 2025.

Analysis in 2020 showed that the UK is on track to meet these targets (WRAP, 2020). Whilst continued improvement in the strength of the circular economy is an effective way to reduce energy consumption through reuse, recycling, and recovery, the most effective measures to control the N requirements of feeding a growing population are rooted in more robust preventative measures, as illustrated in the figure below (Figure 2.2.2).

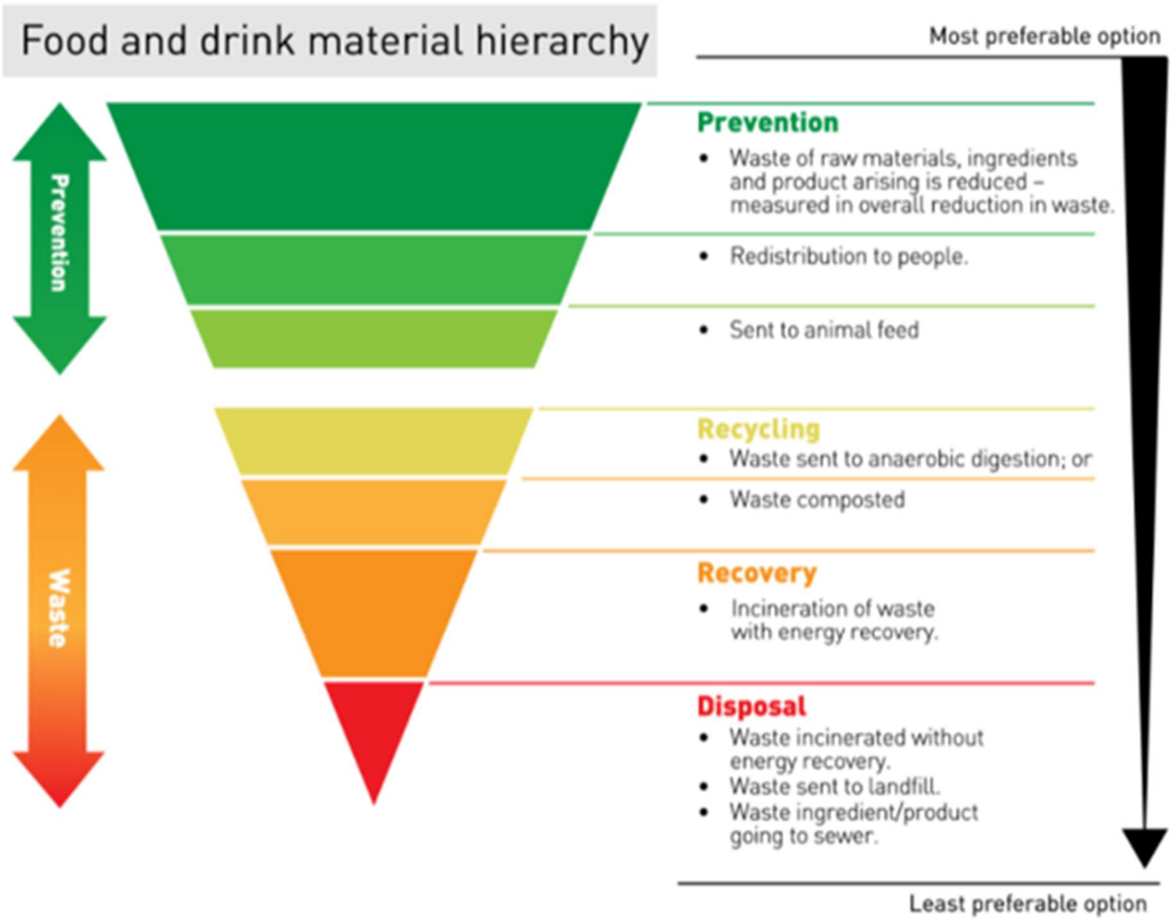

Figure 2.2.2 - Illustration of the options to reducing food waste as a part of a circular economy, ranked by preference (WRAP, 2020)

2.2.1 Primary production

Primary production food waste refers to all initial stages of the food supply chain, encompassing pre-harvest, growing, handling and storage operations. At the farm level, food waste can occur for a number of reasons, including harsh weather conditions, pest infestation, or spoilage. In the UK, additional factors, including excess production, fluctuating market prices, quality control standards, and food aesthetics can also be major reasons for waste at the primary production stage (Jeswani *et al*., 2021). Food aesthetic has been an emerging reason for food waste: it has been estimated that up to 31% of the UK's food is discarded as waste as it does not conform to typical aesthetic standards (Porter *et al*., 2018). Losses of these types stimulate excess food production on farms, as farmers seek to ensure they produce food of sufficient quality in quantities that remain economically viable. Broadly, these drivers can be separated into direct and indirect drivers, as Figure 2.2.3 illustrates. Direct drivers are those over which farmers are able to exert some degree of control, such as biological and environmental factors, technology, and infrastructure whilst indirect drivers are influenced by the wider supply chain.

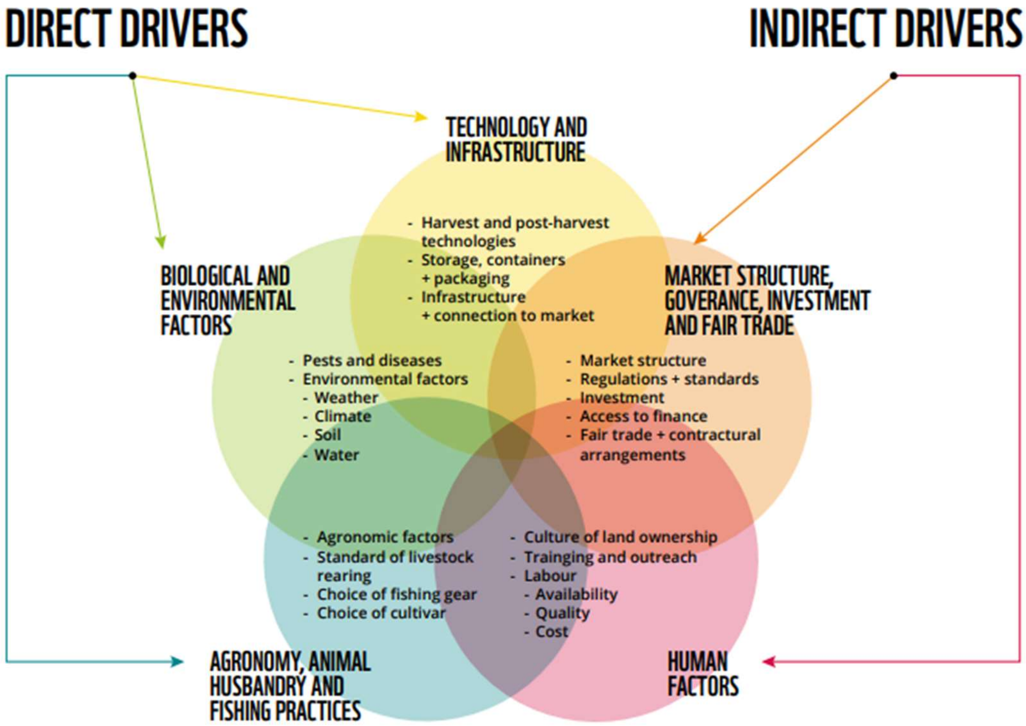

Figure 2.2.3 - Summary of direct and indirect factors driving food waste at the farm-level, illustrating the complexity of competing factors for approaching reductions on farms (WWF, 2021a)

Whilst the effective intervention on the indirect factors driving on-farm food waste would involve multiple elements rather than singular solutions, there are actions that can be taken to address the direct factors and improve the resource efficiency of food production at the farm level. It is common, for example, for unused agricultural products to be used for other purposes, such as processed into non-food products and use in bioenergy generation, but these do not reduce the N requirements of the food originally grown. In order to reduce the N requirements required to produce enough food for harvest, therefore, other actions are needed to reduce the amount of food that is wasted on-farm. In 2006, the BOC Foundation and Defra published a practical manual for farmers to minimise waste, (Defra, 2006) focusing on the crop production rather than livestock, with recommendations including:

- Consideration of crop production requirements with attention to security of the markets;
- Planning for inventory control and marketing strategy to avoid excess long-term spoilage;
- Monitoring the crop as harvest approaches as harvesting at non-optimum times can lead to poorer quality and more out-grade;
- Ensure machinery settings of harvester, and subsequent graders and cleaners are suitable for the crop to avoid crushed grain and sliced roots;
- Improvements in store design to ensure full ventilation and reduced inaccessible residues to prevent deterioration, as well as control of temperature and humidity;

- Use of food waste for other purposes that ensure the N remains in the chain, such as feeding spillages or other stock to livestock and utilising the N in manure spreading.

A number of other techniques have emerged more recently which include:

- **Increasing the longevity of livestock** (discussed in more depth in Section 2.3.3) which would reduce the waste of resources associated with animal rearing. Increasing the productive lifespan would require fewer animals to be reared for a given level of production and the prevention of early death of these animals to ensure that the effectiveness of nutrients invested is maximised.

- **Using alternative methods to identify pests and diseases earlier.** Some new advances have seen artificial intelligence and deep learning harnessed to identify a range of pests and diseases. Here, sensors attached to drones or fixed scanners are capable of processing images to identify whether insects pose a potential threat to the crop, or identify disease allowing farmers to isolate these areas before they are allowed to spread further. This has the potential to reduce the amount of crop lost during the production stages.

Recommendations published by the European Commission (EU Platform on Food Losses and Food Waste, 2019) highlight other, more holistic areas which would improve resource efficiency on-farm, which include:

- **Requiring national authorities to provide farmers with better access to data/information on market outlooks** so that they can better align their produce with market needs and avoid oversupply. This would effectively allow farmers to identify market trends more readily and plan their harvests accordingly.

- **Supporting farmers to improve animal health and welfare, and access to innovation**, including providing access to innovation and greater uptake of breeds and varieties and farming practices that can boost resilience and shelf-life, providing access to low-risk plant protection products, and improving access and affordability of new veterinary medicines. Strengthened animal welfare standards are viewed as a mechanism to reduce food losses (e.g. WWF, 2021a). Improved standards in relation to breed selection, animal rearing, slaughter, including the transportation between farm and slaughterhouse would reduce loss by poor living conditions and disease. These standards may take the form of Good Agricultural Practices (GAPs). Evidence shows that where the food industry has adopted GAPs, farmers have benefited from improved agronomy, access to technology and training (WWF, 2021a; Brown *et al.*, 2021)

- **National authorities, academia, and farmers associations should carry out further research on marketing standards**, in particular looking at the relationship between marketing standards and food waste and consider how resource efficiency could be maximised, for both economic and environmental reasons. The Directorate-General for Maritime Affairs and Fisheries (DG MARE) at the European Commission have recently published a study concerning the evaluation of marketing standards for fisheries and aquacultures, finding that there is no evidence that the standards have either increased or reduced food waste, but similar studies for the rest of the agriculture sector are not widely available. Whilst research behind marketing standards for aesthetic specifications illustrates that they can be safely relaxed, additional research is also needed relating to relaxation of food standard specifications which would further reduce food waste.

- In addition to this, national authorities should **strengthen the position of food producers in the supply chain** to protect primary producers from unfair practices such as short notice cancellations and unilateral contract changes, so that risk is more equitably shared between producers and the markets. The European Union implemented the unfair commercial practices directive in 2019 (Directive (EU) 2019/633), an update of a 2008 Directive transposed into UK law, but robust monitoring and enforcement regimes are required to ensure adherence to the law.

- **Strengthen financial support to farms to drive modernisation with a focus on tackling food losses and waste**. This overarching ambition includes supporting shorter, sustainable food chains, which are associated with lower rates of food waste, to maximise their contribution to the grocery sector. Other actions could include supporting markets for food and parts which are currently lost or wasted through the creation of new products, and by helping farmers improve their harvest, storage and logistic techniques and identifying innovative solutions and technologies within the sector. For example, plant-derived waxes can be applied to fruit and vegetables to improve the shelf-life of vegetables, such as through the company Apeel (https://www.apeel.com/). In addition, as discussed above, artificial intelligence is now able to more readily identify plant pests and pathogens, with emerging solutions using cameras and drones to help farmers identify issues earlier to minimise loss.

- More recent studies have included recommendations on **monitoring farm-level food loss and waste** (WWF, 2021a). A rigorous measuring and monitoring process can identify the points at which food is wasted and allow for a more targeted approach to reducing waste. Whilst costly to set up on-farm monitoring capabilities, it can improve the effectiveness of interventions.

2.2.2 Processing and manufacturing

A significant proportion of waste emerges during processing and manufacturing stages of the food supply chain. Food waste prevention is a key priority for food and drink manufacturers, and many companies make reducing waste part of their internal environmental management system and overall sustainability strategies, which requires close collaboration with other stages of the food supply chain, both upstream and downstream. A number of opportunities outlined in European Commission recommendation for manufacturers can help improve efficiencies at processing facilities, including:

- **Better planning/forecasting for raw material purchases** which could include the use of digital tools that will help the organisations balance supply and demand forecasts;

- **Manage, measure, and report on food loss and waste quantities** as this will allow the identification of particular hotspots. Websites such as Food Waste Atlas (https://thefoodwasteatlas.org/) allow governments and companies to report their food waste and can help improve understanding of how food loss and waste is occurring. This allows for more effective and targeted action.

- **Seek packaging solutions that can enhance food quality, freshness and safety**. Whilst a number of packaging solutions can allow for shelf-life extension, a balance needs to be struck with reducing packaging material. Innovative and interactive packaging solutions, such as temperature-sensitive sensors (e.g. shelf-life indicators), can help retailers and consumers understand the shelf-life of products and so prioritise use of food.
- **Increase sales of co-products and create more innovative products that utilise these co-products.** Innovative products may require more funding (public and private) dedicated to research, but can be an effective way of utilising surplus food and by-products. For example, spent grain from the brewing industry can be used as a food additive which is high in protein and fibre content, and can be used in a number of baked goods.
- **Improve date marking practices and consumer understanding of other relevant food information on packaging**. By facilitating correct and consistent implementation of the Food Information Regulations 2014 regarding 'use by' and 'best before' dates, as well as clear and meaningful 'open life' instructions to consumers. Effective product labelling formed part of the CCC's policy recommendations in its Policy Report for the Sixth Carbon Budget.

2.2.3 Distribution

The retail and distribution stages are often the least wasteful for food, but retailers and wholesalers play a pivotal role in influencing the reduction of food waste along the supply chain. Recommendations of the European Commission (EU Platform on Food Losses and Food Waste, 2019) largely centre on improving collaboration with other stages of the supply chain (namely primary production and manufacturing) to provide conditions that enable the reduction of food waste and actions that can improve consumer understanding by either providing better information of how to use the food.

- **Establish trustful relationships with suppliers and share data and forecasting information to match supply and demand.** Coordination through food waste prevention and joint business plans among supply chains can improve understanding of demand variability, improve forecasting, and minimise excess. These plans could include sharing risks of variable supply and demand with suppliers, especially for produce that is particularly sensitive to external factors, such as unpredictable climate conditions.
- **Use digital and automatic ordering system to improve shelf management practices**
- **Greater use of food repurposing**. Unsold fruits and vegetables can be repurposed using in-house processing capacity to process foods for other uses as they near their end of their shelf-life.
- **Make food waste a company priority** and ensure staff are engaged and trained on the importance of food waste, and set targets and key performance indicators for measuring food waste reduction. Train staff on frequently marking down products to support waste prevention and create a coherent marketing system that does not encourage bulk buying of the same foods, but rather offers discount deals across a range of foods.

- **Use consumer research to better understand causes of food waste at home and tailor products, discounts, and promotions to help consumers prevent food waste**. Actions here could include creating awareness campaigns during periods of typically high levels of food waste (e.g. Christmas) and provide information to consumers in-store about storage and recipe ideas for food.

2.2.4 Consumption

Food waste at the consumer level includes both domestic and service sector waste, and is often considered the largest contributing sector to overall food waste along the supply chain. Domestically, food is wasted for a number of reasons, including overbuying of produce, poor storage conditions, and a lack of recipe ideas meaning that excess food isn't used up before its 'use by' date. Many of the actions required to reduce domestic food waste are the responsibility of retailers and processing facilities, including providing more clarity of food labelling, improving packaging effectiveness to increase storage life of foods, and helping provide new recipe ideas to help use up leftovers. Other actions that can reduce food waste are well documented, such as creating meal plans and coordinating recipes to use up ingredients from other recipes within the meal plan to improve the overall effectiveness of the plan, prepare and store perishable goods soon after purchase, and understanding how ripening gases can affect the longevity of other foods such as the natural gases emitted from ripening bananas which can make other fruit spoil faster.

The food service sector is itself very wasteful, with some estimates indicating this sector contributes 12% to food waste across the EU (Fusions, 2016). As the sector is very heterogeneous and fragmented, the actions to reduce waste are varied and often site-specific. WRAP's recent roadmap to reducing food waste includes a specific roadmap for the hospitality and food service sector[3], and outlines methods to reduce food waste in order to meet the commitments of UN Sustainability Goal 12.3 and the Courtauld Agreement for those that have signed up. Amongst the actions for the service sector outlined in the roadmap are:

- Mapping out where food waste comes from, whether the kitchen, deliveries, leftovers etc;
- Using site measurement tools to help quantify waste. Guardians of Grub[4] offer tools to measure and monitor food waste as well as providing businesses with an understanding of how much food waste is costing their business;
- Creating a baseline against which improvements can be measured to track progress towards any internal targets, and ensuring that food wasted down the drain is quantified in these targets;
- Embedding awareness with staff inductions and introduce regular reviews of wasted food quantities and agree on targets within weekly meetings to encourage collective action;
- Promoting the use of portion size control options, and empower staff to offer doggy boxes for any leftovers so that they can be taken away by the consumer;

[3] https://wrap.org.uk/resources/guide/hospitality-and-food-service-sector-action-plan
[4] https://guardiansofgrub.com/

- Including actual food waste data clauses in Waste Management Contracts to encourage regular data reviews of waste and verification against data from kitchens.

The WRAP roadmap targets that by 2026, 100 large companies (>250 employees) commit to the WRAP's Target Measures Act, with all of those companies measuring, reporting, and taking action on reducing food waste across the majority of sites. In addition, it is hoped that these major companies would have whole supply chain waste management plans in order to reduce waste upstream as well.

2.2.5 Recycling residual food waste

Even after the full implementation of all actions to reduce food waste at the different stages of the supply chain, it is inevitable that some food will be continue to be wasted: inedible parts of food, for example, are not well targeted by the above actions and will continue to flow through the supply chain as waste. Historically, a large proportion of this waste has been sent to landfill where it decomposes, creating landfill gases rich in methane, a potent greenhouse gas. Whilst landfill gas capture techniques have improved over the past few decades, leading to emissions reductions of over 70% from the landfill sector since 1990 (Brown *et al.*, 2021), food left in landfill still represents a waste of N.

In some areas, food waste is redirected to anaerobic digestion facilities. At these sites, waste is heated and stirred for two to three months in a series of large sealed vats. This can produce substantial quantities of methane which is then captured and can then either be burned at power generation facilities or injected into the gas grid itself to mix with natural gas for use in homes and commercial buildings. This sector has been rapidly expanding: since 2000 there has been a 143% increase in the number of power stations generating electricity with biogas (Brown *et al.*, 2021).

Redirection of food waste for composting can provide an effective means to ensure that the N is not lost from the overall cycle. Some Local Authorities are now routinely collecting food waste separately from other waste. In Oxfordshire, for example, food waste is collected and handled using two different technologies: In-Vessel Composting (IVC) and anaerobic digestion. IVC is used to process food that is provided alongside garden waste, where it is shredded and treated in large tunnels and allowed to decompose and break down into compost. This compost, after undergoing maturation, is sold back to the farming sector and offers a valuable source of N for crop growth. Anaerobic digestion follows the same principle as described above for power generation, and the residual food waste from the process is pasteurised and stored before being sold to the farming sector as a valuable fertiliser.

2.2.6 Wastewater treatment plants

Wastewater treatment plants (WWTPs) in the UK deal with the nitrogenous waste excreted by humans. Currently, the flow of wasted N through wastewater treatment plants is far smaller than that contained in livestock manure, but if dietary shift toward more plant-based diets is achieved, then making better use of the N (and also phosphorus) will gain in importance for achieving a circular N cycle.

Some of the sludge left over from primary, secondary and tertiary wastewater treatment can be anaerobically digested, and some of the digestate spread on land (sometimes after a compositing step). However, the remaining N is either:

- Lost as Nr in effluent water leaving WWTPs;

- Volatilised as NH_3 or NOx during treatment;
- Lost through microbial denitrification to N_2 during treatment;
- Of the sewage sludge and digestate that is captured during treatment, not all can be spread on land due to contamination with heavy metals and other toxins from industrial effluent, and regulations limiting to which crops it can be applied. This may be incinerated or sent to landfill, where further emissions of NOx (from combustion) and other Nr can arise.

Corrado et al. (2020) analyse the flow of N through the food chain and waste sector across Europe, highlighting key hotspots of Nr release and N_2 loss from the system, as well as defining future scenarios of how the flows may change Figure 2.2.4 below.

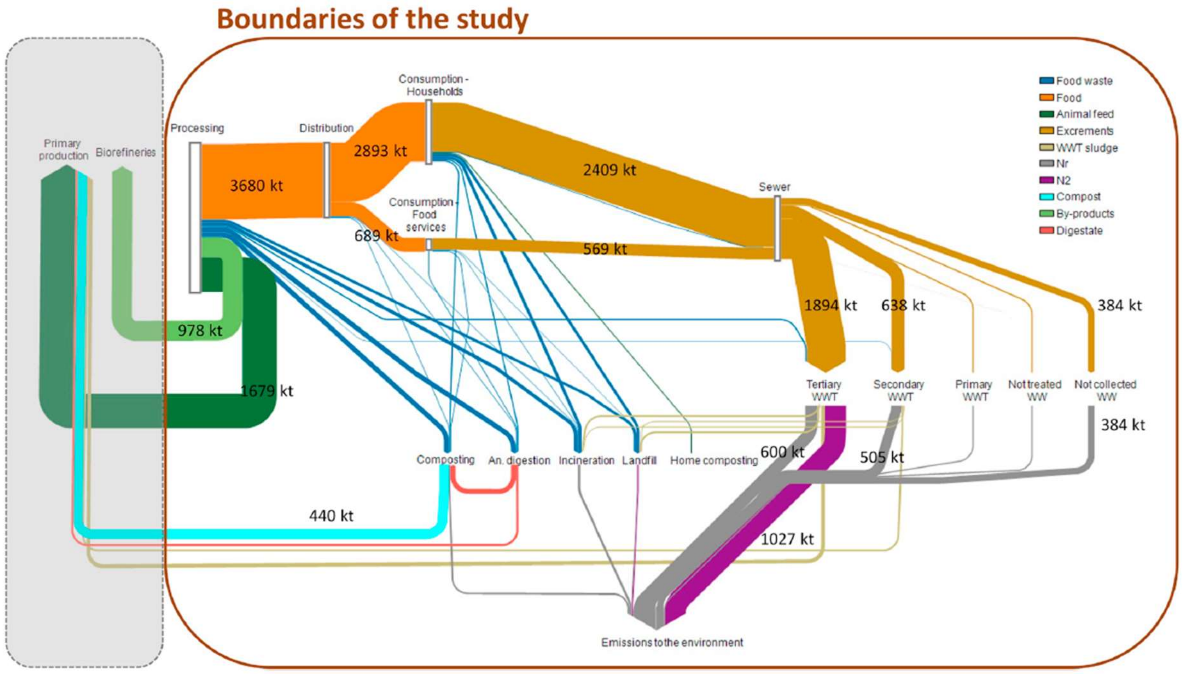

Figure 2.2.4 Flow of N through the food chain and waste sector across Europe, from Corrado et al. (2020).

They conclude that wastewater treatment, as well as untreated wastewater, are hotspots for Nr emissions post farm-gate. Interventions foreseen by the current EU legislation on wastewater - largely increased use of tertiary treatment – would reduce Nr emissions by almost 50%, but also increase N_2 emissions by 30%, thereby contributing to a linear rather than circular flow of N. This is because tertiary treatment allows a large fraction of Nr to be converted to N_2. Alternatively, if advanced N recovery (see below) were applied to 75% of wastewater, this would likely reduce both Nr and N_2 emissions by around 40%, as well as increasing the N recovered by over 30%.

In the UK in 2012 (Defra, 2012), around 80% of treated sewage sludge generated in WWTPs was spread on land (no more recent figures could be found), which represents a good current level of nutrient recycling. Around 75% of sewage sludge in 2012 was treated with anaerobic digestion, which eliminates pathogens and increases the fraction of N in the form of ammonium in digestate compared with raw sludge, which is easily taken up by plants.

However, N is still lost from the system during treatment, and in untreated waste. In the UK, the vast majority of the population is connected to public sewers, but as recently highlighted in the media, discharge of untreated effluent into watercourses from combined sewers (those carrying both sewage and rainwater runoff) during high-rainfall events occurs regularly[5].

In the UK, key interventions to reduce losses of Nr and increase the circularity of N from the wastewater treatment sector could be:

- **Reduce the proportion of untreated wastewater** released into the environment. Defra has outlined a plan[6] to introduce new legislation to reduce the frequency of storm overflow discharge events;
- **Increase the use of tertiary treatment to remove Nr from wastewater;**
- **Reduce contamination of sewage with heavy metals and other potentially toxic elements** (PTEs), to allow a larger proportion of sewage sludge to be applied to crops. This may allow an even greater proportion of sewage sludge to be spread on agricultural land;
- **Increase the use of advanced N recovery technologies in WWTPs**, such as ammonia stripping and struvite precipitation, to create usable mineral fertilisers from wastewater. This could help to recycle 40% more of the N in wastewater where applied, although technical and economic barriers may delay implementation of such techniques at scale (Corrado et al, 2020).

2.3 On-farm measures

2.3.1 Introduction

The previous sections describe the huge impact that demand-side measures could have on N flows. However, given a particular level of agricultural production, there are significant improvements that can be made at a farm and wider-economy scale to improve N use efficiency and reduce N waste, encouraging a more circular agricultural N cycle. In the current climate of predicted increases in global livestock production, these supply-side measures have been the focus of most research and national (and EU) policy formation.

The recently published UNECE guidance on integrated sustainable N management (UNECE, 2021) outlines 24 principles, which outline the key pathways through which N management can be improved, alongside key caveats and practical considerations. Here, we would like to highlight some of these principles we think are especially relevant:

- **Principle 5: Nitrogen input control measures influence all N loss pathways.**

This is a useful heuristic when assessing potential for co-benefits or pollution-swapping, as reduction in overall N flows are likely to reduce most or all forms of N loss and pollution. Measures that reduce overall N flows should also ultimately reduce global demand for synthetic fertilisers. If this leads to reduced levels of fertilizer production, then there is a

[5]https://deframedia.blog.gov.uk/2021/03/29/measures-to-reduce-harm-from-storm-overflows-to-be-made-law/

[6]https://www.gov.uk/government/news/measures-to-reduce-harm-from-storm-overflows-to-be-made-law

double-win, as reactive N and GHG emissions (N_2O, CH_4 and CO_2) associated with energy-intensive fertilizer manufacturing processes will also be reduced.

- **Principle 6: A measure to reduce one form of pollution leaves more N available in the farming system, so that more is available to meet crop and animal needs. In order to realize the benefit of a measure to reduce N loss (and to avoid pollution swapping), the N saved by the measure needs to be matched by either reduced N inputs, increased storage, or increased N in harvested outputs.**

This is a crucial point. If, for example, a livestock farmer reduces application losses of ammonia from slurry, this should be reflected in reduced application rates to the land. However, given that the quantity of N available to the livestock farmer has not been reduced, an alternative fate needs to be found for that N. This may also require increased transport of manure from locations of high manure production to locations of low manure production and high crop production.

Together, principles 5 and 6 illustrate the difference in impact of measures that affect the total N flows – the "activity data" in emissions inventory terminology – and those which affect the proportion of N lost from the system and in which forms – the "emission factor". In general, measures to reduce emission factors have more complex implications for N flows, with potential for pollutant swapping. However, the two types of measures can be connected: If the saved N can be recycled effectively, then it can lead to reduced overall N flows.

In addition to the principles listed in UNECE (2021), further ones could be added:

- **Manure management on farms consists of several linked stages in sequence, and measures to reduce emission factors upstream are ineffective if measures are not also applied downstream.** This message is also made clear in the UNECE guidance on ammonia mitigation (Bittman et al., 2014). The practical implication of this is that measures targeting manure application practices are particularly important, and the marginal abatement cost of upstream and downstream measures *should be considered as part of a package, rather than independently*.

- **There are interactions between measures, which change cost-benefit calculations**

If two measures are applied at the same stage of manure management (e.g. fitting a lid on a manure store and acidification), then the cost effectiveness of the marginal abatement achieved by the second measure is reduced considerably. In other cases, the measures may be mutually exclusive, or be applicable in completely different circumstances.

2.3.2 Sources of literature

The identification of key interventions below relies on a summary of recent literature:

- UNECE guidance on integrated sustainable N management (UNECE, 2021)
- UNECE-TFRN guidance on ammonia mitigation (Bittman et al., 2014)
- Rapid Evidence Assessment of interventions to improve air quality: agricultural/rural interventions (IOM, 2018)
- The "Nordic Nitrogen and Agriculture" report (Hellsten et al., 2017).
- SRUC non-CO_2 abatement in the agriculture sector to 2050 (Eory et al., 2015)

Evidence of current UK uptake of measures is sourced from the UK's Informative Inventory Report 2021 (Churchill et al., 2021), the underlying statistics from DEFRA's "British survey of fertilizer practice (BSFP, 2019) and "Farm practices survey for England" (Defra, 2020a and b) and Eory et al. (2015) where applicable.

Interventions span a range of different types, covering different parts of the agricultural N cycle and interventions with a different emphasis. To facilitate comparison with the analysis undertaken on measures with high GHG mitigation potential identified as part of the CCC 6th carbon budget, the same groupings are used here:

- **Livestock diets**
- **Livestock health**
- **Livestock breeding**
- **Waste management** (manure management measures)
- **Crops and soils**
- **Productivity improvements** (part of the "land release measures" discussed in the 6th carbon budget)

In addition, "system" measures such as mixed farming are considered separately, as they do not fall easily into any of the headings above.

Although "system" measures are considered below separately from more specific "incremental" measures to improve current systems, it is worth noting that the different approaches – whilst generally championed by different sets of stakeholders – are not always mutually exclusive. Many of the technical measures are still applicable to alternative systems, so ideology should not be a barrier to taking advantage of all relevant approaches - e.g., use of feed additives even for grazing ruminants to reduce GHG emissions. Moreover, technical measures to increase efficiency or substitute inputs may actually be the first steps on the road to a farm moving from a conventional intensive system to a more sustainable one (Lampkin et al 2015).

Below, the potential impact of key interventions is summarised. Where available in the literature, the potential emissions reduction (in %) is given. Note that this % reduction refers to the change in emissions where implemented compared to the emissions under a standard, unabated "reference" system, and relates only to the emissions from that particular stage of the livestock or crop management. For example, the 70-90% reduction from air scrubbing relates to a 70-90% reduction in NH_3 emitted from housing where it is applied, not from the entire livestock / agriculture sector.

2.3.3 Livestock diets, breeding and health

2.3.3.1 Key measures for reducing nitrogen waste

As discussed in Section 2.3.1, reducing the amount of N excreted by livestock in urine and faeces will reduce all types of N_r loss. Improving livestock diets, breeding and disease management and welfare can be effective tools to achieve this (UNECE, 2021). Effective measures for reducing Nr emissions through improved livestock diets, breeding and health are shown in Table 2.3.1.

Table 2.3.1. Effective measures for reducing N_r emissions through improved livestock diets, breeding and health. Downward arrows indicate a reduction in losses: ↓, small to medium effect; ↓↓, medium to large effect (UNECE, 2021)

Measure	NH_3	NO_3^-	N_2O	NO_x	Total N loss	CH_4	Biodiversity	Dependencies	Current uptake [3]
Low protein diets	↓↓	↓↓	↓↓	?	↓↓	↔/↑	↔ no direct impact		~20%
Breeding for improved performance[2]	↓	↓	↓	↓	↓	↓	↔ no direct impact	1	Dairy & Beef ~ 50% Sheep: ~ 40%
Increased livestock health and disease management[2]	↓	↓	↓	↓	↓	↓	↔ no direct impact	1	~75%
Extend the grazing season for cattle (also a "system" measure)	↓↓	↔ /↑	↔ /↑	↔ /↑	↔ /↓	↔ /↑	↓/↑ depends on local vs. global		N/A

Key:

1. *This measure will only reduce UK Nr emissions if improved efficiency of production does not lead to increased production;*
2. *Authors' own assessment, as this measure is not evaluated in UNECE (2021) in a general way;*
3. *Evidence of current UK uptake of measures is sourced from the UK's Informative Inventory Report 2021 (Churchill et al., 2021), the underlying statistics from DEFRA's "British survey of fertilizer practice (BSFP, 2019) and "Farm practice survey for England" (FPS, DEFRA, 2020) and Eory et al. (2015) where applicable.*

Low-protein diets

In the UK context, a key method of reducing emissions of all N_r compounds is to tailor livestock feed rations to match protein consumption with their demand, be that for growth or production of milk, eggs or wool. Excess protein is excreted in urine and in faeces, so by limiting intake this limits N excretion. A key factor for ruminants is the digestibility and energy/protein ratio of the feed. If productivity (growth rate, milk production etc.) is limited by poor digestibility or insufficient quantity of carbohydrates, then the protein present in the feed cannot be utilised fully so will be excreted. Therefore, **high-digestibility but low-protein rations encourage high N use efficiency and low N excretion**. Food such as fresh grass/clover can have protein contents of 20% or more (by dry matter weight), whereas maize silage for example has a protein content of around 8%. Where feeding can be closely controlled (e.g., for pigs), one option is to provide very low-protein feed alongside amino acid supplements to precisely tailor N intake (UNECE, 2021). Another related measure involves increasing the proportion of non-starch polysaccharides (i.e., fibre) in the diet of pigs, which increases bacterial growth in the large intestine and locks more N in bacterial protein rather than as TAN (IOM, 2018).

The impact of low-protein diets varies by livestock type and situation, but as a rule of thumb N excretion can be reduced by 5-15% for every 1% reduction in crude protein content.

The potential for improvement in the UK is likely to be highest for cattle and sheep (IOM, 2018) where a large proportion of time is spent grazing high-protein grasses and other forage. According to the Farm Practices Survey (Defra, 2020), 66% of holdings are taking action to reduce GHG emissions, of which 27% are improving N feed efficiency (45% of large holdings). The true uptake may be greater than this, as some holdings may already be feeding low-protein diets so not making improvements.

Increased animal and herd-level productivity, including through better health and disease management

Increases in productivity per animal tend to increase N use efficiency (NUE), as a smaller fraction of protein intake is used for maintenance or non-productive growth. For animals reared for meat a key factor is rapid growth, whereas for dairy cows and laying hens it is the combination of productivity per cycle and productive lifetime that matters. At the individual animal level, such productivity increases can be achieved through breeding, optimal feeding strategies and health and welfare management. At the herd/system level, reduced mortality increases overall productivity, and in cattle system-level improvements can be achieved through productive use of male calves (e.g. for rosé veal), or alternatively use of sexed semen to reduce the frequency of male calves. It is important to note that the relationship between increased productivity and increased NUE is not fixed, and depends on how increased productivity is achieved. If productivity is increased through high-protein feeding strategies, then this could even reduce NUE (see above). Another important caveat is that a "rebound effect" - whereby increases in efficiency lead to producing more with the same, rather than the same with less – must be avoided, or reductions in Nr emissions at the system level will not occur. Eory et al. (2020) estimate that better disease control could increase productivity of cattle and sheep by around 6.4% and 10.5% respectively.

Understanding the additional potential for improvement in livestock health and productivity is not straightforward. According to the FPS (Defra, 2020) the proportion of holdings using high quality bulls and rams at least some of the time for breeding is 50% for cattle, and 40% for sheep. Around 75% of holdings currently have a farm health plan.

Extended grazing time

The majority of NH_3 lost through volatilisation shortly after excretion in animal housing comes from urine. **When deposited on pasture, urine quickly infiltrates the soil, reducing the potential for NH_3 volatilisation.** Extending the grazing season increases the proportion of urine and dung which is returned to pasture (or an arable rotation where a ley or cover crop is grazed). The efficacy depends on the degree of grazing time extension possible. Bittman et al. (2014) cite a 50% reduction in NH_3 emissions possible for 22/24h grazing relative to a zero grazing reference system. Around 14% of TAN in urine and dung is volatilised following deposition on pastures, compared with generally over 50% from housing, storage and application from manure deposited in housing (with no abatement) (EEA, 2019).

There is potential for some of this reduction in NH_3 emissions to be offset by an increase in N loss as nitrate leaching, or N_2O, N_2 and NOx emissions where urine and dung deposition is concentrated, and exceeds the ability of plants to take up the additional N making it more available for microbes. The risk of this is especially high from late summer through to winter, when swards are not growing quickly. Grazing in waterlogged conditions can cause "poaching", resulting in localised higher N_2O emissions and soil compaction. These impacts can be mitigated, however (see Section 2.3.6).

Another factor to consider is the reduced control over cattle protein intake whilst grazing. Protein content of fresh grassland swards can vary considerably, and at some times of year can greatly exceed intake requirements, leading to higher N excretion than from cattle fed a controlled diet. Nevertheless, as much of the extra N excreted is quickly immobilised in the soil and spatially diffuse, this is unlikely to cancel out the benefits of reduced ammonia emissions.

The potential for extending grazing periods of cattle in the UK is difficult to quantify. Grazing periods vary by cattle type and location, governed by climate, soil conditions and other factors. Dairy cows are on average housed for around 280 days per year, versus around 190 for most beef cattle and 130-150 for dairy replacements, heifers and calves (Churchill et al., 2021). Some grain-fed beef cattle are kept in housing 100% of the time. For dairy cows, there has been a gradual reduction in average grazing time, from around 152 days in 1990 to 82 in 2019. This indicates there may be some potential to reverse that trend.

2.3.3.2 Key measures for reducing GHG emissions

Table 2.3.2 outlines measures included in the scenarios related to livestock diets, breeding and health from the 6th carbon budget.

Table 2.3.2. Low Carbon GHG mitigation measures included in the 6th Carbon Budget agriculture mitigation scenarios related to livestock diets, health and breeding (CCC, 2020a); [1]According to CCC (2020a)

Measure	Balanced Pathway abatement 2035 ($MtCO_2e$)	Widespread Engagement abatement 2035 ($MtCO_2e$)
Livestock diets (36% of mitigation in 2035)[1]		
Higher sugar content grasses (MM21)	49.6	49.3
High starch diet for dairy cows (MM31)	6	6.3
Ruminant feed additive: 3NOP (MM35)	956.4	883.4
Ruminant feed additive: nitrate (MM44)	416.5	373.5
Precision feeding (MM32)	22.7	13.4
Livestock health (15% of mitigation in 2035)[1]		
Health Cattle (MM30)	410.0	479.4
Health Sheep (MM48)	216.0	254.6
Livestock breeding (8% of mitigation in 2035)[1]		
Breeding Genomics (MM26)	224.8	200.6
Current breeding (MM29)	75.8	67.7
Low methane (MM27)	21.8	19.7
GM Cattle (MM28)	? (26.6 by 2050)	?
Other Livestock Measures		
IncreaseMilkFreq (MM37)	60.8	53.4

Regarding livestock diets, higher sugar content grasses, high starch diets for dairy cows, and precision feeding all reduce the GHG intensity of production by increasing the digestibility of the diet, which in turn reduces enteric fermentation emissions and volatile solids excretion (the source of methane from manure) per unit of production. Feed additives 3NOP (3-nitrooxypropanol - a chemical that reduces the production of enteric methane by ruminants when added to their rations) and nitrates directly inhibit enteric methane production in the rumen. During the fermentation process hydrogen is generated, and, via a microbial process, it reacts with CO_2 in the rumen, forming CH_4. The rumen processes can be modified, for example with chemical compounds which serve as an alternative hydrogen sink (Hristov et al., 2013). The measure can be implemented by mixing 1.5% nitrate homogeneously into ruminant diets. The nitrate would (partially) replace non-protein nitrogen (NPN) sources (e.g., urea), or, if NPN is not present in the diet, then high protein content components, like soya. Of the dietary measures, 3NOP addition has the highest mitigation potential (Table 2.3.2).

Measures to improve livestock health and breeding measures reduce GHG emissions through a similar mechanism to those reducing N_r emissions; higher productivity tends to be associated with a higher feed conversion ratio, and lower rates of disease reduce losses through mortality and reduced production. However, the same caveat applies to GHG emissions as to NUE, that this relationship is not fixed and depends on how increased productivity is achieved. Lifecycle GHG emissions could potentially be increased if productivity increases occur through e.g., greater proportion of soy in the diet, with associated land-use change related GHG footprint.

Breeding for low methane emissions is somewhat different, in that low enteric methane emissions per se would be a breeding goal, rather than increased productivity. Of the health and breeding measures, improving the health of cattle and sheep, and making use of genomics in livestock breeding would have the highest mitigation potential (Table 2.3.2).

Increased milking frequency also increases milk yield (Table 2.3.2; CCC, 2020a) and at the same time improves the amino acid and N utilisation of the animal, reducing its N excretion (Moorby et al., 2007). The reduced N excretion reduces both direct and indirect N_2O emissions from manure management. The measure can be implemented by the use of robotic milking parlours. This entails purchase of a robotic milker (typically costing £50-80k per 60 cows) and changes to stock management (e.g. keeping cattle closer to the milking parlour). Milk yields are assumed to increase by 10%, which can partly offset the infrastructure costs (robotic milk parlour).

2.3.3.3 Trade-offs and synergies between measures to reduce N waste and GHG emissions (and other impacts)

As indicated above, increased animal and herd-level productivity through better health, breeding and diet is a key theme of GHG mitigation from livestock. For breeding and health-related (non-dietary) measures there are fundamental synergies achieved through a high feed conversion ratio, which tends to reduce enteric fermentation emissions, volatile solid and N excretion. Measures that reduce N excretion, such as breeding, animal diet and health and increased milking frequency, can have 'major' co-benefits for air quality and water quality (CCC, 2020a).

The situation for livestock diets is nuanced, however, with the potential for some trade-offs in certain situations. High digestibility and low fibre content ruminant diets result in low enteric methane production, and lower volatile solid excretion (the basis of manure methane emissions). Where high digestibility is paired with a low crude protein content (as in the case of maize silage, for example) suggested under "MM31 high starch diet for dairy cows" (Table 2.3.2), then this will also favour reduced N excretion. "MM21 High sugar content grasses" would also allow synergies, all else being equal, as this has been shown to increase N utilisation. However, from a N waste perspective, the additional N inputs to growing these feed crops must be considered, compared with making use of higher-fibre fodder from extensive grassland, for example. Low-starch pig diets aimed at reducing the TAN content of manure do have the potential to increase methane emissions, from both enteric fermentation and manure decomposition (IOM, 2018).

Extending the grazing season for cattle has potential impacts on a variety of other outcomes in addition to Nr emissions. The impact on GHG emissions is uncertain. CH_4 emissions from manure deposited on pastures are lower than for stored manure (especially liquid slurry), due to largely aerobic conditions. However, higher enteric methane emissions can occur from low-digestibility forage, as well as limiting the opportunity to provide supplementary feed additives to reduce enteric methane (depending on the diurnal distribution of grazing). Additional grazing on high-sugar grasses rather than rough pasture (see section 2.3.3.2) would reduce the potential trade-off with higher enteric methane emissions, but on the other hand may require higher N inputs and have lower biodiversity value than extensive grazing. As previously mentioned, there is also potential for higher N_2O emissions in some cases. Finally, soil carbon sequestration can be increased by grazing, if additional extensive permanent pasture is created from arable land or intensive pasture to accommodate the increased grazing time. The balance of overall emissions and sequestration may be context dependent (Garnett et al., 2017).

Other synergies include:

- Reduced costs associated with bedding, manure storage and application. If extended grazing season is associated with a shift to lower stocking densities and lower inputs, there is the potential to improve profitability of production by reducing variable costs of inputs (FFCC, 2019);
- Potential for a positive impact on local biodiversity if additional extensive permanent pasture is created from arable land or intensive pasture;
- Increased activity and forage choice is likely to be beneficial for animal welfare, with a knock-on positive effect on health and productivity.

Another potential trade-off of extending the grazing season is that if using low-intensity pastures requires more land for the same level of production than a more intensive system (note the if), then globally this has consequences as there is an opportunity cost to using this land for grazing rather than alternative uses, and risks exporting impacts (e.g. biodiversity loss) to other regions (see Section 2.5 for further discussion of this theme).

There is no evidence for trade-offs between addition of enteric methane inhibitors (3NOP and nitrate) and N_r emissions. However, one co-benefit of adding nitrate to the diet is that it can replace some of the non-protein nitrogen (if present), or high-protein feeds such as soy meal in the diet. This would in turn reduce the amount of such feedstuffs imported, and in turn the associated N footprint.

There are no major trade-offs of the livestock breeding, diet and health measures for other environmental outcomes, but some considerations to bear in mind are:

- The benefit of increased productivity or efficiency per animal in reducing N_r emissions assumes that production levels remain constant. If production increased in response to improved efficiency (Jevon's paradox), then this would negate the benefits.
- From a whole system point of view there are trade-offs between precise control of diets and adoption of agro-ecological extensive (unfertilized) livestock grazing systems, which has implications for land use choices and associated biodiversity and ecosystem services.

A likely co-benefit of measures to improve livestock health, disease management and longevity would be higher animal welfare (which may in fact be part of the measures undertaken).

2.3.4 Waste management

2.3.4.1 Key measures for reducing N waste

Emissions of N_r from livestock housing and manure storage can comprise over 50% of the N excreted in urine and dung in animal housing (IPCC, 2019). Below, key measures to tackle emissions from livestock housing and manure storage are presented (Table 2.3.3). Note that reducing time in housing and extending the grazing season is another approach to reducing emissions from housing and manure storage, described further in Section 2.3.7.

As introduced above, a key dependency common to most of the measures below (exceptions being air scrubbing and slurry acidification) is that the reduction in emissions of Nr leaves a higher N content remaining in the manure, so abatement must also be applied to subsequent stages (storage, application) to avoid these gains being partially offset by increased emissions downstream. This also has the knock-on impact of requiring an adjustment to application rates, and potentially an alternative fate to be found for surplus N and P in manure that cannot be spread on a farmer's own land.

Table 2.3.3. Effective measures for reducing N_r emissions from livestock waste management. Downward arrows indicate a reduction in losses: ↓, small to medium effect; ↓↓, medium to large effect (UNECE, 2021)

Measure	**NH_3**	**NO_3^-**	**N_2O**	**NO_x**	**Total N loss**	**CH_4**	**Biodiversity**	**Dependencies**	**Current uptake / potential[6]**
Low emission housing (group of measures, excl. Air scrubbing)	↓/↓↓	↔	↔	↔?	↓	↔	↔ no direct impact	1	33% for pigs, 75% for broilers
Air scrubbing (acid and biological	↓↓	↔	↓	↓/?	↓↓	↔	↔ no direct impact		?
Increasing storage capacity[5]	↓↓	↓↓	↓↓	↓↓	↓↓	↔	↔ no direct impact		
Covering slurry stores with impermeable cover and base	↓↓	↓↓	↔	?	↓↓	↔/↓?	↔ no direct impact	1	Tanks 24%; lagoons c. 5%
Mechanical separation of stored slurry into liquid and solid fractions	↓↓	?[4]	↓	?[4]	↓	↔	↔ no direct impact	3	~10%
Slurry acidification in housing / during storage	↓↓	↔?	↓	↔/↓?	↓↓	↓↓	↔ no direct impact	2	0% (exp. Judg.)
Anaerobic digestion of slurry and solid manure	↓↓	↓↓	↓	?[4]	↓↓	↓↓	↔ no direct impact	1	<10% of manure
Ammonia stripping and recovery from manure	↓	↓	↓	↓	↓↓	↔	↔ no direct impact		?

Key:

1. *This measure will only reduce overall N losses and Nr pollution if measures are also put in place downstream in the nitrogen flow – e.g. for application to soils;*

2. *This measure is most effective if the quantity of manure applied to land is adjusted downwards to reflect lower N losses during storage;*
3. *Efficacy depends on appropriate storage of the liquid and solid fraction to prevent increased CH_4, NH_3 and N_2O emissions during storage;*
4. *Although this measure does not directly reduce NOx or NO_3 losses, it contributes to circularity of nutrient use and therefore a reduction in emissions from synthetic fertilisers;*
5. *Slurry-Max Project (Waterton et al., 2018);*
6. *Evidence of current UK uptake of measures is sourced from the UK's Informative Inventory Report 2021 [IIR], the underlying statistics from DEFRA's "British survey of fertilizer practice (BSFP, 2019) and "Farm practice survey for England" (FPS, Defra, 2020) and Eory et al. (2015) where applicable.*

Housing measures

Losses from housing primarily occur through volatilization of ammoniacal N (TAN – aqueous ammonia and ammonium ions), which is quickly produced following excretion from urea and other simple nitrogenous compounds. The volatilisation process is a physico-chemical one, so is governed by factors such as pH, temperature, airflow, time exposed to air and surface-area-to-volume ratio of manure (EEA, 2019).

Housing design

The majority of housing measures seek to reduce ammonia volatilization by controlling one or more of the key factors listed above, or by immobilizing ammoniacal N. Effective measures include (from Bittman et al., 2014 and UNECE, 2021):

- frequent manure removal to external storage, aided by use of grooved or slatted floors (for cattle and pigs) and belts (for poultry), minimizes the time for which urine and dung is exposed to the air on floors with a high surface area;
- Use of V-shaped gutters to minimise surface area;
- Use of straw to immobilize TAN;
- Cooling of manure channels in pig housing;
- Barn climatization to reduce air temperature and flow;
- Poultry litter drying;
- Immediate segregation of urine and faeces (cattle), to reduce hydrolysis of urea to ammonia through urease present in faeces;
- Acidification of manure.

The efficacy of these measures varies between 20 and 70% reduction in NH_3 emissions depending on the methods employed. The choice of measures to deploy will vary by livestock species and other considerations, such as whether the measure is being applied as a retrofit or to new housing, and the type of external storage available and its capacity (which influences whether flushing with water is feasible). Use of straw – while effective in immobilising TAN – does have disadvantages later in manure management, as this leads to manure being handled as a solid, which in turn is less amenable than slurry to low-emission application techniques on a variety of crops, due to the need for rapid incorporation through tillage.

No figures could be found on current levels of penetration of low-emission cattle housing in the UK, but it is likely that there is considerable potential for improvement. In the UK Informative Inventory Report (IIR; Churchill, et al, 2021), it was assumed that 33% of pig housing had slatted floors, and 75% of broiler housing used litter drying. The UK Clean Air Strategy (Defra, 2019) proposes mandatory design standards for new intensive poultry, pig and beef livestock housing and for dairy housing, to reduce ammonia emissions, as well as extension of permitting for larger intensive beef and dairy farms by 2025.

Air scrubbing

Acid or biological air scrubbers are an alternative means of reducing N_r emissions to the environment. These do not prevent loss of N_r from manure, but strip ammonia from waste air streams leaving housing. Air scrubbing is only suitable for housing with forced air ventilation, which is more common for pigs and poultry and uncommon for cattle. 70-90% removal of ammonia from air outflow can be achieved, and the stripped ammonium salts or other byproducts can be recycled as mineral fertiliser. One advantage of air scrubbing over measures which retain N in manure, is that they do not depend on additional downstream mitigation to prevent N_r emissions later in the manure management chain.

No figures could be found on current levels of penetration of air scrubbing in the UK. The main barrier is cost, as this is a relatively expensive measure, up to EUR 17 per kg NH_3 removed (Hellsten et al., 2017). Because of this, implementation may be limited to new or expanding buildings, and retrofitting may be challenging.

Measures to reduce emissions from storage

Losses from storage occur through continued volatilisation and leaching of ammonia, but in addition where some oxygen is present nitrification and denitrification through microbial metabolism lead to emissions of N_2, N_2O and NOx. These microbial processes also depend in part on physical parameters such as temperature, airflow, moisture and pH, but also on manure composition (dry matter content of slurry, for example), which affect the suitability of manure as a substrate for microbes and speed of growth.

Covering manure/slurry stores with an impermeable cover and base reduces NH_3 emissions by decreasing the rate of volatilisation from the surface, by 60-80% compared with uncovered slurry lagoons or manure heaps (Bittman et al., 2014). It can also reduce emissions from solid manure heaps, though evidence is less available for this. Impermeable bases prevent NO_3^- leaching. There are mixed results for the impact on N_2O emissions – some studies show reduction, others increase (IPCC, 2019; UNECE, 2021). Covers can be solid concrete, plastic or metal lids, plastic sheeting, or alternatively a slurry bag can be used. Natural crust covers often form on slurry lagoons in the UK during summer. These also help to reduce NH_3 and CH_4 emissions as oxidation occurs in the liquid-solid interface, but they also notably increase N_2O emissions through enhanced nitrification (IPPC 2019 Refinement, Vol 4 Chapter 10, Table 10.21), and also do not prevent rain from diluting slurry. Therefore, impermeable covers are generally preferable.

A key co-benefit of fitting an impermeable cover to slurry stores is that it increases storage capacity (in terms of duration) and increases the nutrient concentration of slurry by preventing dilution with rainwater. The storage capacity for manure is a key determining factor required to allow appropriate timing and application rates of manure to soils (Waterton et al., 2018). It is recommended in the UK that all holdings have at least 6 months storage capacity, but currently 19% of holdings have only 1-3 months storage capacity (FPS, Defra, 2020). In addition, upgrading from open slurry lagoons to tanks and stores with a solid cover reduces the real or perceived risk of serious injury or death to farmers and animals from falling into lagoons (Waterton et al., 2018).

Slurry acidification reduces NH_3 emissions from slurry in housing, storage or application. Ammoniacal nitrogen (TAN) in slurry is in an equilibrium between aqueous dissolved NH_3 (which is easily lost as a gas), and the more stable ammonium ion NH_4^+. Low pH causes more of the TAN to be in the stable NH_4^+ form, thereby reducing volatilisation. Acidification can be achieved either through addition of concentrated acids to manure in housing, storage or before application, or through feed supplements such as benzoic acid (Bittman et al., 2014). The mitigation achieved depends on the resulting pH of the manure, but is around 50% reduction in NH_3 emissions at pH6 (Bittman et al., 2014), and up to 90% at pH5.5. Reduced rates of nitrification and denitrification can also be expected to reduce loss of N as N_2O, NOx and N_2 emissions (UNECE, 2021). An advantage of slurry acidification is that it also reduces emissions from manure application, so is not reliant on combination with low-emission spreading equipment.

Precise figures on uptake of manure covering and slurry acidification in the UK to date could not be found on a per unit N basis. The Survey of Farming Practices (FPS, 2020) includes estimates for number of holdings with covered stores (Table 2.3.4).

Table 2.3.4. Proportion of manure stores with a cover, by type of storage (FPS, Defra, 2020)

Type of storage	Proportion with cover
Solid manure stored in heaps on solid base	16%
Solid manure stored in temporary heaps in fields	1%
Slurry stored in a tank	24%
Slurry stored in a lagoon without strainer	3%
Slurry stored with strainer facility	8%
Slurry in another type of store	5%

The highest rates of covered storage relate to solid manure stored in heaps on a solid base, and slurry stored in tanks, which are present on 55% and 25% of holdings respectively. Only around 16% of holdings have plans to improve or rebuild manure and slurry storage facilities (FPS, 2020). It is assumed that slurry acidification in housing and storage does not currently occur in the UK. In Denmark in 2019, around 1.5% of manure was acidified in housing, 1.9% in storage and 7.5% just prior to application (Neilsen et al., 2021).

There are several barriers to increasing storage capacity and adding impermeable covers in the UK. The biggest barrier is cost; Waterton et al. (2018) identified that the cost of building extra storage capacity is a key barrier for farmers. Even where grant funding is available, it is not always applicable; for example, it can be difficult to achieve a covered surface with large lagoons and some kinds of storage infrastructure are unsuitable for standard retrofit options (Waterton et al., 2018) which would enable them to access grant funding for upgrades. It is more expensive to retrofit covers, so more cost effective to mandate for new stores (Hellsten et al, 2017). There are many components to the manure management systems in place on farms, and these are often inherited, so there is an element of technological lock-in whereby to make even small changes the whole system may need to be changed.

Nevertheless, the UK Clean Air Strategy (Defra, 2019) states a target date of 2027 for all slurry and digestate stores to be covered, so a strategy to address these barriers is required. For acidification, the key barriers to uptake are potential safety concerns of handling concentrated acids, limitation on use of acidified slurry in anaerobic digesters, and the fact that use of strong acids is not allowed under organic regulations.

Manure processing measures

Some effective manure management measures are not aimed at decreasing emissions from storage *per se* (though they may have that side-benefit), but are classed as manure treatment options which facilitate the recycling of nutrients in manure back into arable or grassland systems by increasing its utility as a fertilizer. Figure 2.3.1 below illustrates simple biological, chemical and physical processing techniques.

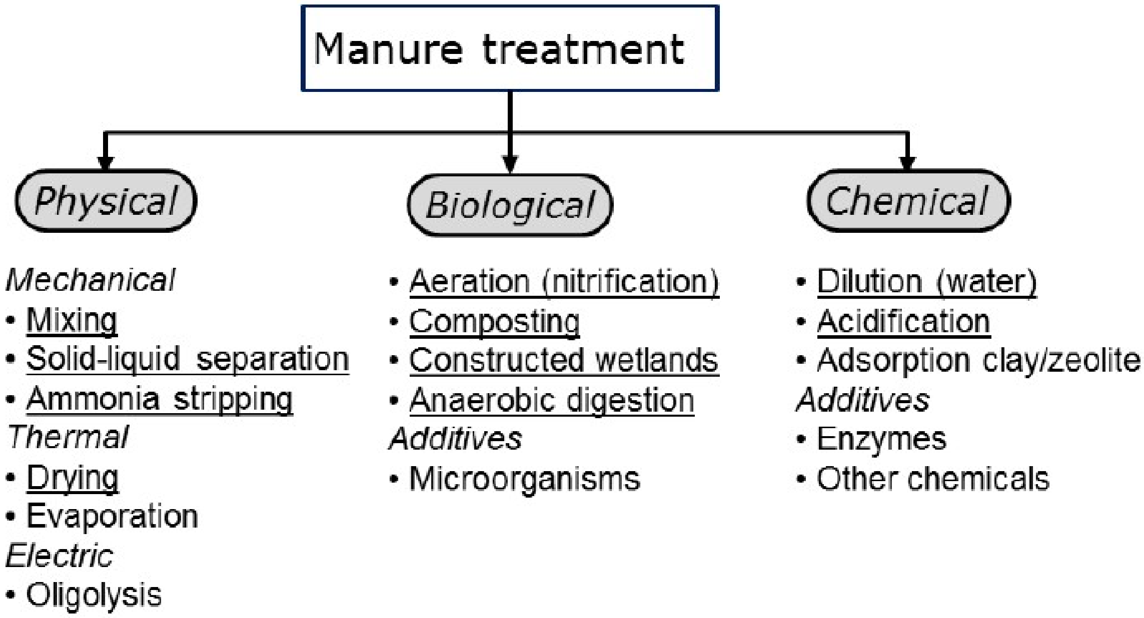

Figure 2.3.1. Simple manure processing techniques. Source: UNECE (2021)

A detailed discussion of the benefits and principles of manure processing can be found in UNECE (2021), pages 99-104, so will not be repeated here. Three key manure processing options are presented below; anaerobic digestion, solid-liquid separation and ammonia stripping.

Anaerobic digestion (AD) is currently implemented as a bioenergy production measure. Anaerobic decomposition produces methane, but also converts organically-bound N into ammoniacal N (nitrate formation is not favoured by the anaerobic conditions) in digestate. This digestate can be a valuable product for farmers and NH_3 loss from manure storage is reduced, but evidence for size of the effect is mixed (IOM, 2018). This is a category 1 intervention due to the co-benefits and system-wide impacts to increase circularity. The digestate is rich in TAN and low in dry matter, which aids handling, increases infiltration rate into soil (and thus lowers NH_3 emissions), provides N in a more plant-available form, and can be combined well with nutrient recovery techniques to produce mineral fertilisers (see below).

Mechanical solid-liquid separation of slurry results in a liquid fraction which has low dry matter content but rich in TAN, and a solid fraction with high dry matter content, rich in phosphorus. The main NH_3 and N_2O emission reduction benefits result from the liquid fraction having excellent properties as a fertiliser due to low dry matter content – rapid infiltration into soil, can be applied to growing crops without fouling, low potential for microbial growth (and associated N_2O emissions), and rapid availability of the N delivered in mineralized (rather than organically-bound) N to crops. This makes the liquid fraction a useful replacement for synthetic N fertilizer, increasing circularity of nutrient use. The solid fraction can also be applied to land, but another key advantage of separation is to reduce the mass of slurry by removing the water, which could improve the economics of transporting manure and thus improving circularity. The solid fraction can also be used as an excellent feedstock for anaerobic digestion (see above). Segregation of N and P which are enriched in the liquid and solid fractions respectively is also a useful outcome for controlling the balance of different nutrients applied to land. Mechanical separation can also be applied to AD digestate.

To be effective in reducing emissions from storage, digestate from anaerobic digestion and the separated fractions from mechanical separation must be stored in low-emission containers to avoid high NH_3 emissions from AD digestate and the liquid fraction (due to high TAN concentration and high pH of digestate), and high CH_4 emissions from the solid fraction. The solid fraction must be spread using low-emission techniques (UNECE, 2021).

Mechanical solid-liquid separation can be combined with **ammonia stripping,** whereby the liquid fraction is exposed to air (with a large surface area) to allow ammonia to evaporate, then collecting it using an acid scrubber. This produces ammonium salts which can then be used as a mineral fertiliser. As with application to land, N_r emissions mitigation is achieved by effectively increasing the re-use of N from manure, thereby reducing overall N flowing through the system. Ammonia stripping creates a concentrated product which can be economically transported. Other kinds of nutrient stripping and manure processing to produce more valuable, transportable and useful fertilizer products from manure also exist, which can be combined with the processes described above:

- Drying and pelletisation of AD digestate or solid fraction to create a P-rich soil conditioner;
- Struvite (magnesium ammonium phosphate) precipitation from AD digestate;
- Ammonium salt concentration from the liquid fraction (using ultrafiltration, evaporation or reverse osmosis).

Current uptake in the UK of these processing techniques is relatively low.

- According to the FPS (Defra, 2020), 11% of pig and poultry farms process waste using anaerobic digestion, but no information is available for cattle manure, but it is assumed to be lower than for pigs;
- 9% of all livestock holdings, but 14% of larger holdings, have a slurry separator;
- No estimates were found for current use of ammonia stripping in the UK, but it is assumed to be negligible.

Potential barriers to uptake of AD is mainly the cost, and organizational, technical and business skills required. Economies of scale mean larger centralized reactors are more economical than smaller ones on farms (IOM, 2018), but these need capital to set up so require sufficient incentives for investors. The subsidy structure has a considerable effect on the profitability of the plant. Limitations on transport distances of slurry also mean that centralized reactors require a high concentration of slurry sources nearby, and the transport increases costs for farmers. These may be ameliorated by prior manure processing (solid-liquid separation) however, to reduce mass.

For solid-liquid separation, the key barrier is the costs of storage and application equipment required for the solid fraction to avoid high downstream emissions.

For ammonia stripping, the main barriers are the cost of the system itself, and challenges in selling scrubber liquid on to fertiliser manufacturers (UNECE, 2021).

2.3.4.2 Key measures for reducing GHG emissions

Table 2.3.5 outlines measures included in the scenarios related to livestock housing and waste management in the 6th carbon budget.

Table 2.3.5. GHG mitigation measures included in the 6th Carbon Budget agriculture mitigation scenarios related to waste management (CCC, 2020a)

Measure	Balanced Pathway abatement 2035 ($MtCO_2e$)[1]	Widespread engagement abatement 2035 ($MtCO_2e$)[1]
Anaerobic digestion of cattle manure (MM22)	424.6	378.8
Anaerobic digestion of pig manure (MM49)	249.4	235.3
Covering slurry with impermeable plastic cover (MM47)	126.7	130.4

Anaerobic digestion of manure reduces GHG emissions in two ways; emissions of methane from manure are reduced, and when the digested methane is captured and fed into the gas grid, or used to power CHP plants, then CO_2 emissions from fossil fuel combustion are also saved.

The most certain GHG mitigation impact of covering slurry stores with an impermeable cover is to reduce indirect N_2O emissions through reduced volatilization of NH_3. However, although results of experimental studies are inconsistent, on balance there is also likely to be reduced emissions of direct N_2O and CH_4 emissions when impermeable covers are used (CCC, 2020a; IPCC 2019 Refinement, Table 10.17 footnotes).

2.3.4.3 Trade-offs and synergies between N waste and GHG emissions (and other impacts)

Measures to reduce emissions from housing and storage

In general, measures to reduce emissions from housing and storage create synergies for Nr emissions and GHG mitigation, by reducing indirect N_2O formation after atmospheric deposition and leaching. There are no clear trade-offs between measures to reduce NH_3 emissions from housing or storage and GHG mitigation, except for the increased energy requirements of operating a closed environment with climatization and forced ventilation, as opposed to natural ventilation. However, this has not been quantified.

There may be either co-benefits or trade-offs for animal welfare, depending on the measure employed. Slatted floors have in general been associated with reduced welfare, in comparison to systems with straw bedding which tend to increase livestock comfort. Barn climatization would in general improve animal welfare by reducing exposure to extreme temperatures.

Both housing and manure storage measures which reduce NH_3 volatilisation will also likely have co-benefits for reducing NMVOC emissions and odour nuisance.

Manure processing measures

Anaerobic digestion provides a clear synergy between GHG emissions mitigation and nutrient cycling. However, there are some caveats to how AD is implemented which must be borne in mind. In general, operating the digester plant solely with livestock manure is usually not financially viable due to low CH_4 / volume ratio, therefore most digesters co-digest other organic materials (e.g., food waste, maize silage, energy crops). If the proportion of crop-based feedstocks is high, the synthetic N required to grow the feedstock (with associated losses) will largely cancel out benefits of increased manure nutrient cycling, as well as reducing the life-cycle GHG mitigation potential. To counteract this potential for perverse environmental consequences, in Germany and Denmark, a minimum manure quota is required to qualify for the feed-in tariff available on the electricity or gas generated. The UK would need to adopt a similar system, to ensure AD plants are run primarily with food waste and livestock manure as feedstocks as far as possible.

2.3.5 Key measures for reducing nitrogen losses from soils

As described in Section 2.3.1, measures to reduce N losses to the environment need to ensure that more N becomes available to meet crop and animal needs and the risk of pollution swapping needs to be avoided. Key tools to achieve this for soils are nutrient management plans, precision application and placement of fertilizers, low emission spreading techniques, different types of fertilizer and the use of inhibitors to reduce N transformations to polluting forms (see Table 2.3.6).

Table 2.3.6. Effective measures for reducing N losses from soils. Downward arrows indicate a reduction in losses: ↓, small to medium effect; ↓↓, medium to large effect (UNECE, 2021)

Measure	NH_3	NO_3	N_2O	Total N loss	CH_4	Bio	Dependencies	Current uptake[3]
Nutrient management plans (and implementation of them)	↓	↓↓	↓	↓↓	↔	no direct impact	1	75%
Deep placement of mineral fertiliser	↓↓	↔ /↓	↓/↑	↓	↔	no direct impact	2	?
Precision application: variable rate application	↓?	↓?	↓?	↓?	↔	no direct impact		25% (2019)
Low-emission manure spreading techniques (group of measures including band spreading, injection and rapid incorporation of solid manure)	↓↓	↓/↑	↓/↑	↓/↓↓	↔	no direct impact	2	Variable depending on manure type
Replace urea fertiliser with other forms	↓↓	↔	↓/↑	↓↓	↔	no direct impact	2	See main text
Urease inhibitors	↓↓	↔	↔ /↓	↓↓	↔	no direct impact	2	? 1-5%
Nitrification inhibitors	↔ /↑	↓/↓↓	↓↓	↔ /↓	↔	no direct impact	2	? Minimal

Key:

1. *The impact relies on the nutrient management plan being well-informed and well-executed. It therefore also necessitates actually applying nutrients at the appropriate rate, time and form (rather than just planning to do so!);*
2. *The avoidance of trade-offs between ammonia reduction and increase in losses through other forms of Nr relies on downward adjustment of N quantity added, to account for reduced losses;*
3. *Evidence of current UK uptake of measures is sourced from the UK's Informative Inventory Report 2021 (Churchill et al., 2021), the underlying statistics from Defra's "British survey of fertilizer practice (BSFP, 2019) and "Farm practice survey for England" (FPS, Defra, 2020) and Eory et al. (2015) where applicable.*

Nutrient management plans

Nutrient management planning is the process of estimating all of the nutrient requirements of crops and grassland on the farm, usually at the field scale, and deciding how these will be met with available sources of organic and inorganic nutrients, giving priority to organic nutrient sources (UNECE, 2021). The plan should include the "4Rs" of nutrient stewardship, including rate, time, form and place. The plans need to be flexible depending on the weather and other factors during the growing season, so are underpinned by guidance and decision support tools. Ideally, plans should include provision for soils testing (pH and existing nutrient levels) as well as analysis of N available in manure before application, to correctly calculate the application rate required. Manure analysis is a key facilitative tool for ensuring that N loss savings from housing and storage are not wasted by over-application and resulting high levels of N losses from soil. It is also likely to facilitate transport of manure from livestock to arable farms, as arable farmers can more confidently use manure if they know the correct application rate. Manure analysis is also essential for determining other nutrients, such as the phosphorus content, which also needs to be matched to soil requirements to avoid runoff and eutrophication of aquatic systems.

Table 2.3.6 shows that nutrient management plans can be very effective at reducing total N losses and nitrate losses to aquatic systems in particular. However, this is only the case if they are developed by farmers and/or their advisors to meet the specific needs of their soils, which can vary from field to field and even within fields. It is therefore impossible to stop all N leakage but good plans can save farmers money in reduced fertilizer needs and significantly reduce N losses causing human health and environmental impacts. Current implementation in the UK has been summarized as:

- 57% of holdings, and 75% of farmed area has a nutrient management plan (FPS, Defra, 2020);
- 28% of farmed area has manure tested regularly;
- 93% of holdings refer to the plan at least once per year;
- Only around 20% of holdings calibrate manure or slurry spreaders regularly, following testing or change in manure characteristics. This could indicate a limited ability of farmers to take advantage of reduced upstream N losses;
- In 2019, only around 15% of tillage area and 2% of grassland area was tested for N (BSFP, Defra, 2019).

There is clearly a lot that could be done to improve this situation, especially on smaller and medium sized less technically advanced farms, that can bring significant benefits to the farmer and the environment. Barriers to progress include the requirement for knowledge, time and care to construct the plan, as well as cost and time for manure and soil testing to properly implement the plan. It has been estimated that 78% of plans (by land area) were based on professional advice in 2020 (FPS, Defra, 2020), which is a solid base but peer to peer learning is required to build the trust required for farmers to take the risk of cutting back fertilizer additions that have traditionally brought them profit. For example, crop yield response to fertilisation sharply increases at low fertilisation rates, but as fertilisation rate increases the additional gain in yield diminishes. At the economic optimum the cost of the additional N fertiliser results in the same amount of additional income ('break-even ratio') from the sales of the product (AHDB, 2020). As described above the yield response depends on a variety of well predictable and less predictable factors (e.g., crop variety, plant-available N content of the fertiliser and soil, soil pH, growth conditions during the season, pests and diseases). Most farmers use some form of decision rules and tools to optimise their fertiliser use (Beegle et al., 2000; Defra, 2019) but may keep an over-application margin as a protection from potential yield penalties which could happen with better-than-expected growing conditions (Eory et al., 2020). Under fertilisation might also happen, resulting in suboptimal utilisation of land, and avoiding this requires farmers planning their fertiliser needs based on a recommendation system, considering field and crop characteristics (i.e., creating and using a nutrient management plan). The advantages of planning from the reduced synthetic N application, combining savings both in organic and synthetic N use, and the financial cost of inorganic fertilizer need to be clear to farmers and make sense for their farm business. If following the plan means manure has to be stored or livestock farmers cannot spread all manure on their own land, other solutions must be found, which is costly in time and effort. Or, alternative uses need to be found for the manure.

Precision application: deep placement of mineral fertiliser

Deep placement of fertiliser is an effective way to reduce NH_3 losses by reducing exposure to the air, so is particularly useful for urea where losses would be high for surface application. Placement can be through slot injection (often simultaneously with seed drilling), or with immediate incorporation. NH_3 losses can be reduced by 80-90% (Bittman et al., 2014) and it has been shown to mitigate N_2O emissions in a Swedish field trial with cereals (Rychel et al., 2020), although deep N placement benefits are likely dependent on weather conditions and soil type. We have found no quantification of uptake in the UK but it may be hampered by the investment cost of the specialised machinery required. Another potential barrier to uptake is the risk that concentrated areas of nutrients in the soil increase the risk of leaching, and that plant root growth is reduced, making them more susceptible to droughts.

Precision application: variable rate application

Variable rate application is one aspect of a broad group of practices grouped under the umbrella of “precision agriculture”. Variable rate application of fertiliser aims to match crop N requirements with application rates at very fine spatial (and temporal) scales – as opposed to nutrient management plans which are generally at whole-field scale. In terms of crop production, precision farming is a wide group of rapidly developing technologies enabling the farmer to respond to inter- and intra-field and temporal variability in crop needs when applying inputs (e.g. seed, fertiliser, water, pesticides), increasing input use efficiency (Aubert et al., 2012; Diacono et al., 2017). Precision agriculture is underpinned by: technology to record characteristics of the crop or soil (e.g. yield mapping, canopy sensing, soil mapping) to a high spatial resolution; a decision-support tool to turn that data into decisions about fertiliser application; and guidance technology to control application machinery. The principle is that N fertiliser application can be reduced, with a reduction in all forms of Nr emission and N loss and increase in NUE, as well as reduced fertiliser manufacturing emissions.

Precision farming methods can be used both for crop and grass production (Berry et al., 2017). Experimental evidence on the N fertiliser use and yield effect shows a large variation, between -57% and +1% and -2% to 10%, respectively (Anas et al., 2020; Casa et al., 2011; Ehlert et al., 2004; Link et al., 2008; Mantovani et al., 2011; Van Alphen et al., 2000; Welsh et al., 2003a; Welsh et al., 2003b), as discussed by Eory et al. (2020). In some cases, precision farming has very little impact, as efficient systems are already deployed. However, most studies show positive impacts where precision farming methods are applied in terms of yield and reducing

N use. Most potato and wheat farmers in the UK perceived a -5% - +5% effect of the technology on N fertiliser and fuel use, and a 5-10% increase in wheat yield (Barnes et al., 2017). Eory et al. (2020) modelled the effects of precision farming and found that N use was assumed to be 5% lower, the yield 7.5% higher, and fuel use 3% lower. A National Statistics survey for Defra in 2019 shows that in England a variety of precision farming technologies are employed (Figure 2.3.2).

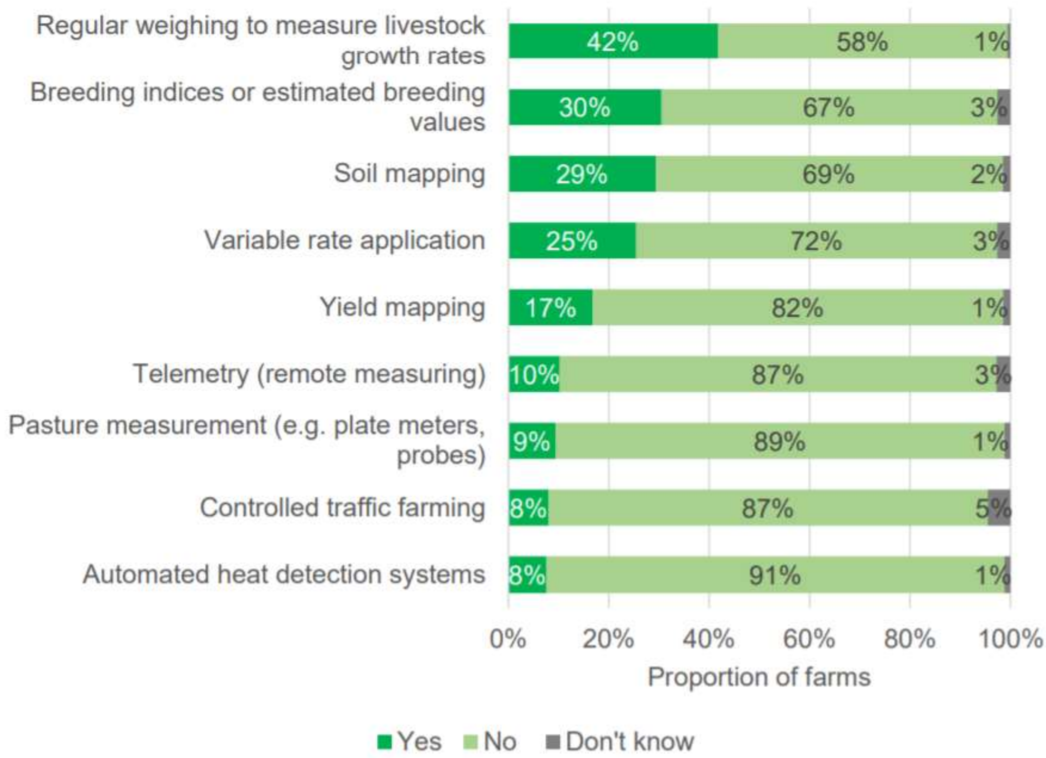

Figure 2.3.2 Precision Framing Techniques used on farms (National Statistics survey (Defra, 2020b). For farms where the technique was applicable.

The survey shows that 25% use variable rate application (c.f. 16%; Defra, 2013), though only 17% (c.f. 11%; Defra, 2013) uses yield mapping (Defra, 2020). The implementation rates are highest for cereals with 81% using precision farming techniques to improve accuracy and 60% to reduce environmental impacts, followed by other crops (72 and 53% respectively), dairy (62 and 37% respectively), mixed (56 and 30% respectively), pigs and poultry (55 and 33% respectively) and different grazing types (around 40 and 20% respectively). For all farms the values were 59% using precision farming techniques to improve accuracy and 38% to reduce environmental impacts (Defra, 2020). This is dependent largely on farmer attitudes towards the methods, and studies have found that many in the industry are resistant to a reliance on technology, instead preferring traditional methods of management (Aubert et al., 2012; Lindblom et al, 2016). The most common reason for not adopting precision farming methods is perception of ease of use in day-to-day activities, lack of context, and the limited impacts that the methods may have. The relevance to the farm also varied by the farm type, with over three quarters (78%) of lowland grazing livestock farms saying the techniques were irrelevant to their farm, as opposed to just under half (47%) of cereal farms (Defra, 2020).

Eory et al. (2020), state that the financial implications consist of the capital and maintenance cost of the equipment, the subscription costs to data providers and software costs. Savings can be expected from reduction in fertiliser and fuel use, and income can increase from improved yield quantity and quality. Further gross margin impacts can include a change in labour requirement.

Current attitudes in the farming industry are that N savings via precision farming methods are currently very limited via the basic approaches available for crop fertiliser application. Reliance on sporadic satellite imagery, tractor mounted sensors and previous years yield data do not have the temporal consistency required to effectively manage N application in a way that will drastically reduce losses. Where these approaches may have the most impact in the future is in reducing N fertiliser inputs to grazed grasslands, which receive highly heterogeneous applications of N in the form of animal waste. By mapping where N is already in excess, applications can be tailored to only add fertilisers to areas where it is required (Figure 2.3.3). This is an emerging technology, which may develop over the next few years and allow N savings from precision farming efforts to improve significantly.

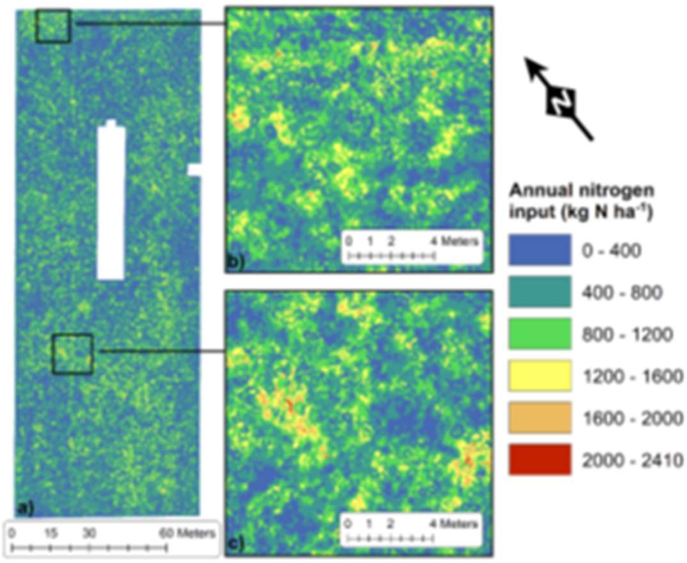

Figure 2.3.3: (a) Annual N applied from fertiliser, urine and dung deposition in kg of N ha^{-1} based on the addition of the nine grazing times of the 60 dairy cows in 2017, (b) and (c) two different sections of the annual N map showing location of high aggregation of N input. Figure 4 in Maire et al. (2021) (In Press).

Low emission manure spreading techniques

Low emission manure spreading is a group of measures which aim to reduce losses of NH_3 by volatilisation during and after spreading manure on soil. Examples are open- and closed-slot injection of slurry, band-spreading of slurry using trailing hoses or shoes, and rapid incorporation of surface-spread solid manure. These measures work by reducing the surface area and/or and time that manure is exposed to air. Large reductions in NH_3 emissions are possible – up to 70-90% NH_3 reduction with injection of slurry, and up to 90% from immediate incorporation of solid manure (Bittman et al., 2014).

Injection and incorporation are also only applicable to certain circumstances: incorporation of solid manure is only possible on arable land and before sowing; injection of slurry is only possible on grassland and at certain points in arable cycle. Band spreading with a trailing shoe or hose has fewer constraints than injection, though emission savings are lower, unless combined with a technique such as slurry acidification or mechanical separation (spreading the liquid fraction), which helps to limit emissions.

Current implementation in the UK varies by manure type and where applied. The UK Informative Inventory Report (Churchill et al., 2021) estimates that although 100% of digestate from anaerobic digestion is applied with low-emission techniques, only 9% and 12% of pig and cattle slurry respectively is injected, and between 7 and 23% of manure is applied with rapid incorporation, trailing shoe or trailing hose techniques, depending on the manure type. There is clearly scope to increase uptake of these measures, although a major barrier is that investment costs in machinery can be high, so equipment rental schemes may be appropriate.

Replace urea (chemical) with other mineral fertilisers

Chemical urea is quickly decomposed into CO_2 and NH_3 upon application to soil, by urease enzymes naturally occurring in the soil. Volatilisation of NH_3 from urea is far higher than for other N fertilisers such as ammonium nitrate, so switching the type of fertiliser could result in considerably lower N losses, up to 90% reduction in NH_3 (see trade-offs section below). This measure needs to be coupled with decreased application rates from the saved NH_3. This should mitigate increased losses of other Nr species (see trade-offs section). Both the absolute quantity of urea containing fertilisers, and the percent of total synthetic N applied has actually increased since the late 1990s, from around 6% in 2000 to 18% in 2019 (Churchill et al., 2021). The agronomic and economic reasons for urea application are likely important drivers of this trend, but nonetheless it indicates there could be potential to switch away from urea to other fertilisers (e.g., organic fertilisers). Ammoniacal N is not appropriate for all kinds of fertilization, so may require changes to fertilization strategy. This measure is not relevant for organic systems.

Urease inhibitors and nitrification inhibitors

Urease inhibitors slow the process of hydrolysis of urea into NH_3 and CO_2 by naturally occurring urease enzymes in the soil. This reduces loss of NH_3 by volatilisation by allowing more time for urea to be absorbed into the soil, and reduces local peaks in pH which favour volatilisation. Use needs to be coupled with decreased application rates. Nitrification inhibitors inhibit conversion of ammonium N to nitrate N by soil microbes, which reduces N_2O emissions as a byproduct of the nitrification process, and can therefore also decrease N loss through leaching, as ammonium is less liable to leaching than nitrate (Eory et al., 2019). Both of these can be used with organic and synthetic fertilisers. Urease inhibitors can reduce NH_3 volatilisation by 18 – 95% depending on conditions (Bittman et al. 2014). There is limited evidence under field conditions of agronomic benefits, as well as cost and the use of inhibitors is not permitted under organic regulations. Urease inhibitors are currently used for a few percent of urea application (T.Misselbrook, pers. comm.), but no quantification of current use of nitrification inhibitors was found, and is assumed to be minimal.

2.3.5 1 Key measures for reducing GHG emissions

Low emission manure spreading techniques may also reduce GHG emissions. This is a crucial measure to implement if measures to reduce N emissions have been applied upstream in the manure management chain, as the measure relies on adjusting to lower losses by reducing application rate, to avoid N leaching and loss to the atmosphere. UNECE (2021) state that while there is some risk of trade-off between ammonia and other forms of N loss from the applied slurry, when considering the farm and landscape scale, there is the opportunity to decrease these N losses, as the increased N use efficiency, as a result of the measure, allows a reduction of fresh N inputs. Indirect N_2O and NOx emissions resulting from atmospheric ammonia deposition to forest and other land are also reduced. Importantly, an alternative fate for saved N needs to be found if manure N available is greater than the amount that can be applied, but a key benefit is the potential to displace synthetic fertilisers and save costs.

Overall, precision farming can reduce GHG emissions and GHG emission intensity of crop production in multiple ways: increasing yield while reducing N fertiliser application, reducing tillage and thus increasing soil carbon sequestration, reducing fuel consumption and reducing other inputs to field operations (impacting off-farm emissions) (Balafoutis et al., 2017).

The sixth carbon budget scenarios (CCC, 2020a) exclude the take-up of four crop and soil related measures assessed; pH crops, crop health, bio-stimulants and precision crop farming. This was done to avoid double counting, as CCC assumptions on crop yield improvements are included as options to release land for other uses, and already imply a more efficient use of N. Although they have not included the abatement savings from these measures, the CCC state that it is important that farmers are encouraged to take these up to reduce emissions from crops and soils. The CCC (2020a) cites evidence of a large gap between the best and worst performing farms and a wide distribution of yield rates, irrespective of soils and climate. Better management practices through measures such as good soil structure and fertility (e.g., through crop rotation); selecting the optimum planting period and tillage; ensuring good crop nutrition (both optimum fertiliser and trace elements) and protection from weeds, pests and diseases could support higher average yields and close the performance gap between the best and worst farms.

2.3.5.2 Trade-offs and synergies between N waste and GHG emissions (and other impacts)

The use of microbial inhibitors (which slow fertiliser release to allow crops to consume N before the microbes) have been shown to reduce N_2O emissions from artificial fertiliser in the UK by 25-50% (Cowan et al., 2020), but would add significant costs to N fertiliser use, and the long-term ecological impacts are not fully known. Slowing the release of N from fertilisers with inhibitors has also been shown to increase NH_3 emissions in some cases, so pollution swapping must be considered (Lam et al., 2017). As the average emission factor (EF: the % of N applied that is converted to N_2O) of N is higher for ammonium nitrate (approx. 1%) than urea (approx 0.5%), changing fertiliser use may reduce N_2O emissions, but urea has significantly higher NH_3 losses associated with its use, so pollution swapping becomes a major factor, though urease inhibitors may reduce NH_3 emissions from urea by up to 90% (Cowan et al., 2019). Even the use of agroecological/regenerative farming methods such as N fixing cover crops (legumes, clover etc.) will result in N emissions similar to artificial fertiliser use when crops are tilled into the soil and mineralization occurs as crop residue mineralization results in approximately 1% losses of N as N_2O (IPCC, 2006). However, these methods will still likely reduce overall GHG emissions due to the energy intensive industrial processes used to create N fertilisers, which currently account for 2% of all energy use in the world (Sutton et al., 2013).

As stated above, nitrification inhibitors can reduce N_2O emissions as a byproduct of the nitrification process by 35-70% (UNECE, 2020), and can therefore also decrease N loss through leaching, as ammonium is less liable to leaching than nitrate (Eory et al., 2019). Nitrification inhibitors will also reduce losses of N_2 and NOx. There is some evidence that in certain conditions nitrification inhibitors could increase volatilisation of NH_3 as N is retained in that form for longer, and also that delay in conversion to nitrates can inhibit uptake by plants.

2.3.6 Key measures for reducing N losses from crop and land use

Landscape measures refer to those measures related to designing crop rotations and non-crop vegetation features of the landscape to intercept potential sources of N pollution before a problem is caused (UNECE, 2021). Effective measures for reducing N losses from crop growing and land use are shown in Table 2.3.7.

Table 2.3.7. Effective measures for reducing N losses from crop growing and land use. Downward arrows indicate a reduction in losses: ↓, small to medium effect; ↓↓, medium to large effect (UNECE, 2021)

Measure	**NH_3**	**NO_3**	**N_2O**	**Total N loss**	**CH_4**	**Bio**	**Dependencies**	**Current uptake**
Increased land cover of perennial crops or set-aside/unfertilised grassland	↔	↓↓	↑/↔ /↓	↓/↓↓	↔	↑		?
Agroforestry[3]	↓	↑/↓	↔ /↑	↓	↔	↑	4	?
Afforestation and hedgerow creation	↓	↓↓	↑/↓	↓↓	↔	↑↑	4	?
Cover / catch crops in rotations[3]	↔	↓	↑/↓	↓	↔	↑	1	?
Nitrogen fixing plants in rotations[3]	↔ /↓	↓	↓	↓	↔	↑	1,2	See text in Section 2.3.6.1
Constructed wetlands	↔	↓↓	↑	↑	↑	↑		?
Avoid grazing in high-risk areas	↔	↓↓	↓	↓	↔	↑		See text in

Key:

1. *Overall reductions in N loss, N_2O emissions and leaching depend on appropriate timing of cover/ catch crop establishment and incorporation into soil;*
2. *Effect is relative to use of mineral fertilisers;*
3. *These measures are not category 1 with large mitigation effects according to UNECE (2021), but are included to allow discussion of issues related to their use as GHG mitigation or agro-ecological measures;*
4. *The impact of these measures depends on the spatial configuration of N sources and sinks in a landscape*

2.3.6.1 Increased land cover of perennial crops, set aside / unfertilised grassland belts, agroforestry, hedgerows and afforestation

Permanent vegetation in the landscape, in the form of trees, hedgerows, fertilised or unfertilised grassland increases Nr retention through higher soil carbon stocks, immobilisation in plant biomass, and interception of lateral flows of leached Nr. Perennial crops (e.g. improved grassland) and agroforestry generally occupy productive agricultural land, whereas set-aside / unfertilised grasslands and afforestation are likely to be more efficiently used as buffer strips along field margins, for example next to water courses. Trees also mitigate high concentrations of volatilised NH_3 in the air (for example from intensive pig or poultry facilities) by filtration, preventing transport to sensitive areas. With the exception of perennial crops, these measures are implemented on non-cropped land and work by intercepting flows of Nr generated from sources elsewhere. Their efficacy in mitigating Nr pollution from elsewhere therefore depends on the spatial relationship between the sources of Nr and the location of the measure. There is likely to be an attitudinal barrier to taking land out of production, as well as a knowledge gap for uptake of systems such as agroforestry which are not yet common in the UK (FFCC, 2019). Perennial crops remove flexibility in how land can be used (i.e. an opportunity cost). Economically upfront costs are not high, but there is potential for income reduction (though perhaps mitigated by lower costs). It is difficult to assign a particular uptake figure for these measures, as there is no definition of what full implementation actually looks like. There is however likely to be a large scope for increasing the area of agroforestry and length of linear hedgerow / rough grassland features.

Cover/catch crops and N fixing plants in rotations

Cover/catch crops are non-cash crops that are incorporated into the main crop rotation over winter to reduce N leaching by capturing excess mineralised N in periods when crops are not growing. Nitrogen fixing plants (mainly legumes such as clover) are also part of crop rotations, and can be used as cover/catch crops in themselves, as part of leys in the fertility-building phase of organic crop rotations, or as intercropping plants. Biological N fixation by legumes makes up the key means of adding N into organic farming systems, increasing N retention in the soil as compared with mineral fertilisers due to N largely being available in "slow-release" organic compounds. These measures contribute to reducing synthetic fertiliser use but need good timing of cover crop establishment to immobilise mineralised N from the previous crop, and of incorporation to mitigate risk of N loss. In the UK 26% of holdings are increasing use of clover in grassland, and 18% increasing use of legumes in crop rotations. No information was found on use of cover/catch crops, but CCC (2020a) assume that it is minimal.

Constructed wetlands

Constructed wetlands are areas designed to receive runoff and drainage water from fields or other Nr sources such as manure storage areas. Plants and drainage are designed to encourage anaerobic conditions causing microbial denitrification of nitrates to N_2, thereby reducing losses to water bodies.

Avoid grazing (or manure and fertiliser application) in high-risk areas

High risk areas include those in close proximity to, or connectivity with vulnerable surface waters and groundwaters. Preventing grazing and manure / fertiliser application in such areas reduces the quantity of nitrates leaching into these waters. Sensitive areas could also include those prone to waterlogging, which would be vulnerable to compaction and poaching with associated N_2O emissions.

There are rules in place for grazing in nitrate vulnerable zones (NVZs) in the UK, but there is potential for expanding the area classified as NVZs to increase uptake of this measure and also for strengthening enforcement (see Section 3.4.2 for more detail on this). According to the FPS (Defra, 2020), 62% of holdings routinely take action to keep livestock out of water courses.

The cost of establishing and maintaining fencing to stock-proof riparian margins is a key barrier to greater compliance, especially in certain upland areas where watercourses are less well-defined.

2.3.6.2 Key measures for reducing GHG emissions

Landscape measures can provide increased carbon sequestration in soil (all) and biomass (woody vegetation in particular) and promote growth and nutrient use efficiency reducing N losses. GHG mitigation measures included in the 6th Carbon Budget agriculture mitigation scenarios related to land use are shown in Table 2.3.8.

Table 2.3.8. GHG mitigation measures included in the 6th Carbon Budget agriculture mitigation scenarios related to land use (SRUC 2020)

Measure	Balanced Pathway abatement 2035 ($MtCO2e$)[1]	Widespread engagement abatement 2035 ($MtCO2e$)[1]
Grass and legumes (MM20)	524.4	540.4
Cover crops (MM2)	187.3	206.3
Grass Leys (MM8)	188.8	172.7

Red – measures with cost effectiveness under the C price

Biological N fixation using grass-legumes mixtures (MM20 in Eory et al., 2020) can reduce N_2O emissions arising from the use of synthetic N fertilisers by substituting biologically fixed N in crop production (Lüscher et al., 2014). Biological N fixation occurs as legumes form symbiotic relationships with bacteria (Rhizobia) in the soil that can transform atmospheric dinitrogen to N compounds the legumes can utilise, diminishing their need for synthetic fertilisers. Besides the fixed N supporting the growth of the legume crop (e.g. clover), part of the N also becomes available to the grass, reducing the need for fertiliser. This effect becomes substantial above a clover content of around 20%-30% in the sward.

Cover crops are non-cash crops that can be integrated into the main crop rotation (MM2 in Eory et al., 2020). They are typically grown either to maintain soil cover during fallow periods (Ruis and Blanco-Canqui, 2017), or are planted alongside main crops to reduce bare soil area and reduce erosion. The former is either ploughed in as green manure or killed with herbicides (often with glyphosate) under no-till regimes. Cover crops can be divided into catch crops, grown to prevent N leaching (Cicek et al., 2015), and green manure, grown to improve soil physical conditions (Alliaume et al., 2014) and main crop nutrition (Dabney et al., 2011). Cover cropping serves to maintain soluble organic carbon input to soil (Rutledge et al., 2017), prevent erosion (De Baets et al., 2011), decrease N leaching (Blombäck et al., 2003), and increase main crop productivity (Lal, 2004).

Grass leys are perennial non-woody biomass that is planted as part of an arable and temporary grassland rotation (MM2 in Eory et al., 2020). The introduction of perennial plants, including grass leys, into an arable crop rotation can increase the positive effects of rotation practices (Gentile et al., 2005; Prade et al., 2017). Loss of soil organic matter (SOM), with corresponding negative effects on crop yield and CO_2 emission, is possible if arable-only rotations are practiced over the long-term (Prade et al., 2017). Diversification of arable cropping systems with grass leys serves to increase the quantity and continuity of below-ground residue returned to the soil (Fu et al., 2017; West and Post, 2002). This in turn can support microbial activity and diversity, and ensures continuity of root-derived C inputs to soil, increasing SOM. A key issue in the integration of grass leys into arable rotations is loss of crop production (Maillard et al., 2018).

2.3.6.3 Trade-offs and synergies between N waste and GHG emissions (and other impacts)

Landscape measures can have direct and indirect co-benefits via:

- Mobilisation of P and K from deep soil horizons by agroforestry, to make available to crops which in turn allows increased NUE (Lampkin et al. 2015) and reduces N losses;
- Biodiversity (especially unfertilised grassland and woody vegetation), including pollinating insects and biocontrol agents which in turn promote productivity (Lampkin et al., 2015);
- Wind shelter provided by trees and shading provided by trees is beneficial to animal welfare and also crops during dry conditions (FFCC, 2019), and soil erosion is reduced;
- Agroforestry can increase overall productivity per area of land, leading to economic benefits;
- Potential for woody biomass to be harvested for bioenergy, with associated energy sector CO_2 reductions;
- Potential cost savings of taking unprofitable land out of production.

Trade-offs associated with landscape measures are:

- Increased retention of Nr could potentially lead to higher N_2O and N_2 emissions, but this effect is likely to be minor (UNECE, 2020);
- Integrating non-crop vegetation into the landscape may reduce overall cropped area and production levels, therefore risking exporting emissions. Therefore, non-crop vegetation measures need to be spatially targeted to make sure they are as effective as possible, and on land where the opportunity cost of foregone agricultural production is relatively low.

Cover/catch crops and N fixing plants in rotations can have direct and indirect co-benefits via:

- Cover/catch crops reduce soil erosion, and increase addition of organic matter to soil (compared with mineral fertilisers), increasing soil carbon;
- Cover/catch crops also help with weed suppression;
- Biodiversity benefits as a food resource, including for pollinators;
- Provide additional grazing for ruminants as part of mixed farming system;
- Deep-rooted cover crops (such as canola) can help to maintain soil permeability in low-tillage systems, and tap into P and K sources in deeper soil horizons.

Trade-offs associated with cover/catch crops and N fixing plants in rotations:

- There is a risk of a pulse of N mineralisation following incorporation of catch/cover crops and legume leys into the soil, which can lead to increased N losses through volatilisation, denitrification and leaching. Timing of incorporation to allow uptake of mineralised N by the following crop can mitigate this, and using a diverse cover crop / legume sward to provide a more complex residue structure and better nutrient release profile. Also, grazing of the cover crop can help to reduce the quantity of above-ground residues incorporated (Lampkin et al., 2015).

Constructed wetlands can have direct and indirect co-benefits via:

- Carbon sequestration in soil and biomass;
- Providing habitat for wetland biodiversity in the farmed landscape.

Trade-offs associated with constructed wetlands:

- Potential for increased N_2O and CH_4 emissions due to anaerobic conditions;
- Due to the emphasis on denitrification, this encourages loss of potentially valuable Nr to N_2, supporting a linear rather than circular N flow.

Avoiding grazing in high-risk areas by leaving riparian margins and other areas ungrazed can also benefit biodiversity and increase biomass and soil carbon stocks (as it is a kind of set-aside; UNECE , 2020).

2.3.7 System measures

The following measures concern changes to the overall farming systems – potentially involving collaboration between multiple farmers across regions – to increase N use efficiency at the landscape scale. Production of alternative foods such as microbial protein, meat culture and insect protein could also be seen as "system" measures, but these are discussed separately in Sections 2.1.1 on dietary change and 2.1.4 on reducing the impact of imported livestock feed.

Mixed farming and manure redistribution

As described in Section 1.4.1, farm specialisation and large-scale spatial segregation of arable and livestock farming in the UK hinders effective recycling of nutrients between arable and livestock systems. Mixed farming combines livestock and arable agriculture either on the same farm or within the same landscape, providing opportunities to close nutrient cycles at a local scale and increase landscape-scale N use efficiency. This reduces the need for synthetic fertiliser application. This can involve both close proximity of separate crop and livestock systems allowing easy transport of manure and/or fodder, or direct integration of livestock grazing into crop rotations through use of temporary grass and/or legume leys. Mixed farming at the holding level provides farmers with full control of how and when to integrate grazing or manure into arable rotations, but there are several models for inter-farm cooperation and exchange (e.g. "muck for straw") at larger spatial scales (Martin et al., 2016). These include collective land use planning where farmers effectively share land, direct exchange of local resources (forage, straw, manure) between local farms, and exchange mediated via local or national organisations. Organic farming fosters mixed enterprises or long-term cooperation between neighbouring ones, due to a reliance of organic arable agriculture on manure (Martin et al., 2016).

At larger scales nutrient cycles can be closed through redistribution of manure (or manure nutrients) over longer distances from high concentrations of livestock production to mainly arable areas (which also produce livestock feed). However, as discussed in Section 1.4, transport of slurry of farmyard manure over long distances is problematic due to its bulk. One analysis in Ireland suggests that beyond distances of around 50-75km, transport of manure becomes energetically questionable (Fealy & Schröder, 2008). Another analysis of the economics of raw pig manure transport from Ireland found that for distances up to 15 km transport of manure is the most cost-effective option to deal with excess supply, but at greater distances other treatment options (e.g. anaerobic digestion) become more cost-effective (Nolan et al., 2012). Therefore, nutrient stripping techniques discussed in the section on waste management, or methods to dry manure to reduce mass, may be a vital part of the solution for closing nutrient cycles at larger spatial scales (e.g. national scale).

Nutrient stripping techniques also have the advantage of producing a fertiliser more similar in composition and nutrient release profile to synthetic fertilisers, which can therefore act as a more direct replacement for crops.

Potential barriers to transitioning to more mixed farming are complex, but include the increased decision-making complexity and knowledge requirements of running a mixed farm or cooperating with neighbours, sunk costs in equipment and risk aversion to change. "Muck for straw" swaps may also be contingent on demand for manure; there is evidence to suggest that arable farmers may value the flexibility of solid manure which can be left in piles on fields until required, and therefore currently have little demand for slurry (Waterton et al., 2018). As mentioned above, there are also practical barriers to manure replacing synthetic fertilisers for all aspects of crop fertilisation, which may require a radical change in farming practices to overcome.

Co-benefits

Co-benefits of mixed farming include reducing the need for imported phosphorus (with associated production impacts), as well as benefits of a more heterogeneous landscape for biodiversity and associated ecosystem services (pollination, biocontrol) (e.g. Benton et al., 2003).

Trade-offs

Due to its bulk, as the transport distance increases so do the associated GHG and NOx emissions, as well as financial costs of transport. At some point these will outweigh the benefits of nutrient cycling in reducing N_2O and other Nr emissions and reducing synthetic fertiliser costs. Concentrating nutrients as far as possible will help to increase the practical threshold distance.

Indoor / controlled environment agriculture

Methods such as vertical farming and glasshouse production are on the rise across the world, with particular developments in EU countries such as the Netherlands, Belgium and Spain. The majority of the salad crops consumed in the UK are already grown using these methods (largely imports from EU). Although capital costs are large, production is shifting towards Controlled Environment Agriculture (CEA) as a means by which to increase production where space is limited (high density populations such as EU, Japan, Singapore) or where water and fertile land is limited (South Africa, Gulf States, Australia). In these systems, the nutrients used are recycled in a semi-closed loop, thus NUE is drastically improved. The only major N pollution source from CEA systems (if using renewable energy) would be waste solution entering the sewage networks, although this could be easily treated at point (e.g. algae stripping). Although crops are currently limited to mostly fruit and salads (though aeroponic potatoes have been successfully demonstrated), there is potential for this sector to grow in the future, allowing for a step change in sustainable agriculture solutions. A major barrier to implementation is the significant start-up cost.

Co-benefits

- Indoor agriculture can produce up to 100 times the amount of food per unit area. A reduced land footprint has numerous co-benefits, related to the benefits of counterfactual uses of that land (Eory et al., 2019);
- Up to 99% lower water footprint (Eory et al., 2019);
- The closed system allows effective biocontrol, reducing the need for pesticides.

Trade-offs:
- High energy footprint associated with heating and lighting;
- Opportunity cost of not being able to use the land for other purposes.

2.3.8 Holistic packages of measures

The previous sections in 2.3 have outlined the impact of fairly specific measures for mitigating different aspects of N pollution. Not all measures will be applicable in all circumstances, so the shortlist presented above can be thought of as a "menu" of effective measures to choose from, depending on context. However, as also made clear throughout there are both dependencies and potential synergies between measures, by which the impact of a measure can either be enhanced or nullified. Moreover, a holistic approach to adoption of measures can bring overall benefits by increasing overall system N use efficiency or increasing circularity. Therefore, decision makers at all scales from individual farms through to national level should consider policies to implement packages of complementary and synergistic measures which minimise trade-offs, but with flexibility for individual farmers.

The UNECE guidance for sustainable N management provides useful case studies of suitable packages of measures for a conventional intensive and organic dairy farm respectively, and a Mediterranean tomato farm (UNECE, 2021, pages 189-193).

2.4 Reducing N impact from imported food and animal feed

The UK imports a significant quantity of N in the form of food and animal feed, comparable to N inputs through biological N fixation in the UK. This food and animal feed has a N footprint overseas, and this section considers measures to reduce the overseas footprint of the UK's food and feed imports. There is substantial overlap here with questions around dietary choice, as for example indicated in Poux & Aubert (2018) and van Grinsven et al. (2015); a dietary shift away from livestock products would substantially reduce the amount of animal feed imported, and therefore the UK's overseas N footprint. However, there are certain principles which apply even when diets remain constant. Put simply, there are two ways to reduce the overseas Nr emissions footprint of UK consumption:

1. Reduce (or at least do not increase) the quantity of food and animal feed imported to the UK, and
2. Reduce the Nr emissions per unit product grown overseas for UK consumption.

2.4.1 Limiting the quantity of food imported

There are several potential means of limiting the requirement for food and feed imports, discussed below. However, it must be borne in mind that imports are not bad *per se.* Reducing imports may be an effective measure to reduce global Nr footprint, if regulation is less strict in the producing country than in the UK, or if the UK has a comparative advantage in producing a product. Another factor to consider is the increased opportunity for circularity of N flows when food and livestock feed is produced domestically.

Regarding imported human food:

- Encouraging **seasonal eating** would make best use of domestic production and minimise the proportion of food we choose to buy from overseas;

- An increase in **indoor horticulture** in the UK could help to reduce reliance on imports to provide year-round fresh produce to UK consumers. Indoor agriculture has a low Nr footprint (see Section 2.3.7), but the GHG and wider environmental footprint also should be considered.

Regarding livestock feed:

- Currently, pig and poultry production (and to some extent dairy cattle) in the UK are dependent on imported soy meal, with a high GHG footprint and biodiversity impacts related to land use change. However, Nr emissions from soy production are quite low because it is a N-fixing crop. Increased UK production of traditional grain legumes (pea, faba bean) and alternative ones such as lentils and lupins suitable for the UK climate (NIAB, 2021) could provide manifold benefits. On the one hand this would introduce more legumes into crop rotations to replace synthetic N fertiliser with associated reductions in Nr emissions (see Section 2.3.6). On the other hand, it would provide a domestic high-protein source of animal feed, and also a domestic source of additional plant proteins which would be required under a reduced meat and dairy consumption diet. For example, pea protein is a key ingredient in popular "meat replacement" products such as "Beyond Burger[7]". However, beyond the extent to which additional legumes are required for additional fertility building, further expansion in production should be contingent on it being demonstrated that there is a lower environmental footprint than imported soy (against a variety of types of impact, not just N or GHGs).

- Waste-derived and non-human-edible sources of protein for animal feed can be exploited as far as possible to satisfy animal demand for protein. This may require changes to legislation and regulations, or innovation to make them more digestible as a food resource (e.g., adding phytase to oilseed cake to increase amino acid and P digestibility for pigs and poultry). Microbial and insect-based protein are two promising non-plant types of feed identified by Eory et al. (2015) (see Section 2.1.4).

Clearly, any interventions leading to increased UK production will have implications for UK Nr emissions, land use and other outcomes (e.g. quantity of exports) which must be balanced against the benefits of reduced overseas impact.

2.4.2 Reduce the Nr emissions footprint per unit product of imported food

Potential measures to reduce the Nr emissions footprint per unit product of imported food and animal feed include:

- Food and feed importers working with overseas suppliers to ensure sustainability – via contract terms and conditions, which are part of supermarket branding etc. This could also be through international labelling / certification schemes;

- The UK government lobbying for international commitments (preferably binding) to reducing N waste, such as was done in the Columbo Declaration (see Section 3.1.1);

[7] https://www.beyondmeat.com/products/the-beyond-burger/

- Including N-waste related sustainability criteria as part of food standard regulations included in trade agreements.

These measures will be especially important if actions taken by the UK to cut N waste from UK agricultural production come at a cost which increases the price of domestic produce. This may cause UK farmers to be undercut on price, leading to increased imports if consumers are driven by price. The same result could occur if UK farmers proactively reduce production levels, for example by moving to lower-yielding agro-ecological systems. If imports are increased then to ensure overall global N pollution reduction the production standards of these imports must be ensured. Equally, demanding high production sustainability standards from imported food and feed itself may help to prevent imports being cheap enough to undercut UK production, and thereby could help in limiting import quantities.

2.5 Alignment of measures to reduce N waste with an agroecological transition

Here, we consider the synergies or trade-offs between measures to reduce N and agroecological principles or management practices, primarily through the lens of the commonly recommended measures to reduce N waste discussed in the previous section.

2.5.1 What is agroecology?

The definition of agroecology can be variable depending on the context, but here we follow the definition used by the Food, Farming and Countryside Commission (FFCC) of *"farming in ways that learn from, work with and enhance natural systems"* (FFCC, 2019), which in turn follows the 10 principles of agroecology set out by the FAO (2018) of: Diversity, Synergies, Efficiency, Resilience, Recycling, Co-creation and sharing of knowledge, Human and social values, Culture and food traditions, Responsible governance and Circular and solidarity economy. This is also the definition accepted by WWF.

In practice, this definition is broad and can encompass different types of farming systems, depending on the location. Lampkin et al (2015) provide further useful description of principles and management practices associated with agroecology in the UK context.

Agroecological principles:

- Promoting recycling of biomass (e.g. plant material and agricultural residues) and optimising nutrient availability;
- Ensuring favourable soil conditions for plant growth, particularly soil organic matter and biota;
- Minimising losses from the agricultural system, e.g. through water harvesting, soil and energy management;
- Maximising species and genetic diversity (plants and livestock);
- Enhancing biological interactions and synergies to promote ecological processes and services.

Agroecological management practices:

- Relying on soil biota, e.g. earthworms, to enhance soil structure and fertility, the formation of water stable aggregates, and soil water infiltration;

- Using legumes and symbiotic N-fixing bacteria to fix biological N (e.g. leguminous leys in fertility-building phase of crop rotation);
- Using biologically active soil amendments (e.g. composts) to suppress soil-borne diseases and enhance soil structure and fertility;
- Practicing passive biological control of pests using field margins or beetle banks to encourage presence of beneficial insects;
- Designing cropping systems to disrupt pest life cycles or attract pests away from sensitive crops (including push-pull systems);
- Using crop rotation to manage soil fertility, weeds, and pests and diseases;
- Using diverse cultivar and species mixtures (including combining crops and livestock), to improve resource use efficiency and reduce the spread of pests and diseases;
- Combining livestock species with different grazing behaviours and ensuring effective resource utilisation to maximise nutrition and health benefits.
- Relying on minimal artificial inputs from outside the farm system.

Approaches such as agroforestry and permaculture, organic farming, integrated pest management, biodynamic farming, conservation tillage and regenerative agriculture all fall under the scope of agroecological farming systems (FFCC, 2019). A transition to agroecology in the UK over the next 10 years, as laid out by FFCC (2021) and IDDRI (2018) would see changes in relevant characteristics such as:

- A reduction in livestock numbers, with those remaining being fed on pasture and waste;
- A much higher proportion of livestock manure falling on pasture, and less in housing and stored;
- An increase in mixed and organic farming, with livestock integrated into arable rotations grazing on grass and legume leys, which also fix N.

2.5.2 Alignment of measures to reduce N waste with an agroecological transition

2.5.2.1 Dietary change

As introduced in Section 2.1, dietary change is a central part of the vision for a transition to agroecology (Poux & Aubert, 2018). Specifically, the dietary changes envisaged are to eat "less, but better" meat and dairy, with a particular emphasis on a shift away from pork, poultry and dairy production (with their associated high proportion of concentrate feeds reliant on synthetic fertilisers), but with smaller reductions in beef and lamb production. Ruminants would continue to play a key role in nutrient cycling in arable rotations, and in maintaining extensive grasslands which serve as a source of biologically fixed N that is transferred to arable land via grazing as well as providing biodiversity benefits.

Indeed, without dietary change, a large-scale transition to agroecological practices such as organic farming would likely lead to export of Nr and GHG emissions through increased imports, due to the lower per hectare yields (of the system as a whole) compared with the current levels. Poux & Aubert (2018) quote values of roughly 25% yield reduction for cereals, and between 20% and 45% yield reduction for other crops. A review by van Zanten et al. (2018) showed that feeding livestock on grassland and food waste alone ("livestock on leftovers") across Europe can deliver between 9 and 23g of animal protein per person per day, compared with the 46g per day currently consumed in the UK. Therefore, dietary change is a key enabling factor to ensure an agroecological transition has globally positive environmental consequences.

The key dietary changes presented in Westhoek et al. (2015) necessary to reduce N waste (Section 2.1) generally align with the agroecological vision in the recommendation to reduce overall meat and dairy product consumption. However, the scenarios in Westhoek et al. (2015) do show a greater impact of reducing beef and dairy production on Nr emissions than of reduction in pigs and poultry, which at first sight appears contradictory to the FFCC vision. This is based on the superior feed conversion efficiency of pigs and poultry compared with cattle, meaning pigs and poultry currently (EU-wide) have a lower N footprint per kg of protein in the product than beef and lamb in particular. This pattern is also mirrored by GHG emissions intensities, due also to feed conversion efficiencies, much lower enteric methane emissions from non-ruminants, and can be magnified even further if the "carbon opportunity cost" of land use is included in the footprint (e.g. Ranganathan et al., 2016).

On the other hand, a point to note is that the diet recommendations of Westhoek et al. (2015) modelled the reduction in Nr emissions from different types of meat production based on current practices, which did not take into account the opportunities for reduced Nr emissions intensity of beef and sheep production afforded by allowing cattle to graze extensively on unfertilised pastures, where biological N fixation is the chief source of "new" Nr in the system. Nr emissions from manure deposited on soil are likely to be lower than total emissions from housing, storage and application from manure deposited in housing (see Section 2.3.3), and the impacts more spatially diffuse. Therefore, larger reductions in Nr emissions may be possible from reductions in pork and poultry consumption (only) than is indicated by Westhoek et al. (2015), if accompanied by an agroecological transition. The answer to this question for the UK will hopefully be resolved when IDDRI publish the full results of the regionalized version of the TYFA mode (TYFAregio) for the UK, due to be released in autumn 2021.

The difference in dietary recommendations around which types of meat to cut out correlates with a long-running and polarized debate between sustainable intensification ("land sparing") versus agroecological farming ("land sharing").

A sustainable intensification approach to livestock production emphasizes production of a given quantity of animal protein on the least land possible. This is achieved through an emphasis on indoor pig and poultry production with high feed conversion efficiency, with high inputs of synthetic fertiliser for feed, aspiring to achieve (but currently not fully achieving) a "closed-system" with a tightly controlled diet, health and abatement applied on housing, manure storage and application to mitigate Nr emissions. The land spared can then be used for other purposes such as carbon sequestration, as in the CCC's 6th Carbon budget scenarios. However, local farmland-adapted biodiversity is not accounted for, and implicitly the imports of high-protein animal feed from areas at risk of deforestation continues, so impacts on carbon sequestration and biodiversity in these locations are not reduced. Remaining Nr emissions are concentrated in intensive facilities acting as point sources.

An agroecological approach emphasises ruminants as an important part of an organic agricultural system, recycling nutrients in arable rotations and making use of land unsuitable to grow arable crops. A reduction in land demand due to diet shifts allow sustainable extensification (see Section 2.1), and widespread adoption of lower-yielding but lower N-input systems, where Nr emissions per unit product are higher than for pork or poultry but diffuse and more dominated by leaching. Here, the carbon sequestration potential and biodiversity value of extensive grazing is highlighted (compared with more intensively managed grassland and arable crops), but the opportunity cost for carbon storage and non-farmland biodiversity of not using the land for something else (e.g. bioenergy crops, afforestation, rewilding), may not be accounted for.

It is clear that depending on how different impacts are weighted, different diets could appear preferable. However, there is no need for such a strict dichotomy, and a "3 compartment model" (where high-yield farmland, low-yield farmland, and natural/semi-natural areas coexist in the UK) may bring better results than either pure strategy for some outcomes, such as biodiversity (NFS, 2021). For Nr emissions, this 3-compartment model could work well, making use of spatial planning to encourage low-yielding extensive farming (with diffuse emissions) or semi-natural habitats in sensitive areas, and more intensive agriculture in less sensitive sites. Diet shifts involving a reduction in all meat and dairy products (as in the CCC scenarios and the -50% all meat and dairy scenario of Westhoek et al., 2015) could result in such a shift.

2.5.2.2 Reduction in food waste

Reducing food waste is also a core part of an agroecological transition. Any policies to reduce food waste post farm-gate (i.e. which do not imply anything about production methods) are therefore completely consistent with agroecology.

On-farm production measures to reduce waste may or may not align well with the agroecological transition, depending on the type of agroecological management. For example, any use of novel pesticides to reduce losses of crops to disease would likely not be permitted under organic agriculture rules. However, increasing crop and animal health through breeding and higher welfare conditions would be compatible.

Shortening supply chains is also consistent with connecting consumers better with the food they eat, as well as farm diversification if farmers sell produce directly.

Increasing circularity of nutrient flows via increased anaerobic digestion and composting of food waste will also help to provide a supply of organic N and other minerals such as phosphorus to organic agriculture.

2.5.2.3 Reducing N footprint of imported food and feed

Key points:

- At the heart of the vision for the agroecological transition is the goal of closing N cycles at the territorial level (Poux & Aubert, 2018). Therefore, the agroecological pathway of reduced poultry and pork - which disproportionately rely on imported soymeal as a source of protein compared with beef cattle and sheep – will lead to reduced demand for livestock feed from overseas (Poux & Aubert, 2018, van Grinsven et al, 2015);

- Shifting our fruit and vegetable consumption to align more with domestic production peaks (perhaps complemented by more domestic indoor horticulture), would also help to close the N cycle at the UK level;
- Use of UK land for large quantities of alternative domestically-grown livestock feeds, such as UK pulses or lupins, may or may not be compatible with the agroecological transition, depending on the scale. Whilst a certain increase in these N-fixing crops could play an important role in fertility-building stage of crop rotations, there is a limit to the quantity of additional production this will generate. A key principle of agroecological approaches is to use livestock to complement arable rotations and reducing the competition between producing feed for livestock and crops for direct human consumption, so converting too large a fraction of land to grow these crops may conflict with this principle;
- Development of insect protein feeds for livestock to displace high-protein plant-based feeds, especially if food waste or manure is used as a feedstock, would align well with the agroecological principle of effective recycling of nutrients. However, attention needs to be paid to the efficiency of the process compared with other potential ways to recycle food waste and manure – e.g. through anaerobic digestion and spreading of digestate / manure on land.

2.5.2.4 On-farm measures to reduce N waste

The alignment of on-farm measures to reduce N waste and agroecology has been assessed based on the expert judgement of the project team. Several of the key on-farm measures to reduce N waste align very well with the transition to agroecology, or at least do not conflict with this:

- **Measures to improve circularity of nutrient flows**, including encouraging mixed farming and greater transport of manure or manure nutrients from areas of high to low availability, and potentially nutrient recovery. The caveat here is that currently the Soil Association does not permit the use of sewage sludge or struvite in its definition of organic agriculture (though other forms of agroecological management would permit this). Moreover, the Soil Association also has strict guidelines about the origin of farmyard manure, where use of manure sourced from intensive systems is discouraged (Soil Association, 2021). These restrictions may impede transition to agroecological arable systems.
- **Increasing grazing period of ruminants**, especially if combined with lower stocking rates to match the background rate of N fixation. Consideration of local conditions in this way helps to maximise profitability (FFCC, 2019). Discussed further in Section 2.5.1.
- **Physical low-emission housing and manure storage measures** where animals are housed at least some of the time which is necessary for some cattle breeds and some conditions to avoid damage to soils such as poaching during wet periods, but see below for exceptions.
- **Low-emission manure spreading** and rapid incorporation of manure, on farms where at least some of manure is stored.

- **Agroforestry, permaculture and shelter-belt creation** (e.g. hedgerows)
- **Use of cover/catch crops** is also a key part of conservation agriculture to protect soil, and is one of the key practices forming part of the definition of agroecology
- **Use of legumes in arable rotations** to biologically fix N (to decrease synthetic N use) is a key part of organic agriculture.
- **Precision fertilisation techniques** such as variable rate application of manure.
- **Limiting or avoiding grazing and manure spreading in high-risk areas**

Some measures may not be very compatible, or are not relevant to agroecological systems so prioritising these may cause farmers to invest into non-agroecological routes:

- Use of ammonium nitrates rather than urea-based fertilisers, which is not that relevant due to the emphasis on organic fertilisers and bioloigical fixation in agroecology;
- Low protein diets to reduce N excretion, which implies a larger proportion of livestock feed being delivered in the form of low protein fodder (such as maize silage) and concentrates in controlled conditions. This contradicts the emphasis on grazing for food intake (where protein intake is more difficult to control) in agroecology;
- Use of slurry acidification with strong acids is currently not permitted under Soil Association rules for organic agriculture;
- Use of synthetic urease and nitrification inhibitors is also currently not permitted under Soil Association rules for organic agriculture. However, natural alternatives such as neem seed oil (a nitrification inhibitor) have been shown to be effective, which may be permitted (UNECE, 2021);
- Although cover/catch crops are an important part of conservation tillage, in the UK broad-spectrum herbicides are generally used to terminate them before main crop establishment, which is not permitted under organic regulations. Mechanical termination options (e.g., a "roller-crimper") are available, though may be less cost-effective;
- Indoor horticulture is not generally included within the agroecological vision, although it could perhaps function in parallel as an entirely separate system.

In addition to thinking about the alignment with key N mitigation measures with agroecology, there are some other agroecological management practices with implications for N losses. In particular:

- **Conservation tillage** (e.g. minimum or zero-tillage) practices (including cover crops) can help to reduce leaching by maintaining soil carbon stocks. This helps to retain N in the soil and prevents mineralisation and subsequent loss of N following loss of soil carbon. In addition, there is some evidence that reduced tillage allows buildup of mychorrizal networks in the soil, which increase NUE of crops (Lampkin et al. 2015);
- **Diverse grassland swards** can help to increase the NUE of the plants and reduce the risk of leaching from urine and dung patches or applied manure/fertiliser. Due to the mix of species, N uptake is spread more evenly across the growing season, so precise matching of the timing of application with plant growth is less critical (Lampkin et al, 2015);

- **Intercropping** can lead to increased N use efficiency of the system, both by spreading out the period of maximum N uptake (useful where slow-release organic fertilisers are used), and where legumes are used (e.g. pea and wheat intercropping) by allowing an additional phase of N fixation which reduces the need for fertiliser application to the crop.

2.6 Mitigation of reactive nitrogen emissions from combustion sources

It is important to consider the impact of non-agriculture sectors on the atmospheric reactive N cycle. As illustrated in Table 1.4.2, the primary source of NOx, for example, is not agriculture but from the transport sector, in particular from road and marine transport, and industrial combustion. Therefore, consideration of the policies and the effectiveness of any planned and implemented actions across the other major sources of reactive N in the UK is important as major shifts would be expected to have a disruptive influence on the reactive N cycle in the UK.

Road transport

Road transport has recently been emerging as a key source for both greenhouse gases and a wide range of air pollutants (including NO_x) and as a result has been the subject of a number of high-profile national strategies, including the Road to Zero (Department for Transport, 2018), Clean Air Strategy (Defra, 2019), and, more specifically, the 2018 supplement to the Air quality plan for nitrogen dioxide (NO_2) in UK (Defra, 2018). As Table 1.4.2 illustrates, road transport was the largest single source of NOx emissions in 2018.

Road transport NOx forms during engine operation when combustion conditions favour the oxidation of atmospheric N (N_2). The amount of NOx formed during combustion is dependent upon the temperature and pressure within the combustion chamber itself. When temperatures exceed 1300°C, the formation of NOx accelerates rapidly (e.g., Sindhu et al., 2018). Oxygen in the exhaust drives further chemical reactions which can help to reduce NOx in the exhaust to N, water, and CO_2. When concentrations of oxygen in the exhaust are high, these chemical reactions are not favoured and so the emissions rate of NOx are considerably higher. However, if the oxygen levels are reduced through balancing the levels of fuel and air, a number of other harmful pollutants are formed, such as unburned hydrocarbons and carbon monoxide. Petrol engines typically have a good balance of oxygen and a relatively inexpensive three-way catalytic converter can reduce NOx very effectively. Diesel engines, however, operate using a compression ignition design, which uses much more air for combustion meaning that oxygen levels are typically much higher, creating an environment unfavourable for the reduction of NOx. This is why diesel vehicles are known to be principally responsible for the current levels of NOx from road transport.

As a step to mitigate emissions of air pollutants, including NOx, from road transport, the European Union introduced EURO standards, requiring new vehicles to meet emissions standards (on a g/km basis) for a range of pollutants, including NOx. These were first introduced in 1993, and several iterations have set more stringent standards ever since. Whilst at first, vehicle manufacturers were able to meet these emission standards through simple engine efficiency improvements, more recently the requirements have become more difficult, and as a result, technologies have emerged which are able to significantly reduce air pollution.

The two primary technology types are exhaust gas recirculation (EGR) and selective catalyst reduction (SCR).

SCR technology is particularly important for the N cycle as it consumes aqueous urea, a diluted form of ammonia. As NOx emissions are emitted through the tailpipe, they are combined with aqueous urea, a reducing agent, in the presence of a catalyst where it reacts to form atmospheric N and water vapour. They are an effective tool to meet the requirements of more recent engine standards: when operating at full efficiency and with a suitable catalyst, SCR systems are capable of reducing NOx emissions by over 90% (e.g. Selleri et al., 2021). However, as a urea-based solution, SCR has the potential to influence other aspects of the N cycle. For example, if engines do not inject enough aqueous urea, NOx emissions are significantly increased, or if too much is added then pure ammonia is emitted through the tailpipe. Automated injection systems that monitor the flux of effluent gas through the tailpipe mitigate this to a large extent and reduce the impact of this consideration significantly. Another consideration is the manufacture of aqueous urea itself. The ammonia component of urea is synthesized from hydrogen (typically from natural gas) and from atmospheric N. Whilst ammonia leakage and urea manufacturing industries, are a potential source of additional reactive N to the atmospheric, these amounts are considered to be insignificant in comparison to the major sources of ammonia. The effectiveness of SCR is also temperature dependent, requiring temperatures to reach a minimum of 200°C in order to be fully operational. Before this point, the reaction is slower meaning that substantial amounts of NOx can be released. These conditions are common when an engine is cold and can take several minutes to fully warm up. However, as above, the impact of cold start emissions is secondary in comparison to the overall savings offered by SCR.

The UK's 2018 Road to Zero Strategy includes strategies to retrofit buses and HGVs to convert them to using technologies compliant with the latest EURO standards, particularly SCR. As highlighted above, the net result of this would be to reduce emissions of NOx considerably, with the potential for an insignificant increase in NH_3 emissions. In addition, other fiscal measures, such as the HGV Road User Levy are aimed at incentivizing the uptake of low emission technologies, reducing NOx and improving air quality in urban areas in particular.

More recent focus on decarbonization in the road transport sector has led to the continued penetration of electric vehicles into the road fleet. As electric vehicles do not burn fuel, they do not emit NOx, N_2O, or NH_3 from their tailpipes. Of course, if electricity is generated using fuels that generate high levels of reactive N species (more typically N_2O and NOx) then this is not a net saving of emissions. However, investment in renewables and natural gas over recent years mean that the emissions of reactive N from the electricity generation has been decreasing in kind. Certainly, on a localized scale, the introduction of electric vehicles will offer substantial emissions savings. A number of schemes and financial incentives are announced within Road to Zero which aim to accelerate the adoption of ultra-low emission vehicles including:

- The overarching ambition to end the sale of new conventional petrol and diesel cars and vans by 2040;
- Binding targets for the market share of low-carbon fuels, with requirements to reach 6.7% market share by 2032;

- Electricity used to recharge plug-in vehicles is charged with a reduced rate of VAT, at 5% compared to the standard rate of 20%;
- No benefit-in-kind liability for electricity provided to charge employees' own electric vehicles.

Table 2.6.1 illustrates emissions projections reported to EIONET by the NAEI on behalf of Defra. They are based on the integration of all existing measures to encourage the uptake of low-emission vehicles and the continued adoption of vehicles satisfying more stringent EURO emissions standards.

Table 2.6.1 – Emissions projections for air pollutants (NAEI, 2020). Note no equivalent data is available for N_2O

Pollutant / Sector	2018[8]	2020	2025	2030	2018 – 2020 trend
NOx	259	214	130	80	-69%
Cars	134	116	81	52	-61%
Vans	92	76	40	21	-77%
HGVs	32	22	8.7	6.1	-81%
Motorcycles	0.50	0.41	0.18	0.12	-75%
NH_3	4.4	4.5	4.8	4.9	+11%
Cars	3.8	3.7	3.9	3.9	+3%
Vans	0.34	0.45	0.60	0.68	+103%
HGVs	0.29	0.29	0.29	0.29	+1%
Motorcycles	0.01	0.01	0.01	0.01	+6%

Marine transport

Domestic and international shipping combined emit more NOx per year than the entirety of the road transport sector. In the case of international shipping, these emissions will not occur directly over land, and so their contribution to the UK's land-based N cycle is dependent on the dispersion of pollutants once they have left the vessel tailpipe. However, there is evidence demonstrating that heavily eutrophicated seas can arise in areas of intense shipping (e.g. Raudsepp et al., 2019) can lead to increases in primary productivity. Atmospheric N deposition to the sea can alleviate the nutrient limitations, particularly of Nand phosphorus, which can drive blooms of plankton. For example, in the Baltic Sea, whilst shipping itself is estimated to only contribute about 1.3-3.3% of the total N input, the impact on a number of biogeochemical variables can reach about 10%. This indicates that N is being either dissolved in undersaturated waters, or fixed by cyanobacteria. Cyanobacteria blooms have a number of detrimental effects that would affect marine ecosystems relied upon by the UK fishing sector.

[8] Note that whilst 2019 inventory data was published earlier in 2021, projections using the revised emissions totals are not available

Algal blooms limit light penetration whilst creating hypoxic near-surface conditions and locally increase the acidity of water, all of which would directly impact the diversity of ecosystems in the area.

The main mechanisms for the formation of NOx from shipping are the same as for road transport: the oxidation of atmospheric N at high temperatures and pressure, which can be reduced in the presence of oxygen in the exhaust. The dominant fuels used in shipping are heavy fuel oil and marine diesel and gas oils. Engines using these fuels operate in a similar way to road transport diesel engines, and so measures to mitigate the air pollution impact of shipping largely align in nature to those outlined in road transport, namely, improvements to engine efficiency and the use of alternative fuels where the formation of NOx is not favoured. As with road transport, the emissions of N_2O and NH_3 from shipping have historically been considered of less importance and so regulation and policies do not typically incorporate these pollutants explicitly.

On an international scale, ship pollution standards are regulated by the International Convention on the Prevention of Pollution from Ships (MARPOL) by the International Maritime Organization (IMO), an agency of the UN. The mechanism to reduce emissions of NOx from ships draws parallels to the EURO standards of road transport, in that ships constructed after certain dates must not emit more than a limit, which is progressively made more stringent over time. The most recent limit introduced is known as Tier III, introduced in 2016. These limits are approximately 5-times stricter in comparison to the Tier I standards first implemented in 2000. Tier III standards only apply to certain regions of the ocean, known as emission control areas (ECAs). ECAs have been established in the English Channel and the North Sea, but until recently, these have only regulated the emissions of SOx and particulate matter. The scope of these ECAs was expanded to include NOx and was due to come into effect in January 2021. Winnes et al. (2015) assessed the potential effectiveness of this measure to reduce NOx emissions in the English Channel, the North Sea, and the Baltic Sea (another new NOx emission control area). Against a baseline scenario, their modelling suggested a 29% emissions reduction by 2030, and a 67% reduction by 2040. It is expected that in order to comply with the regulations, ships that operate in these regions would need to adopt similar solutions to those developed for the road transport sector, namely the retrofit of EGR or SCR technologies, or adopt new fuels which do not emit as much NOx. The most advanced solution is the uptake of liquefied natural gas (LNG), which typically is installed in a dual-fuel system, where a small amount of fuel oil is used to aid ignition, but the majority of energy is derived from LNG for the remainder of the voyage. Whilst NOx emission factors for LNG are not well established, it is widely agreed that the use of this fuel would bring considerable emissions savings. Current NOx emission factors for fuel oil and diesel oil are approximately 1,950kg per net energy consumed (EMEP/EEA Guidebook, 2019), whilst initial estimates for LNG indicate a value closer to 150kg per net energy consumed (Sharafian *et al.*, 2019). The rate of turnover in the vessel fleet, however, is very slow and so the impact of the adoption of new fuels is buffered by the preference to retrofit existing vessels.

Domestically, focus on the air pollutant and greenhouse gas contributions of the marine sector have only emerged more recently and all UK-focused actions in the marine sector are at an early stage of adoption. The first UK port to adopt its own air quality strategy was the Port of London Authority, which did so in 2018. Following its publication, in 2019, the Department for Transport published requirements for all major ports to develop and publish a robust air quality strategy by the end of 2020, which would include actions, targets, and baseline emissions inventory estimates, and the development of metric and monitoring capacity to track progress towards the targets. Common actions highlighted in these port air quality strategies include: the development of shore-side power infrastructure, allowing a vessel to power its auxiliary functions with electricity rather than its own engines; ways to reduce emissions from off-road machinery, through the adoption of new models that satisfy more stringent environmental standards or are electricity-powered; and offering discounts/incentives for the use of cleaner fuels and propulsion technology.

In addition to this, in 2019, the Department for Transport published its Clean Maritime Plan, and Maritime 2050 which outlines the UK's Government's support for moving towards zero emission shipping by 2050 (Department of Transport, 2019a, 2019b). The actions outlined in the policy document will set the basis of further legislative action from the Government, developing a broader understanding of what actions would be most effective at reducing emissions of GHGs and air pollutants, such as the inclusion of international shipping emissions within the country-wide carbon budgets.

As with road transport, specific emissions projections for individual actions are not available in the policy documents and have not been published as evidence to support the proposals, and NAEI projections do not provide sufficient granularity to isolate projections of NOx and NH_3 emissions from shipping alone. Prior to the UK's exit from the EU, Government reported its planned policies and measures to the European Environment Agency annually (NAEI, 2020).

In the longer term, solutions to the IMO's ambition of reducing greenhouse gas emissions by 50% in 2050 compared to 2008 values are being considered. For this, it is widely agreed that the adoption of non-fossil fuels will be required. It is not viable for large vessels, such as containerships, to adopt battery-electric technology in its current state. For the shipping sector in particular, other fossil-free solutions are being explored and a major candidate for long-term adoption is the direct combustion of ammonia. There is currently no favoured mechanism for how ammonia could be used for propulsion but these the primary candidates are:

- Direct combustion in an internal combustion engine;
- Direct combustion in a gas turbine;
- Indirectly as a 'hydrogen carrier' for a hydrogen fuel cell system;
- Chemical reaction of ammonia in a solid oxide fuel cell system.

If either of the fuel cell systems reach technology-readiness and become the favoured mechanism for enhancing ammonia's energy, then there will be no NOx tailpipe emissions associated with its use. Ammonia synthesis uses unreactive atmospheric N and hydrogen, and so its use as a fuel would not disrupt the N cycle directly, although insignificant amounts of leakage at production facilities may occur. If combustion occurs in either an internal combustion or a gas turbine engine, then there are some additional tailpipe emissions which need to be accounted for. Ash and Scarbrough (2019) suggest that the NOx emissions associated with the direct combustion of ammonia would be sufficient to require in installation of SCR technology in order to meet the restrictions imposed in ECAs. It is unlikely, however, that significant amounts of ammonia will be used as marine fuel even in the long-term (e.g., Lloyds Register, 2019). Instead, biofuels are considered the most promising fuel for shipping. Again, whilst the emission factors depend on a variety of factors and the combustion conditions in engines, unmitigated NOx emissions levels may be too great for ECAs and require the use of after treatment technologies such as SCR and EGR.

Industrial combustion and power stations

Emissions of air pollutants for large installations, including industrial sites and power stations, have historically been regulated through the Industrial Emissions Directive (IED), an EU Directive which came into force in 2011. The Directive uses a polluter-pays principle to regulate emissions, and sets benchmarks on a sectoral basis, establishing best practices and the latest emission control techniques for each, known as Best Available Techniques (BAT). A reference document of BATs can then be used by EU Member States to require the adoption of alternative techniques and technologies by industrial sites in their countries to reduce emissions of key air pollutants including NOx. In parallel to this, decarbonization efforts driven by UK national policy to reach GHG emissions targets and other EU policy (such as in the Emissions Trading Scheme; ETS) have caused a recent shift away from fuels that would otherwise emit significant amounts of GHGs and air pollutants. Most notably, the reduction in coal-firing in the UK, and increased use of natural gas continues to reduce emissions from the energy generation sector (e.g., Smith et al., 2020). Historic declines since 2007 in NOx emissions have also been driven by the use of Boosted Over-Fire Air (BOFA) abatement systems. The UK has adopted these industrial pollution laws into its own legislation and they continue to be in force after the UK's EU Exit, including the IED, ETS, and a UK version of the European Pollutant Release and Transfer Register (E-PRTR), and the Medium Combustion Plant (MCP) Directive. Domestic regulation includes the Industrial Decarbonisation and Energy Efficiency Roadmaps to 2050[9], which identifies potential pathways for many of the UK's most polluting sectors to reduce their GHG emissions in line with overall net zero ambitions. As with other forms of decarbonization, if appropriate alternative fuels and abatement after treatment technologies are used, these changes should also reduce emissions of reactive N species.

The package of policies reported by the UK on industrial decarbonization and air pollution mitigation to the European Environment Agency estimate an emissions saving of between 43kt and 52kt NOx per year by 2030, which is between 16% and 19% of the latest 2019 inventory figures for energy generation and industry.

In addition, the general shift away from the combustion of fossil fuels and towards renewables in power generation, in light of the UK's climate and GHG policies, will inevitably bring substantial emissions reductions. The UK Renewable Energy Roadmap (DECC, 2011) outlined the Government's intentions to increase renewable capacity by 2020 in response to EU-wide policy. In 2019, data shows that renewables contributed 35% to the overall electricity generated (BEIS, 2020a), and it seems likely this is only set to increase as the UK ramps up its commitment to the Paris Agreement. Therefore, emissions of NOx from the industrial and power generation sectors are set to reduce significantly over the coming decades.

[9]https://www.gov.uk/government/publications/industrial-decarbonisation-and-energy-efficiency-roadmaps-to-2050

3. Identifying the policy/regulatory frameworks in a four-country context

3.1 International and national progress with N Policy

3.1.1 International Developments on tackling N pollution

Sutton et al. (2021) show that despite multiple nitrogen-relevant UN agencies and conventions since the 1st Earth Summit in 1972 (that led to the establishment of UNEP), global N waste has steadily increased, tripling in magnitude over the last five decades (Figure 3.1.1). The figure shows the sequences of major UN agreements covering the main environmental threats of N, namely: water resources, air pollution, climate change, biodiversity and stratospheric ozone depletion.

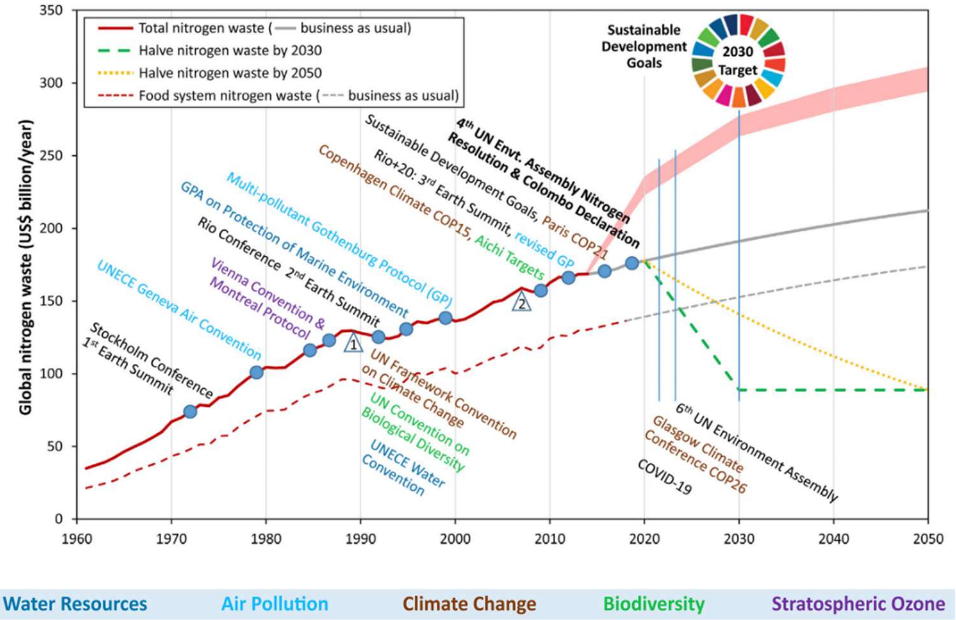

Figure 3.1.1 Sequence of major UN agreements relevant to nitrogen pollution in relation to past and possible future nitrogen waste (Sutton et al. 2021).

The 1972 conference recommended ''monitoring the environmental levels resulting from emission of carbon dioxide, sulfur dioxide, oxidants, [and] nitrogen oxides (NOx).'' Since then, carbon dioxide has become synonymous with climate change, and sulfur dioxide with acid rain but to achieve the SDGs and planetary boundaries, N also requires action (Sutton et al. 2021). The Colombo Declaration (UNEP, 2019) agreed on a goal of halving N waste by 2030 through National Nitrogen Action Plans and also endorsed the UNEP Road Map for the UNEA-4 Resolution on Sustainable Nitrogen Management. Halving global nitrogen waste benefits all players since reducing N waste enables available Nr resources to go further and is more equitable than everyone having to increase NUE because less waste means less action is needed (Sutton et al., 2013; Sutton et al., 2021). At the same time, it allows flexibility for national and local actors to fine-tune action according to their own priorities (by sector, source, nitrogen form, effect, etc.).

As N has impacts that are covered by several different Multilateral Environmental Agreements (MEAs), and to tackle the problem of fragmentation between nitrogen threats across parallel UN activities, a newly established UNEP N Working Group is setting up an Inter-Convention Nitrogen Coordination Mechanism (INCOM). INCOM will be an intergovernmental body under the auspices of UNEP, implementing the UNEA-4 Resolution on Sustainable Nitrogen Management (UNEP/ EA.4/Res.14)). The UNEP N Working group is being supported by the GEF funded Towards an International Nitrogen Management Systems (INMS) project which is currently undertaking an International Nitrogen Assessment (INA), the first at global scale.

3.1.2 European Developments

The European Green Deal outlines how to make Europe the first climate-neutral continent by 2050. It 'maps a new, sustainable and inclusive growth strategy to boost the economy, improve people's health and quality of life, care for nature, and leave no one behind' (European Commission, 2020).

The Farm to Fork Strategy is at the heart of the Green Deal (European Union, 2020a). In the EU Biodiversity Strategy to 2030 (European Union, 2020b) and Farm to Fork Strategy, the European Union has set the ambitious goal of reducing nutrient losses to the environment from fertilizers by at least 50%, and has announced that in 2022 the Commission, with Member States, will develop Integrated Nutrient Management Action Plans (INMAP) "to manage nitrogen and phosphorus better throughout their lifecycle and address nutrient pollution".

The EU Farm-to-Fork Strategy (COM, 2020) recognises the major impacts of excess nutrients (particularly N and P) and sets out the goal of reducing nutrient losses by at least 50%, while ensuring that there is no deterioration in soil fertility. Actions include reducing use of fertilisers by at least 20% by 2030, by implementing relevant environmental and climate legislation, identifying the nutrient load reductions needed to achieve these goals for each member state, applying sustainable nutrient management and managing nitrogen and phosphorus more effectively. All Member States will develop an integrated nutrient management action plan to address nutrient pollution at source and increase the sustainability of the livestock sector. Other actions are the application of precise fertilisation techniques and sustainable agricultural practices, and of recycling of organic waste into renewable fertilisers.

In May 2021 the European Commission adopted the EU Action Plan: "Towards Zero Pollution for Air, Water and Soil" (See press release; EC, 2021) – a key deliverable of the European Green Deal setting out an integrated vision for 2050. The plan unites relevant EU policies to tackle and prevent pollution, including use of digital solutions, and will identify gaps in existing EU legislation.

The Action Plan for the 2050 goal of a healthy planet for healthy people, sets key 2030 targets to reduce pollution at source. Key actions in the plan that related to N pollution include:

- improving air quality to reduce the number of premature deaths caused by air pollution by 55%;
- improving water quality by reducing waste;
- improving soil quality by reducing nutrient losses and chemical pesticides' use by 50%;
- reducing by 25% the EU ecosystems where air pollution threatens biodiversity.

- ➢ significantly reducing waste generation and by 50% residual municipal waste.

The Plan also outlines a number of flagship initiatives and actions, including:

- ➢ aligning the **air quality standards** more closely to the latest recommendations of the World Health Organisation,
- ➢ reviewing the standards for the **quality of water**, including in EU rivers and seas,
- ➢ **reducing soil pollution** and enhancing restoration,
- ➢ reviewing the majority of **EU waste laws** to adapt them to the clean and circular economy principles,
- ➢ fostering **zero pollution from production and consumption**,
- ➢ presenting a Scoreboard of EU regions' green performance to promote **zero pollution across regions**,
- ➢ **reduce health inequalities** caused by the disproportionate share of harmful health impacts now borne by the most vulnerable,
- ➢ **reducing the EU's external pollution footprint** by restricting the export of products and wastes that have harmful, toxic impacts in third countries,
- ➢ launching Living Labs for **green digital solutions and smart zero pollution**,
- ➢ consolidating the **EU's Knowledge Centres for Zero Pollution** and bringing stakeholders together in the Zero Pollution Stakeholder Platform,
- ➢ stronger **enforcement of zero pollution together** with environmental and other authorities (EC, 2021).

3.1.3 Progress in Nordic Countries

The Nordic Council of Ministers report, Nordic Nitrogen and Agriculture (Hellsten et al., 2017), summarises sources, pathways and impacts of reactive nitrogen in the Nordic countries and policy efforts to control Nr. While Nordic countries have introduced measures to reduce N waste, N losses are relatively high as compared to the policy targets set.

The Nordic report lays out the current N policy controls used in Nordic countries (Table 3.1.1) and, in agreement with Bechmann et al. (2016), they note that, although there are many commonalities in measures in the four Nordic countries (Denmark, Finland, Norway and Sweden), there are large differences between the regulatory frameworks:

- ➢ In Denmark, better utilisation of manure and other regulation has reduced N loading by up to 50%, but the fertiliser system is expensive and allows low flexibility to farmers;
- ➢ The Swedish advisory programme "Focus on nutrients" has been effective in reducing N losses through farm visits allowing knowledge transfer and flexibility in application;
- ➢ The Finnish "Agri-Environment program" payment system has succeeded in recruiting farmers and has reduced especially phosphorus loadings from fields;

- In Norway, the legislation on manure management, the Regional Environmental Programme and subsidies for environmental investments motivates farmers to implement measures, mainly regarding phosphorus losses.

The Nordic report highlights that both stringent regulations (Denmark), and voluntary and advisory efforts (Sweden) have been successful in reducing N losses from agriculture, but that in Denmark the complexity of the regulations is becoming too high for the farmers to accept. There is a need to simplify the regulations, but still obtain the same level of environmental benefit through other channels.

Table 3.1.1 Summary of Current Policy Controls in Nordic Countries (Hellsten et al., 2017)

Information and counselling	Rules and regulations	Investment support
Sweden		
The advisory Program "Focus on nutrients" (Greppa näringen") focuses on increasing nutrient management efficiency by increasing awareness and knowledge. The campaign is characterised by voluntary participation, farm specific measures, repeated farm visits and follow-up on each farm.	Regulations regarding the spreading, storing and use of manure consider regional differences. All livestock farms must have sufficient manure storage. In southern Sweden requirements for coverage of slurry and urine tanks apply. Regional, specific rules for when manure spreading should occur, and how quickly the manure should be incorporated into the soil apply. The sensitive areas also have restrictions on type of spreading techniques. In southern Sweden, 50–60% of arable land shall be under vegetative cover during the autumn and winter.	Environmental support schemes: -Cultivation of ley -Catch crops, spring cultivation -Riparian buffer zones -Maintenance of ponds and wetlands Environmental investments: -Construction of wetlands -Different investments for improved water quality -Two step ditch -Controlled drainage
Norway		
The agricultural advisory service conveys information on best management practices including reduced tillage, grassed buffers and waterways and sedimentation ponds. They are also the main actor making fertiliser plans.	The rules and regulations consist of production grants requiring fertiliser and pesticide plans; national and regional environmental programmes with subsidies for measures to reduce nutrient losses and rules for when, how and where to spread manure.	Environmental support schemes: -Tillage in spring instead of autumn -Reduced tillage -Catch crops, spring cultivation -Riparian buffer zones -Maintenance of ponds and wetlands Environmental investments: -Construction of sedimentation ponds -Hydrotechnical installations to reduce surface runoff -Manure storage

Information and counselling	Rules and regulations	Investment support
Denmark		
Information on manure handling, animal housing and optimized feed practice. Information regarding fertiliser norms and buffer zones (restrictions) near sensitive areas. Program for catchment advisors (oplands-konsulenter), to promote, suggest and facilitate local solutions to meet requirements for catchment based N leaching reductions from agriculture. Information about organic farming and other types of environmentally friendly systems: both at the consumer side (e.g. for branding and local food chains), and at the producer side (e.g. for new types of manufacturing and about conversion).	Rules for storage of slurry and manure (e.g. minimum storage capacity, no runoff from manure heaps and mandatory slurry tank floating barriers). Rules for how and when manure spreading should occur (e.g. broadcasting banned, and ban on winter spreading of slurry for spring crops). Mandatory fertiliser and crop rotation plans (min. proportion of area with winter crops and catch crops). Ban on autumn soil tillage before spring crops. Voluntary buffer zones around sensitive areas.	Subsidies to: -Promote better manure handling and animal housing (BAT) -Establish strategic wetlands -Low-N grasslands in environmentally sensitive areas -More organic farming, extensification and afforestation
Finland		
Many environmental projects, e.g. - TEHO, TEHO+ (2008-2014; http://www.ymparisto.fi/tehoplus) - JÄRKI (2009 -; www.jarki.fi) - LOHKO, LOHKO II (2015 -; www.mtk.fi/lohko) - Supported advisory services (Råd 2020; http://www.mavi.fi/sv/stod-och-service/radgivare/neuvo2020/Sidor/default.aspx)	Regulations regarding the spreading, storing and use of manure or organic fertiliser products. Larger animal units need to have environmental permits. Stricter upper limits for N fertilisation for those who have joined the Finnish Agri-Environmental Programme, otherwise according to the Nitrates Directive. Also municipal environmental protection regulations (e.g. manure spreading).	- Investment support e.g. for livestock farming investments and for investments that improve the status of the environment e.g. investments to improve the efficiency of manure management, subsurface drainage and controlled subsurface drainage, Support for non-productive investments e.g. construction of wetlands

Policy recommendations for the Nordic countries from the report (Hellsten et al., 2017) include:

- implementing the most cost effective, practical and feasible measures first such as low N feed, covered slurry and manure storages and low ammonia emission spreading techniques;
- extend some current rules and regulation e.g., regarding new livestock houses, and coverage of manure tanks and spreading of manure, slurry and digested manure;
- simplify current farm-regulations;
- scientifically based voluntary actions such as the Swedish advisory program "Focus on nutrients" to be continued and implemented in other countries;
- feedback to farmers regarding the environmental progress (e.g., through the press) to make the farmers proud of their achievements;
- information campaigns about the effects of changed consumption behaviour highlighting the environmental benefits;
- N balances, and the distribution of surplus N to different types of losses, may be more relevant as a basis for policy on large (landscape and regional) scales rather than on a small (field) scale.

The policy challenges and knowledge gaps identified by the Nordic report echo the key themes emerging in this report, including:

Challenges

- The challenge with agri-environmental policies is to decrease negative effects, while at the same time maintaining or increasing food production;
- The importance of a holistic policy approach to assessing technical abatement measures, considering the direct mitigating effect and costs but also other benefits and effects;
- Considering system change measures, e.g., reduction of food waste, in addition to technical measures, increasing efficiency in the food chain, and promotion of consumption patterns with lower N footprints, could further reduce overall N losses;
- Include effect of emissions produced in other countries due to increased import;
- More holistic policy is required for the digestion of manure to produce biogas to replace fossil fuels and associated GHG emissions, as the process may have negative implications for carbon sequestration and lead to lower carbon content in soils if the digestate is not returned into the soils as fertiliser;
- The need to produce more with less through precision farming with modern technology.

Gaps

- identifying overlaps and gaps in existing policies on reactive nitrogen;
- relevant assessment tools and research to find the right balance between potential conflicting interests, including emission savings, other environmental effects, costs, and ethical values;
- understanding of the efficiency of voluntary efforts and advisory actions;
- research on novel approaches to mitigate ammonia, nitrous oxides and nitrate losses from agricultural land;
- evaluation of the balance between targeting of mitigation measures and the transaction costs;
- defining, evaluating and comparing environmental outcomes, their cost-benefit implications and trade-offs, e.g., biodiversity versus water protection or climate mitigation versus water protection targets;
- understanding of how to develop an agriculture-based bioeconomy including integration of environmental protection schemes and a better utilisation of N in the whole production chain.

The Nordic report clearly shows the importance of analysing trade-offs to avoid shifting N emissions from one area into another. However, how to assess such effects and prioritise actions is not always clear. For example, should peat be used during storage of solid manure to reduce ammonia emissions or prioritise reduction of peat extraction? The report suggests that summarising synergies and trade-offs in the absence of an equivalent to CO_2-equivalents, like “nitrogen damage equivalents”, would be useful (e.g. see Table 3.1.2).

Table 3.1.2 Measures which reduce one pollutant but increase another, or is associated with other negative environmental effects (Hellsten et al., 2017)

Measure	Trade-off
Measures to reduce ammonia	Using peat during storage of solid manure is disadvantageous when it comes to climate change effects and other environmental effects of increased peat extraction. Acidification of slurry discourages the development of biogas production, which is even more effective regarding the reduction of greenhouse gases. (However this is not the case when acidification is done in field immediately prior to application.) Air purification may increase emissions of N_2O. Air purification, demands mechanical ventilation rather than natural ventilation, hence demands higher energy consumption. Manure incorporation means higher fuel consumption.
Measures to reduce nitrate leaching	Wetlands may increase emissions of methane and N_2O. Controlled drainages on arable may increase emissions of N_2O. Spring tillage, reduced tillage and catch crops may increase the use of pesticides. No till, spring tillage may increase PO_4 losses with surface run-off. Spring tillage and reduced tillage can increase N_2O emissions (compacting the soil). A more compact soil increase emissions. A good soil structure (to avoid standing water) is important for reduced emissions of N_2O.
Measures to reduce N_2O	Spring tillage, reduced tillage and catch crops may increase the use of pesticides. Structural liming may increase release of CO_2. Improved soil drainage may increase N leaching.
Measures to reduce methane	Some animal feeding strategies can increase N excretion, hence increases ammonia emissions. Active aeration (composting) of stored manure generally increases ammonia emissions.

The Nordic report identifies the need to define, evaluate and compare trade-offs and synergies, but that it is often difficult to measure and include the value of the ecosystem services and goods that do not have an agreed market value, such as the effects on biodiversity, groundwater contamination/protection etc.

There is a need to assess combined effects of measures to reduce pollution to air and water such as the GAINS model (Klimont and Winiwarter, 2015), and the FarmAC/Farm-N models (Dalgaard et al. 2014; 2017). The TargetEconN model developed for the Limfjords and Odense catchments in Denmark (Konrad et al., 2015) and the BALTCOST model for the Baltic Sea (Hasler et al., 2014, applied for both N, P and GHG in Nainggolan et al., 2018) are also such examples.

3.1.4 Denmark

The UK Clean Air Strategy 2019 (Defra, 2019) reports that Denmark reduced ammonia emissions by 40% (Denmark Emissions Inventory, 2017) between 1990 and 2016 through actions including:

- regulating to ensure manure is applied using low-emission spreading equipment (band spreaders or injection), and spreading in winter is limited to certain crops;
- regulating to ensure slurry stores are covered;
- regulating to ensure solid manure must be incorporated into bare soil within 6 hours;
- permitting most farms, requiring a fertilizer plan and adherence to N application limits;
- allocating the majority of their EU funded rural development programme to tackling pollution

- setting nitrogen limits at up to 18% below the economic optimum level;
- limiting the amount of mineral fertiliser available for purchase and recording all purchases automatically on a farm's online fertiliser plan.

The Danish government has been quick to establish ambitious climate goals (like a 70% reduction relative to 1990 by 2030) but slow to actually do anything in relation to agriculture. Given the significance of intensive livestock production for rural employment and the economy, they are looking to technology to provide the solution, so that they will not need to reduce livestock numbers (but they will probably have to). Nitrate leaching and ammonia emission are still serious issues in relation to the Water Framework Directive and the Habitats Directive respectively, plus the health concerns related to nitrates and secondary particulates. As for GHG emissions, the focus has been on technology (within the agricultural area, such as cover cropping, and end of the pipe solutions, such as mini wetlands), rather than taking land out of production (Aarhus University, Pers. Comm.) The use of significant government funding in Denmark to tackle pollution appears to be in conflict with the 'polluter pays' principle and contrasts with the 'public money for public goods' approach being considered in England.

Mahmoud and Hutchings (2020) considered the potential for afforestation of agricultural land to reduce N losses to water and GHGs in Denmark. They conclude that targeting the reduction of one pollutant will also affect the non-targeted pollutant, if high-spatial resolution agricultural and landscape data is available as a basis for targeting.

3.1.5 Netherlands

The Netherlands reduced ammonia emissions by 64% between 1990 to 2016 (Netherlands Emissions Inventory, 2018[10], through actions including:

- regulating to ensure manure is applied using low-emission spreading equipment;
- regulating to ensure slurry stores are covered;
- funding for manure banks to supply arable farms with excess manure and reduce over-application on livestock farms;
- providing financial support for a voluntary industry strategy to develop and install low emission animal housing;
- regulating to ensure that all new housing since 2007 meets low-emission criteria, recognised by the government through a certification scheme;
- providing grants for research into innovative manure management techniques and subsidies, and tax breaks to support investment in the new technologies;
- establishing farmer networks for knowledge transfer and peer-to-peer support.[11]

[10] https://ec.europa.eu/environment/air/pdf/reduction_reports/Report_DK.docx
[11] As reported in the UK Clean Air Strategy 2019

The overall improvement in nutrient management was estimated to cost €500 million annually, but resulted in annual societal benefits of €900- 3,700 million, including €150 million in fertiliser savings for farmers (Van Grinsven,et al., 2016).

Despite a land area that is 270 times smaller than the US, the Netherlands is second in the world (after the US) for agricultural exports with yields and production per hectare the highest in the world, especially for onions, potatoes, seeds, and cheese. Since the 1980s, it is also the highest European N hotspot and agriculture is the dominant source, contributing 46% of N deposition (Erisman 2021).

In 2015, the Dutch government introduced the Programmatic Approach on Nitrogen (PAS), a licensing system that enabled businesses to emit N by compensating with technical measures, e.g., air scrubbers, or natural ones, such as extra mowing, that might deliver emission reductions in the future. However, in 2018, the European Court of Justice ruled that the PAS was not sufficient to protect Natura 2000 sites from N pollution and the scheme was frozen, stopping 18,000 projects and taking N rights away from thousands of farmers. As a result, the Dutch Council of State decreed that there should be proof with scientific confidence that N emissions would indeed decrease (INI Nitrogen Alerts, 2020). Until a reduction in nitrogen emissions could be proved, all nitrogen producing industries had to pause and ways had to be found to allow economic activity related to nitrogen emissions to proceed. The building of houses for instance, was not possible, because it would produce additional N. A measure was introduced to bring the motorway speed down from 130km to 100km. The N saved through this enabled the building of 75, 000 houses in the Netherlands.

Ministers from relevant departments introduced a list of measures to reduce N emissions, to provide another means of building houses and other activities. One of the measures for agriculture was buying up the production rights of about 400 pig farms and taking them out of production. This is a voluntary measure and there is €350 million to compensate the sector, so this measure has not been as controversial, but another measure to reduce the amount of protein in dairy feed is compulsory. As for the pig farms, farmers want this on a voluntary basis, which is not possible as there needs to be certainty in emission reductions, to comply with the Habitats Directive. Farmers argue that low protein can damage animal welfare and their livelihoods (INI Nitrogen Alerts, 2020). The agriculture sector considered that it had been made responsible for solving the problems for the rest of the Netherlands and this led to the massive demonstrations covered in the media (cf. Schaart, 2019). Farmers felt that they have to implement measures to reduce N, so that other sectors like the building of housing and roads can continue. In their view, the benefits from a reduction in N emissions from agriculture should be put back into developing the agricultural sector. The farming sector has also contested the validity of RIVM's data, which shows the agriculture sector contributes to 46% of N deposition. The effect on N deposition of reducing the amount of protein allowed in feed has also been estimated by RIVM, and so this has become another contentious issue, causing demonstrations to be targeted at RIVM. Subsequently, the protein measure was dropped by the Dutch Government.

In March 2021, the Dutch parliament passed a law requiring the country's N emissions to be cut in half over the following ten years, so that the quantity of N that is deposited on natural reserves must be lowered rapidly to comply with EU law. Recently, the Dutch Environmental Assessment Agency (PBL) has warned that adopting the new nitrogen policy to achieve climate protection goals would make agriculture and livestock farming impossible in a number of

Dutch provinces, warning "[a] choice for these stricter goals means an unprecedented transformation of the rural area in the Netherlands" (NL Times, 2021). A recent report by Erisman et al. (2021), entitled 'What is the road to a relaxed Netherlands?', has further unsettled the farming community as it recommends a targeted approach to N abatement which some farmers think would put them out of business (cf., Beeks, 2021).

Nitrogen oxide reductions of 50% are possible through actions such as increased energy efficiency, electrification of transport, and sustainable energy, but it is much harder to abate agricultural emissions. Erisman (2021) suggests that governments must play a major role in driving reductions by promoting a positive vision for agriculture and introducing principles such as soil health as the basis for sustainability and spatial planning to help reduce the impact of food production on the environment and the climate. In addition, they should set legal environmental targets within which farmers may practice as they choose, but in instances where targets are not met, the government should intervene. These targets should be concrete, science-based, and centred on environmental resources: healthy soil, air, and water; a stable climate; the conservation of biodiversity; and the protection of nature, landscapes, and animal welfare. The targets could be regional but should be accountable at the farm level and enforced by regional authorities. Clear outcomes such as key performance indicators (KPIs) can be used to steer practice toward targets. An example of a KPI system is the Biodiversity Monitor developed by the World Wildlife Fund, Rabobank, and Friesland Campina for dairy farming (van Laarhoven, *et al.*, 2018). Where successful actions can be measured unambiguously, farmers could be rewarded through direct payments, interest rebates on loans, a higher price through customers, or lower taxes.

3.1.6 France

The French National Assembly and the Senate adopted a new Climate and Resilience law in July 2021 (Assemblée Nationale, 2021), which includes the reduction of the use of N-based fertilisers and encourages more vegetarian menus in school canteens (cf. Pistorius, 2021). The law includes measures to reduce the use of mineral N fertilisers in a bid to lower nitrous oxide emissions by 15% of 2015 levels by 2030 and ammonia emissions by 13% compared to 2005 levels over the same period. If the emission reduction targets are not met for two consecutive years the law calls for the introduction of a levy on the use of the fertilisers in question. The law attempts to ensure the economic viability of the agricultural sectors concerned and not increase possible distortions of competition with the measures in force in other EU member states. It also recommends the preservation and planting of hedges and trees between agricultural plots to store carbon, combat soil erosion and improve water quality. The law also calls for the recognition and better valuation of "positive externalities of agriculture, particularly in terms of environmental services and land use planning." There were farmer protests against the new law earlier in the year which was called a "punitive and unfair" N royalty (cf. Martin, 2021). The largest farmer's union in France (FNSEA) warned that the new law stigmatises the use of synthetic fertilizer without providing alternatives and does not recognize work carried out by both public and private research bodies that has significantly improved fertilization management over the past few decades (through soil assessment, decision-making tools, and providing information on choice selection, etc.). At the same time, low emission slurry spreading technology has improved, making it possible to reduce the share of mineral fertilisers as well as N surpluses. The situation demonstrates the potential problems if such measures were introduced in the UK and suggests the UK can potentially learn from attempts to regulate fertilizer use across Europe.

In terms of the potential future developments to control N emissions in the UK recommended in this report, related to fair distribution of responsibility across the supply chain and tackling unnecessary N related trade of products and feedstocks, there are some interesting parallels with the situation in France. Van der Ploeg (2020) describes a survey of French farmers asking them what they considered to be the most 'threatening developments' to agriculture in France. Forty-four per cent of the surveyed farmers referred to climate change, 32% pointed to 'the market, low prices and volatility' and 31% to 'agri-bashing' of various types. Another set of questions probed the best possible policy to secure the future of French agriculture. Forty-eight per cent argued that "the power balance between farmers, food industries and large retailers needed to be corrected and the same proportion said that food distribution (and consumption) needs to be grounded on the principle of proximity, while 44% argued that food quality and security are the main areas to apply leverage" Van der Ploeg (2020) concludes that 'business as usual' is not very attractive or feasible for most French farmers, as the survey also revealed low preferences for the standard repertoire of solutions such as conquering new markets (13%), applying new technologies (13%) or scale enlargement of farm enterprises (9%).

3.1.7 Germany

In 2017, the German Federal Government drew attention to the problems of excessive release of reactive nitrogen into the environment by agricultural production, energy conversion, and mobility in its first Nitrogen Report and established the need for inter-departmental action. The German Environment Agency (UBA) launched a number of projects including the DESTINO project (DESTINO Report 1, Heldstab, et al., 2020) which has two objectives: firstly, to derive an integrated N indicator across all sectors together with a national N target (Report 1), and secondly to update the National Nitrogen Budget in line with the requirements of the Gothenburg Protocol (Report 2).

Report 1 of the DESTINO project documents the process of deriving the integrated N indicator for N-sensitive environmental sectors: Maintaining biodiversity, avoiding eutrophication of ecosystems, preserving the quality of groundwater, surface waters, and air, and meeting climate action objectives. The national N target quantifies the limits which must not be exceeded if the objectives are to be met.

A National Nitrogen Target for Germany

The anthropogenic N cycle is a highly complex process with different reactive nitrogen species (NH_3, NH_4^+, NO, NO_2, NO_3^-, and N_2O) released, causing numerous negative impacts on the environment (Heldstab, et al., 2020). To overcome the problems of communicating this complexity and to enhance policy action, Geupel et al. (2021) developed a new, impact-based integrated national target for N (INTN) for Germany. The basic approach is to calculate a maximum permitted N loss per year, on the national level, for each impact indicator, such that related quality targets (referred to as state indicators), are met in Germany at the spatial average. Where values for maximum loss rates are available from current legislation, they were adopted directly as target values. Using the six impact indicators (Table 3.1.3), the maximum loss rates for reactive N species, such as nitrate (NO_3^-), nitrous oxide (N_2O), nitrogen oxide (NOx), ammonia (NH_3), and total nitrogen (N total) were obtained. The resulting target sets a limit of N emissions in Germany of 1053 Gg N yr^{-1}. Taking related uncertainties into account, the resulting integrated N target of 1053 Gg N yr^{-1} suggests a comprehensible INTN of 1000 Gg N yr^{-1} for Germany, meaning the overall annual loss of reactive N in Germany would have to be reduced by approximately one-third.

Table 3.1.3 Overview of selected impact indicators and related state indicators to calculate the national nitrogen target as the sum of the related maximum permitted nitrogen losses per pressure indicator (Geupel et al., 2021)

Number	Impact Indicator	State Indicator	Pressure Indicator (Nitrogen Loss Rate)
(1)	Vegetation affected by ambient NH_3 concentration	NH_3 critical level for higher plants: 3 µg m^{-3} NH_3 [22]	NH_3 emissions
(2)	Terrestrial ecosystems affected by eutrophication (deposition)	35% reduction of exceedance of the Critical Load for eutrophication from 2005–2030 [23,24]	NH_3 and NO_x emissions
(3)	Surface water quality (to prevent coastal water from eutrophication)	N_{total} concentration to protect North Sea (2.8 mg N l^{-1}) and Baltic Sea: (2.6 mg N l^{-1}) [25]	N_{total} load
(4)	Groundwater quality affected by nitrate	NO_3 concentration in groundwater: 50 mg l^{-1} [26]	NO_3 leaching
(5)	Nitrous oxide emissions affecting climate change	N_2O emission: long-term goal reduction 80–95% [27]	N_2O emissions
(6)	Human health affected by atmospheric NO_2	NO_2 concentration: WHO effects level for the background: 20 µg m^{-3} [28]	NO_x emissions

Acknowledging the differences in calculation of the selected six different approaches, Geupel et al. (2021) are confident that integrating their results to the INTN is an acceptable simplification, comparable to the planetary boundary for N fixation by de Vries et al., 2020 and are confident that the result is sufficiently robust for the purpose of an additional element for political communication. Additionally, whereas five of their calculated maximum permissible N loss rates can be related to emission sources directly, such a direct relationship to emission sources for the N load to surface waters cannot be established. This suggests that the resulting INTN has to be interpreted as an interim target and that further in-depth assessments would lead to an even lower target value.

For human health effects of nitrogenous air pollutants, Geupel et al. (2021) focused only on the direct effects by NO_2. They chose to exclude indirect effects on ozone and particulates as these are also driven by many other factors and a sound mathematical relationship to N emissions was not possible (see also Section 1.1.2). However, by choosing a low-value background concentration of 20 µg m^{-3}, the indicator is precaution-oriented, so that maximum NOx emissions also improve human health exposure through those indirect effects.

The national N target is composed of independent targets, designed so that related state indicators or quality targets are met on a spatial average in Germany. Therefore, reaching the national N target does not guarantee that the six state indicators considered as spatial-dependent functions are reached everywhere. The compliance with the N loss rate is therefore a necessary condition but is not sufficient to reach the environmental state indicators universally and they recommend that neither the indicators nor the calculated national N target should fully replace existing indicators based on, for example, spatially resolved monitoring networks or detailed modelling approaches.

Geupel et al. (2021) calculated maximum permissible N_2O emissions based on targets defined in the national Climate Action Plan 2050 of the German Federal Environment Ministry and the reported greenhouse gas emissions for 2017. As a long-term objective for 2050, the Action Plan defines a reduction of total greenhouse gas emissions in Germany (in CO_2 equivalents) by 80–95% as compared to 1990 and a corresponding interim reduction target for 2030 of 55%.

The action plan does, however, not define a specific target for nitrous oxide emissions, in common with the UK's Sixth Carbon budget (see Section 1.5). Therefore, Geupel et al. (2021) used the existing sectoral targets for 2030 to derive a target for nitrous oxide emissions in

2050. For N_2O emissions from the energy sector, an average reduction in line with the long-term target for total greenhouse gas emissions is assumed (–87%, as compared to 1990). For N_2O emissions from the waste sector, they assumed that the interim target corresponds to the long-term target (–87% as compared to 1990). For N_2O emissions from agriculture, the interim target for 2030 (32.5%) was extrapolated linearly to 2050, resulting in a necessary reduction of 40% compared to 1990. Comparing these three sectoral reduction targets to the N_2O emission situation in 2015 provides a target for maximum allowable N_2O emissions, which Geupel et al. (2021) included in the national nitrogen target. Since in the industrial sector nitrous oxide emissions were already reduced by 95% between 1990 and 2015, they assumed that the emissions remain at the level of 2015 in this sector.

3.2 The role and design of fiscal measures for N

Fiscal measures such as taxes, levies, incentives, subsidies or emissions trading scheme are used to drive behaviour change, influence the state of markets and the economy, generate revenue or achieve certain goals, including to benefit the environment or society. In the context of reducing N waste and pollution, fiscal measures can help to internalise the costs to public and private actors of the impacts of N pollution on people's health, the climate, air and water quality, biodiversity and ecosystems. However, fiscal measures are politically sensitive and taxes on artificial N fertilisers proposed in other countries have been highly controversial and perceived as punitive and unfair by farmers (e.g., see 3.1.6).

A UNECE guidance document on economic instruments states that they can give more flexibility in deciding on a response than control mechanisms and, if designed well, can lead to lasting behavioural change, technical innovation and cost-effective pollution control (UNECE, 2013b). An effective fiscal system must be clear about its purpose, e.g., environmental improvement, revenue generation, protecting those who suffer health or financial penalties from pollution. Some of these goals can work together, such as benefits to health and environment, but some may be incompatible such as environment and revenue generation if they are not carefully designed. The guidance document describes lessons learnt from the use of economic instruments to reduce NOx, sulphur, VOCs, NH_3 and particulate matter. Four types of economic instruments are identified as the most relevant to these pollutants: tradable permits and quotas; emission and process taxes/charges; product charges and tax differentiation; and subsidies and fiscal facilities (UNECE, 2013b, p. 2).

At the time of the writing of the UNECE (2013b) guidance document, there had been implementation of the European Union (EU) greenhouse gas emission allowance trading scheme in 2005 and national emission trading schemes for carbon dioxide (CO_2) in several countries. Other actions include trading schemes for pollutants such as NOx, e.g., in the Netherlands, emission charges for NOx and SO_2, and national carbon taxes and carbon-related incentives. See Tables in UNECE (2013b) for a summary of the types of incentives and their relative success in different countries. Exemplary cases include:

- *Tradable permits and quotas:* Emission trading programmes (United States of America); manure quotas (Netherlands);
- *Emission and process taxes/charges*: NOx charge (Sweden; Norway, Denmark); emission taxes in several Central and East European countries;

- *Product taxes and tax differentiation*: VOC incentive tax (Switzerland); sulphur tax (some countries); tax reduction for "cleaner" fuels and cars (several countries);
- *Subsidies and fiscal inducements*: Environmental funds (mainly in Central and Eastern Europe); accelerated depreciation schemes (several countries); price guarantees for renewable energy (Germany).

Sweden introduced a tax on N (NOx) in 1992 which reduced NOx emissions by 30-40%, applying it to energy rather than agriculture. The revenue was used to reimburse taxes to plants emitting low volumes of NOx in order to incentivise energy efficiency and reduce any potentially negative impact on competitiveness which led to many companies implementing emission reduction measures ahead of the introduction of the tax (Anastasio, 2017).

The conclusions of the UNECE (2013b) guidance included:

(a) Economic instruments require an effective market mechanism including competition and access to information on emission reduction options and benefits;

(b) Pollution taxes or tradable permits should be embedded with other measures such as standards, policy, legal frameworks for sanctions and judicial action, and voluntary agreements that are mutually reinforcing and not contradictory;

(c) Fiscal measures can be incentives paid for abatement directly or indirectly through the market with cost increases passed on to end users. Corporate tax deductions are only effective for profit-making enterprises;

(d) Preferably, any measures should be announced well in advance and involve consultations with stakeholders to allow producers and consumers to plan their investment decisions in an altered market, but the rollout should be aware of issues of potential stockpiling.

Economic modelling of N tax and N use suggests some important factors to consider and particularly the level of risk aversion in farming communities. While N taxes have been found to reduce N application and N losses to the environment (Rougoor, et al., 2001), economic models of N taxes often focus only on maximising farmers' profits with reduction of N use (cf. Kuhn, et al, 2010; Gandorfer et al., 2011). Such bio-economic models find that risk-averse farmers have lower levels of N application than risk-neutral farmers and N taxes lead to greater reductions of N use if farmers are risk-averse. Finger (2012) argues that this analysis solely based on profit maximizing behaviour may underestimate N reductions. This model takes into account price volatility and yield variation and indicates that a N tax decreases N use, and therefore yields and profits, irrespective of farmers' risk attitudes. A 10%, 20% and 30% N tax would reduce the N use by about 5.01%, 9.65% and 13.95%, respectively. Meyer-Aurich et al. (2020) came to similar conclusions, but their model results show that while moderate N taxes are effective in reducing N fertilizer use at costs below 100 €/t CO_2eq for rye, barley and canola, in wheat production a N tax has limited effects on optimal N use due to the effects on crop quality and the sale prices of wheat.

Henseler et al. (2020) modelled two policy options: a market-based tax on N and a command-and-control set-aside of agricultural land. Their results showed that at global scale, both options create relatively high marginal abatement costs and that the maximal abated greenhouse gas emissions represent only 15% of the quantity required to fulfil the goals of the COP21 policy targets. Compared to the obligatory set-aside option, the N tax was the more efficient policy. For their modelling of Germany, the N tax varied from +20 to +80% increase of N fertiliser price and the abatement of GHG emission ranged from 2 Mt CO_2e to a maximum of around 9–10 MtCO_2e. However, both measures showed a relatively low ecological effectiveness and were less efficient than, for example, the restoration of peat land with a maximum reduction potential of 25 Mt CO_2e at costs below 100 EUR/t CO_2eM. In addition, the N tax brings high relative income loss for intensive crop producing areas, so would require regional adjustment to avoid unacceptable disadvantage for regional producers.

Lungarska & Jayet (2016) modelled the impact of spatial differentiation of N taxes on French farms' compliance costs. Their estimates suggest that realistic regulation via input-based pollution fees should be differentiated in order to significantly reduce the financial burden on farmers of conforming to predefined pollution levels. Some potential adverse effects related to input-based taxation and land use change call for additional fine-scale N pollution regulation (e.g., limitations on crop switching).

Gu et al. (2021) propose a Nitrogen Credit System (NCS) as opposed to a tax. They argue that bearing the cost of N pollution is the responsibility of society, not just farmers, and society as a whole benefits from less N pollution. The agri-food industry and retailers are intermediaries in the food chain and should facilitate the distribution of abatement costs while governments should regulate for clean air and water and healthy soil by establishing pollution standards and driving fair sharing of costs and benefits among farmers, suppliers, processors, retailers, consumers, and financial organizations. An NCS would provide economic incentives (e.g., subsidies based on cost and societal benefit) to farmers to adopt environmentally friendly practices to mitigate N pollution and would require (Figure 3.2.1):

1. certified measures such as limits or caps to abate N pollution for which farmers would earn credits;
2. a budget to subsidize these in proportion to the societal benefit of reduced N pollution as well as abatement costs;
3. administration of the system for granting credits and enforcing compliance with members representing farmers, citizens and consumers, industry, government, and science.

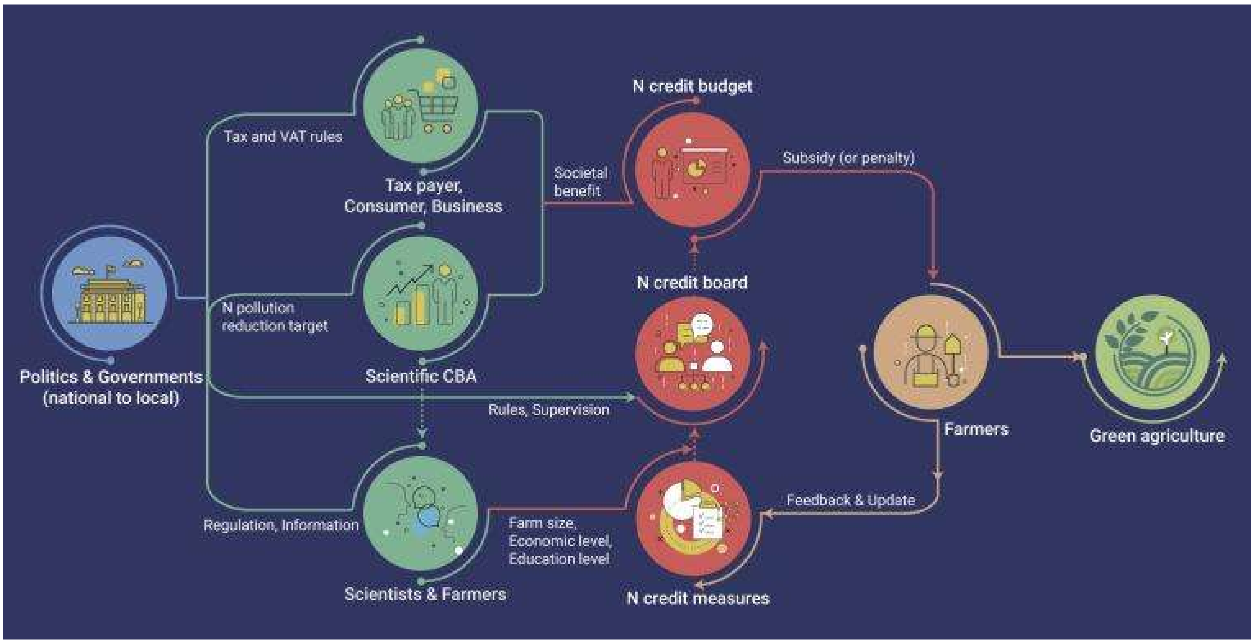

Figure 3.2.1 Framework of a Generic Nitrogen Credit System to Mitigate Global Nitrogen pollution (Gu, et al., 2021)

In the EU and the UK, regulatory approaches such as the "Nitrate Directive" constrain farmers and apply penalties for non-compliance. The NCS could be combined with the current subsidy to include a bottom-up design with all stakeholders actively involved in selection and implementation of mitigation measures which are rewarded based on verifiable results on reduced pollution.

An alternative measure is a levy on N fertilizer use whereby the income generated would be ring-fenced to support the adoption of N-efficient sustainable farming practices that deliver public goods, such as increased biodiversity and improved water and air quality. Such a levy could be applied at the point of sale or according to the area of land to which fertiliser is applied or level of on-farm N surplus. This approach could provide greater incentive and sense of fairness to farmers, while helping to meet the economic, social and environmental costs of the impacts of N pollution.

Private markets for emissions trading and payment for environmental services have already demonstrated success. In England, the private water company Wessex Water has established EnTrade, a business which provides markets for environmental services. It manages contracts with farmers and other land managers to reduce N pollution of watercourses and facilitates nutrient trading between developers and land managers to achieve 'nutrient neutrality' as a condition of planning permission. EnTrade began with a scheme to reduce N loading of Poole Harbour in Dorset, which was incurring high costs for Wessex Water in pollution removal. The business has evolved to support investment in a wide range of environmental services through online markets involving thousands of farmers.

This report recommends that the UK Government commissions an independent economic assessment of the costs and benefits of N pollution, considering options for action. Such an assessment should consider the range of possible fiscal measures in detail and make recommendations to government.

3.3 Nitrogen balance sheets and budgets

3.3.1 Nitrogen balance sheets and budgets

A national nitrogen budget or balance sheet can be considered as an enabling tool as part of a package of actions for N pollution mitigation, with co-benefits for climate, air, water, soils and biodiversity. The production of a N budget or balance sheet is essentially a technical task of collating and presenting existing data to quantify N flows into and out of a geographical area within a given time period. As an enabling measure, it does not in itself reduce emissions; it presents the full picture of where N resources are being used and where they are being wasted. By highlighting the opportunities for improvement, this can then inform and motivate the setting of priorities for action.

The terms 'balance' and 'budget' appear to be used interchangeably (Worrall, 2016). Guidance adopted by the UNECE in 2013 defines a N budget in much the same terms as the Scottish Parliament defines a N balance sheet (UNECE, 2013a; Scottish Government, 2019). This is in contrast to the UK Government's legally-binding carbon budgets which place a restriction on the total amount of greenhouse gases the UK can emit over a defined period, currently 5 years (BEIS, 2021).

Nitrogen balances and budgets have often been produced in a purely agricultural context. A national N budget is rather like a farm level N management plan. Both are tools to enable the stakeholders to manage better at each scale. The OECD has identified N balance as a key agri-environment indicator and OECD/Eurostat published its first Gross Nitrogen Balances Handbook in 2003, updated in 2007 (OECD, 2007). It continues to collate and present data on agricultural nutrient N balance in member states (https://data.oecd.org/agrland/nutrient-balance.htm).

However, the N balance/budget approach has developed to reflect a more comprehensive understanding of the N cycle and its impacts over the last decade. Fan *et al* (2020) claim to be the first to estimate a total N budget (including inert nitrogen) and to attribute N sink and source areas at 1km^2 spatial resolution across Great Britain. In the UK, N balances/budgets had not been effectively deployed to inform government policy and legislation but that is beginning to change as these new tools become available.

3.3.2 The Scottish approach to a nitrogen balance sheet

The Scottish N balance sheet currently under development originated as part of climate change mitigation measures. The Climate Change (Emissions Reduction Targets) (Scotland) Act 2019 requires the establishment of a national Nitrogen Balance Sheet for Scotland by March 2022 in order to "record how nitrogen use efficiency contributes to achieving the targets in this Act".

The targets are to reduce Scotland's emissions of all greenhouse gases to net-zero by 2045 at the latest, with interim targets for reductions of at least 56% by 2020, 75% by 2030, 90% by 2040 from the baseline (Scottish Government, 2019). For nitrous oxide (N_2O) (as well as carbon dioxide and methane), the baseline year is 1990.

In the 2019 Act, "nitrogen use efficiency" is defined as "the ratio of nitrogen removed from the environment compared to total nitrogen inputs". The Scottish Nitrogen Balance Sheet (SNBS) must "quantify all major nitrogen flows across all sectors and media in Scotland, including its coastal waters, the atmosphere and soil and flows across these boundaries". Through regulations, the Government must make provision for:

1. a baseline figure for N use efficiency;
2. how N use efficiency is to be calculated;
3. the timescale in which the N balance sheet is to be reviewed;
4. monitoring and reporting upon the N balance sheet;
5. such other matters as they consider appropriate.

The development of the SNBS has been supported by a technical study by CEH (Carnell *et al*, 2019) and an initial stakeholder consultation in late 2020 and January 2021. The Government published a consultation analysis report and initial Government response in March 2021 (Scottish Government, 2021) and is now preparing regulations to lay before the Scottish Parliament. The consultation responses demonstrated broad support for the Scottish Government's proposed approach to the SNBS, which includes:

- using the SNBS to support a range of **wider policy applications** in addition to climate change mitigation – such as the development of the **new air quality strategy** and to further promote **efficiency in food production**;
- making the SNBS be as **comprehensive** as possible in terms of its **coverage and level of detail** for all sectors of the economy and the environment, and **fully integrated with other policy frameworks and strategies**;
- extending the SNBS beyond the national scale to a range of more detailed **sub-national spatial scales**, where data is available;
- setting **targets for improving nitrogen use efficiency** based on the SNBS once the evidence base is sufficiently established to allow for this to be done robustly;
- producing **annual updates** of the SNBS;
- making any outputs associated with the SNBS as **accessible and widely promoted** as possible, including production of factsheets.

3.3.3 Replicability of the Scottish approach in other UK countries

There is no available literature indicating political intent or work in progress in other UK countries to produce an official national N balance sheet or budget. However, as a technical task of data collation, analysis and interpretation, there should be no barriers to replicating the Scottish Government's approach in other UK countries. The methodology is replicable and the datasets (or equivalents) used to create the initial SNBS should be available for other UK countries (see Annex 3.2).

However, there remain technical challenges in producing a comprehensive national N balance sheet for any UK country. Carnell *et al* (2019) identified a number of key N flows which could not be quantified within the scope of the project and/or require further disaggregation within sectors and sub-sectors. The approach taken by the Scottish Government therefore requires further development, potentially including new methodologies for quantifying N flows. Collaboration between administrations across the UK and internationally (through UNECE and the developing International Nitrogen Management System) to enhance the methodology and identify additional data sources would help ensure that national N balance sheets provide an effective basis for policy and legislation to reduce excess N in the environment.

3.3.4: What would a N budget look like or need to contain for each country

Each country is developing a different package of climate solutions, including legal mechanisms, market incentives, policy initiatives, technical advice and financial support. They include net zero or other emissions reduction targets but also measures to increase climate adaptation and resilience. Reducing excess N will not only help to reach net zero emission but also deliver co-benefits for public health, biodiversity and ecosystems through healthy soils and cleaner air and water, all of which will make society and the natural environment more resilient to living with climate change. Each country's N budget should highlight and pursue these co-benefits in order to achieve an integrated approach, maximum public benefit and value for money.

The detail of each country's N budget will differ widely, according to the variation in demographics, energy supply, industry, land use and farming systems. It is clear from the CCC's analysis, and other evidence presented in this report, that agriculture will feature significantly in each national N budget within the UK.

To be effective in informing and establishing policy, rather than simply quantifying the status quo, a N budget for each country would need to contain the following principles:

- An **overarching vision** of future N use efficiency to reduce environmental losses and minimize the impacts on people, nature and the climate. In line with the Colombo Declaration on Sustainable Nitrogen Management, we recommend that this vision be based on an **ambition to halve nitrogen waste by 2030** along with longer-term targets to 2050.
- A vision that is clearly linked into **net zero targets and the Sustainable Development Goals**, demonstrating multiple benefits for climate mitigation and adaptation, human health and wellbeing, and biodiversity and ecosystems through reduced N pollution.
- Quantification of **total nitrogen flows into, within and out** of the country and their **impacts on people, nature and the climate.**
- **Broad scope of sectors** including agriculture, aquaculture, energy production, fisheries, food and drink production, extractive industries, forestry, horticulture, human nutrition, health and wellbeing, industry, transport, waste management and other forms of land management (such as private shooting estates and nature conservation sites). This will implicate various government functions such as spatial planning, regulation and environmental permitting.
- **Spatial and sub-sector analysis** of N emissions to air, water and soils to identify key geographical areas and priority sectors/sub-sectors.
- **Environmental scope** of N cycles including atmosphere, terrestrial, freshwater and coastal systems and **impacts** on species diversity, abundance and distribution, as well as soil health, air quality and water quality.
- **Transboundary imports and exports**, including food, livestock feed and manufactured fertilizers, and **impacts** of N exports on people and nature in other parts of the world.
- **Targets and actions to reduce N_2O emissions** progressively in line with each country's net zero targets.
- **Targets and actions to reduce NO_x and NH_3 emissions** in parallel with net zero targets, as part of a wider package of climate solutions and benefits to people and nature.
- Analysis and actions for addressing **interrelationships with other greenhouse gases**, to maximize cross-benefits and minimize trade offs.

- **Integration with other policy and legislation** for relevant sectors and environmental issues, consistent with the sectoral and environmental scope outlined above.
- Consistency and linkages with **UNECE Guidance document on national N budgets** and **Guidance document on integrated sustainable N management** to facilitate collaboration with other governments and intergovernmental organizations to ensure the best available data, methodologies and policies are used tackle the causes and impacts of excess nitrogen.
- Demonstrated intent to collaborate with the UK Government and other devolved administrations in an integrated approach to nitrogen management **across the UK.**

Country N budgets within a potential wider international nitrogen budget

The UNECE Revised Gothenburg Protocol requires the "calculation of nitrogen budgets, nitrogen use efficiency and nitrogen surpluses and their improvements" (ECE/EB.AIR/114 Article 7.3(d)) but there is no established system for governments to collect and report this information. This is intended to form part of the International Nitrogen Management System, serving the Gothenburg Protocol and other conventions and harmonizing reporting to allow comparability between countries.

3.4 Existing UK and Devolved Nations policy landscape and policy options

3.4.1 EU and international mechanisms

Across the many sources of excess N and in each country, a range of legal mechanisms is in force. EU legislation and international agreements have been the basis for much of each country's environmental and agricultural regulatory framework; the most relevant of these are set out in the table below (Table 3.4.1). Each of these has corresponding primary or secondary legislation transposing the requirements into domestic law in each country; these are not repeated in the country tables below, unless their content is significantly different from the UK target, limit or objective.

Following the UK's exit from the EU, the regulations transposing EU legislation into domestic legislation are undergoing a process of review and revision. New primary legislation has also been enacted (e.g., UK Agriculture Act 2000) or is in the process of development (e.g., Clean Air Act (Wales)). Only the most relevant legal mechanisms are considered in this report and it should be noted that the policy context and legislative framework is constantly changing. Commitments to new legislation are highlighted where particularly relevant but there is a broader range of policy which is not comprehensively listed here.

Table 3.4.1 EU and international legal mechanisms in force across the UK

EU and international legal mechanisms in force across the UK		
Focus area	**EU & international legal mechanisms**	**Broad UK targets/limit/objective**
Air quality	EU National Emissions Ceiling Directive UNECE CLRTAP Gothenburg protocol	2020 & 2030 targets for NO_x (55% & 73%) & NH_3 (8% & 16%) reduction from 2005 baseline
	EU Directive on Medium Combustion Plants	Establishes limits on NO_x emissions from individual plants
	EU Directive on Ambient Air Quality	Establishes limits on ambient NO_x concentrations and establishment of air quality zones
Water quality	EU Water Framework Directive (WFD) (inc. EU Groundwater Directive)	Broad objective to achieve 'Good status' for all water bodies by a set deadline and requires the conservation status of water-dependent Natura 2000 designated sites that are impacted by water pollution to be improved.
	EU Nitrates Directive (also within WFD)	Limit of 170 kg/ha/year N manure application in NVZs
Climate change	UNFCCC Paris Agreement	Requirement to set a Nationally Determined Contribution (NDC) UK has set legal targets of net zero by 2050, 78% reduction by 2035
Integrated approaches	EU Environmental Impact Assessment Directive	General requirements for environmental controls, not specific to excess N
	EU Environmental Liability Directive	
	EU Industrial Emissions Directive	
	EU Integrated Pollution Prevention and Control Directive	
Biodiversity and ecosystems	EU Habitats Directive	Requirement to maintain or restore Natura 2000 sites to favourable conservation status
	UN Convention on Biological Diversity (CBD)	Aichi Targets for 2020 *Post-2020 Framework is in development*

3.4.2 Domestic legislation

The **UK's exit from the EU** means that environmental and agricultural regulation in each nation is in a period of immense change with significant divergence emerging between the nations. Brexit has also left significant regulatory and enforcement gaps in terms of governance, principles and binding targets. In March 2021, Greener UK's Brexit Risk Tracker rated all areas of environmental policy as either high or medium risk (Greener UK, 2021). However, it also brings opportunities for legislators and regulators to take a new, more integrated approach to N.

Currently, the **lack of integrated regulation** (and the guidance accompanying it) across air, water, climate and other topics in all nations has led to a siloed and piecemeal approach to N management. This fails to realise the potential co-benefits of each action and makes compliance and enforcement more complex for private actors and public agencies. Initiatives such as the Scottish N balance sheet and proposals for integrated farm regulation in Wales are positive signals of a shift towards a more integrated approach. A study for the Institute of European Environmental Policy recommends that 'a fresh approach to the system of regulation for farmers and other land managers in England is required post EU-exit to maintain and improve environmental standards. A new delivery model should aim to build a more collaborative and long-term relationship with farmers, strengthen compliance and be adequately funded' (Baldock and Hart, 2020).

Water quality is the most highly developed area of N regulation due to the EU Water Framework Directive and Nitrates Directive. As well as controlling nitrate pollution, the requirements on all farms (such as nutrient management planning & restrictions on fertiliser use) also provide limited benefits for controlling NH_3 and N_2O emissions. Additional requirements in Nitrate Vulnerable Zones (NVZs) cover all of Northern Ireland and Wales (2021 regulations yet to come into force), around 55% of England and five lowland areas in southern and eastern Scotland. However, a lack of compliance by farm businesses and inadequate resources for effective enforcement means that this regulation has not been successful in reducing nitrate levels significantly and this (combined with phosphate pollution) remains a key driver of freshwater biodiversity loss, particularly in England and Wales. Post-Brexit, the future of water quality regulation is highly uncertain, as the WFD requirements transposed into domestic legislation currently expire in 2027. In England, Defra have recently set a higher target for advice-led Environment Agency inspections, but WWF estimate that the a given farm can still expect at most a regulatory visit once every fifty years on average rather than the 200 years cited in the 2018 Farm Inspection and Regulation Review (Defra, 2018b).

The Farming Rules for Water (FRfW) that came into force in England in 2018 now require that solid manure must be incorporated into bare soil within 12 hours, but implementation is a problem. Evidence of non-compliance can be found in the British Survey of Fertiliser Practice 2019, which provides estimates of the timing of incorporation of organic manure on tillage fields in Great Britain (Defra, 2019c). Regulation 4 of the FRfW states that the land manager (in England) must ensure that reasonable precautions are taken to prevent agricultural diffuse pollution resulting from applying organic manures, including by "incorporating organic manure and manufactured fertiliser into the soil within 12 hours of, or as soon as possible after, its application". The survey found that 11% by volume was not incorporated at all and a further 11% was incorporated more than a week after application. The 30% incorporated 1–7 days later may include some that meets the 'as soon as possible' test, but suggests that around half of the organic manures applied are not incorporated within the regulatory timeframe.

Air quality regulation (for public health) and **climate-related** regulation (as a co-benefit of action to reduce CO_2) have been effective in reducing **NO_x emissions** from transport and energy sectors significantly since 1970, largely due to direct regulation of pollution sources and market interventions. However, fine Particulate Matter ($PM_{2.5}$) remains a key concern for public health in urban areas and this is focusing increased attention on the contribution of NH_3 emissions as a precursor.

There is very little direct regulation controlling **NH_3 emissions** (although there are some co-benefits from water quality regulation). Governments have relied on a voluntary approach but this has failed to reduce ammonia emissions significantly. Despite legally-binding NECD targets at UK level, these have not been adopted into devolved nations' legislation and the UK's 2020 ammonia target for an 8% reduction from a 2005 baseline is almost certain to be missed, having declined by only 2% by 2019 and having increased in recent years following a period of lower emissions from 2008 to 2013 (Defra, 2021a).

The UK Government's Agricultural Transition Plan 2021-2024 (for England) states that "We plan to offer a slurry investment scheme from 2022, to help reduce pollution from farming and contribute to the 25 Year Environment Plan and net-zero commitments. This scheme will help farmers to invest in new slurry stores that exceed current regulatory requirements and are proofed against higher standards that we expect to introduce in the future. Alongside this scheme, we plan to implement new regulations as part of the Clean Air Strategy to cover all slurry stores. We intend that by raising standards, ensuring all farmers meet the basic legal requirements and providing targeted investment support where needed, we will break the cycle of private under-investment in slurry storage and emissions reduction. All slurry stores constructed will have to meet legal construction standards and be suitably maintained to ensure they do not pose a risk of serious pollution incidents in future." (Defra, 2020). The scheme will also enable farmers to adopt other pollution-reducing measures such as low emissions spreaders (to be a legal requirement by 2025).

Only **environmental permitting** (for intensive pig & poultry units) and **planning policy** (for new agricultural developments) aimed at reducing environmental impacts provide direct controls on farm ammonia emissions. These are largely based on the **Habitats Directive** and national conservation designations (SSSIs/ASSIs) which protect certain priority habitats and species. However, controls within the planning system are at the discretion of the local planning authority and vary widely between each authority. The environmental permitting system suffers from a lack of resources for effective monitoring and enforcement. Overall, the permitting and planning systems have not been successful in preventing the harmful impacts of excess N on biodiversity, ecosystems and public health.

Emerging 'net zero' greenhouse gas emissions targets, legislation and policies have the potential to bring about more effective regulation of atmospheric emissions, leading directly (N_2O) and indirectly (NO_x, NH_3) to reductions of excess N, and to trigger a radical shift in land use and land management. However, significant additional resources (beyond recent commitments) are needed for monitoring, compliance and enforcement by statutory agencies in order to make any regulation effective.

ENGLAND		
Thematic area	**Key legislation**	**Implementation**
Climate change	In June 2019, the UK Government amended the Climate Change Act 2008 to set a target of net zero emissions by 2050.	The CCC's 2021 Progress Report to Parliament concludes that faster action is needed; the Gov's Net Zero Strategy was published in October 2021[12]
Air quality	The Environment Bill contains new domestic measures on air quality post-Brexit, including a target on $PM_{2.5}$ ambient concentrations.	The Clean Air Strategy 2019 and Agriculture Transition Plan 2020 make other commitments to new regulation but these have not yet been delivered in relation to ammonia.
Water quality	Farming Rules for water apply to all farms.	Around 55% of England is within an NVZ. Only 14.6/15% of rivers are in good ecological status under WFD. Lack of compliance with and implementation effort around the FRfW have been repeatedly highlighted by WWF and others.
Agriculture and soils	The Agriculture Act 2020 shifted agricultural payments to a 'public money for public goods' approach post-Brexit, phasing out direct payments. Section 33 of the Act introduces new provisions on fertilisers.	Recent reports have highlighted significant failings in farm regulation and enforcement: Farm Inspection and Regulation Review IEEP: Post EU exit Regulatory Framework Guardian article: no penalties issued under 'useless' English farm pollution laws EAC Committee: Performance on Reducing Nitrates
Biodiversity & ecosystems	Public authorities have a duty to have regard to conserving biodiversity. The Environment Bill proposes a legally-binding target for species abundance, a stronger duty to 'conserve and enhance', biodiversity net gain as a condition of planning permission and the production of local nature recovery strategies.	'Have regard to' often means little in practice and is at the discretion of the public body. The new Act may strengthen biodiversity protections and restoration if properly implemented. However, it could also give the Government powers to amend the Habitat Regulations.
Additional sources	EA State of the environment reports	

[12] https://www.gov.uk/government/publications/net-zero-strategy

NORTHERN IRELAND		
Thematic area	**Key legislation**	**Implementation**
Climate change	*Private Member's Bill* was tabled in March 2021 but is opposed by the Executive, which has consulted on its own Discussion Document on a Climate Change Bill.	NI Executive has lagged behind other administrations in setting new GHG reduction targets, although a previous target of 20% reduction by 2020 was met.
Air quality	*A Clean Air Strategy for Northern Ireland – Public Discussion Document* was published for consultation in November 2020 and an *Action Plan on Ammonia Reduction* is "very close to completion" according to department official, following an Expert Working Group report and DAERA response, which may lead to new legislation.	Although the impacts of ammonia emissions are well recognised – perhaps more so in NI than other nations – there has been slow progress in policy and legislation, partly due to heavy resistance from the farming industry.
Water quality	Nutrient Action Programme Regulations (Northern Ireland) 2019 delivering on WFD and previous SSAFO regulations. NI has a derogation from the Nitrates Directive, allowing application of 250kg N/ha/yr for intensive grassland farms.	The whole territory is an NVZ. Nitrate levels in NI are lower than other parts of the UK but less than 40% of water bodies have achieved Good Ecological Status, also in part due to phosphate pollution and other factors.
Agriculture and soils	Sustainable Agricultural Land Management Strategy 2018 - independent expert group report to DAERA	n/a
Biodiversity & ecosystems	The Environment (Northern Ireland) Order 2002 (legislation.gov.uk) establishing a duty on public bodies to protect and enhance the designated features of ASSIs The Conservation (Natural Habitats, etc.) Regulations (Northern Ireland) 1995 as amended by The Conservation (Natural Habitats, etc.) (Amendment) (Northern Ireland) (EU Exit) Regulations 2019	Only 9.8% of land is protected for nature & only 55% of ASSI biological features are in favourable condition. 4.5% of MPAs and 13.7% of terrestrial sites were under favourable management (2019/20), while 74% of ASSIs had not been monitored in the last 6 years.
Other/integrated	Pollution Prevention and Control Regulations (Northern Ireland) 2013	Unknown
Additional sources	Priorities-for-the-Environment-2021-26-1.0.pdf (nienvironmentlink.org) NIEL-Environmental-Scorecard-2021.pdf (nienvironmentlink.org) NMNI-NI-Response--Environmental-Plans-Principles--Governance---February-2021.pdf (nienvironmentlink.org) RSPB NI: Northern Ireland Biodiversity Strategy failing after years of inaction	

SCOTLAND		
Thematic area	**Key legislation**	**Implementation**
Climate change	The Climate Change (Emissions Reduction Targets) (Scotland) Act 2019 and Climate change plan 2018–2032 sets targets & a pathway to net zero emission by 2045 (70% reduction by 2030).	This legislation sets the requirement for a National Nitrogen Budget Sheet, which could lead to integrated action on N pollution.
Air quality	In March 2021, the Scottish Govt. published its analysis of responses to the consultation on an updated Cleaner Air for Scotland strategy, originally published in 2015. There is no commitment to new primary legislation.	The draft new strategy continues to rely on voluntary measures to meet Scotland's contribution to NECD ammonia targets.
Water quality	A raft of legislation applies; only five areas have been designated as Nitrate Vulnerable Zones. There are General Binding Rules for all farms under the Water Environment (Miscellaneous) (Scotland) Regulations 2017.	Nitrate levels are generally lower in Scotland than in England and Wales.
Agriculture and soils	The Scottish Govt. has not published plans for new agricultural support systems or regulation post-Brexit. Its Land use strategy 2021 to 2026 provides a broader policy framework but the recommendations of the 'Farming and Food Production – Future Policy Group' are awaited.	See blog: Where is the future for Scotland's food and farming sectors? - Scotlink
Biodiversity & ecosystems	The Nature Conservation (Scotland) Act 2004 governs SSSIs and places a duty on planning authorities and all public bodies to "further the conservation of biodiversity" and to report on compliance with this duty every 3 years.	A new biodiversity strategy will be published within 1 year of CBD CoP15: Scottish biodiversity strategy post-2020: statement of intent
Other/integrated	National Nitrogen Balance Sheet in progress. Government's Environment Strategy for Scotland: vision and outcomes	
Additional sources	ScotLink is calling for an ambitious new Environment Act to set legally-binding targets for nature: A Manifesto for Nature and Climate - Scottish Parliament 2021 election - Scotlink	

WALES		
Thematic area	**Key legislation**	**Implementation**
Climate change	Environment Act (Wales) 2016 established a requirement to set 10-year GHG targets to 2050 and 5-year carbon budgets. In March 2021, Senedd Cymru approved a net zero target for 2050 and a second carbon budget is due for publication later in 2021.	Welsh Govt. published a response to the CCC's 2020 progress report confirming that Wales is on track to meet its 2020 target (27% reduction) and 1st carbon budget.
Air quality	Welsh Govt. published its first Clean Air Plan for Wales in 2020 & consulted on a White Paper on a Clean Air (Wales) Bill in early 2021. This includes powers to set targets, including for $PM_{2.5}$, with reporting requirements. The Bill is not in the Gov's programme for 2021-22 but is within their 5 year programme. Existing and previous air quality legislation & strategy was UK-wide.	n/a Action is underway to reduce roadside NO_2 concentrations and improve urban air quality
Water quality	Water Resources (Control of Agricultural Pollution) (Wales) Regulations 2021 strengthen control of nutrient management & fertiliser application on all farms across Wales, phased in over 3 years. Replace the Water Resources (Control of Pollution) (Silage and Slurry) (Wales) Regulations 2010.	These regulations have been introduced following the failure of previous measures & lack of enforcement by NRW. The regulations have been heavily resisted by farming groups & supported by env. groups.
Agriculture and soils	Welsh Govt. consulted on a White Paper for an Agriculture (Wales) Bill in early 2021 and the Bill is expected to be introduced in autumn 2021. This will establish a new Sustainable Farming Scheme post-Brexit based on proposals in the 2019 consultation Sustainable Farming and Our Land and National Minimum Standards to consolidate farm regulation into a clear baseline for all.	This a rapidly evolving and controversial area of policy but Govt. proposals will help to integrate control of on-farm N. WEL's response to the Agriculture (Wales) White Paper provides a useful overview.
Biodiversity & ecosystems	The Wildlife & Countryside Act 1981 is the legal basis for SSSIs in Wales. In addition, Section 6 of the Environment (Wales) Act 2016 places a duty on public authorities to 'seek to maintain and enhance biodiversity' and 'seek to 'promote the resilience of ecosystems'. Section 7 places a duty on Welsh ministers to 'take all reasonable steps to maintain and enhance' certain species & habitats listed under the Act.	NRW's Protected sites baseline assessment 2020 showed a failure to maintain or even monitor SSSIs. The Nature recovery action plan for 2020-21 recognises the wider & continuing loss of biodiversity; actions include 'Addressing direct pressures on Resilient Ecological Networks e.g. **pollution**, climate change […].”
Other/integrated	The Well-being of Future Generations (Wales) Action 2015 places a duty on public bodies to work towards economic, social, environmental and cultural wellbeing with goals including 'A resilient Wales', 'A healthier Wales' and 'A globally responsible Wales'. Indicators cut across the goals and include NO_2 levels, GHG emissions and water quality.	This is an integrated long-term government policy for sustainable development which provides a legal basis for action to control N pollution at a strategic level for public bodies, but not specific controls on pollution.

3.4.3 Gaps in regulation and enforcement:

In broad terms across the UK:

- **Water quality** regulation, while the most developed, has not been successful in reducing nitrate levels for a variety of reasons. The SSAFO Regulations have a loophole exempting storage facilities built before 1991, which environmental NGOs argue should be closed. Any viable solution will need to factor in the different sizes of farming operation in the UK (see Table 3.4.2). For example, in Denmark (See Section 3.1.4) permitting is applied on most farms, requiring a fertiliser plan and adherence to N application limits, with small farms not requiring a permit, but incentivized to create a fertiliser plan by a tax relief on mineral fertiliser. Simple nutrient management planning is already a legal requirement in FRfW (England) and by Nitrates Regs (all devolved - but not all in force yet) and may well be part of the new Sustainable Farming Initiative (SFI) to be launched in 2024 in England. Nutrient management is expected to form an element of the SFI and nutrient efficiency is also covered in Defra's transitional 'productivity' scheme, but this will also require monitoring and enforcement. Furthermore, the Farm Inspection and Regulation Review (2018) advocates permitting for all farms, including intensive dairy.

Table 3.4.2 Number of farm holdings by Size group in UK (Defra Statistics Office)

At June of each year

		2012		2017	
		Number of holdings (thousand)	Hectares (thousand)	Number of holdings (thousand)	Hectares (thousand)
Total area on holdings	under 20 hectares	104	694	103	682
	20 to under 50 hectares	42	1 399	41	1 359
	50 to under 100 hectares	34	2 428	32	2 262
	100 hectares and over	42	12 628	41	13 334
	Total	**222**	**17 149**	**217**	**17 637**
	Average area (hectares)		77		81
	Average area on holdings with >=20 hectares		139		149

- **Air quality** and **climate** regulations have been effective in reducing **NO_x emissions**, largely due to direct regulation of pollution sources and market interventions. However, particulate matter remains a challenge and this is focusing increased attention on the contribution of NH_3 emissions.
- There is very little direct regulation controlling **NH_3 emissions**, despite legally-binding NECD targets; only **environmental permitting** (for intensive pig & poultry units) and **planning policy** (for new agricultural developments) act as a control on ammonia emissions.
- The **Habitats Directive** and national conservation designations (SSSIs/ASSIs) establish certain limits on air & water pollution from new development affecting habitats and species; this is delivered through the environmental permitting and spatial planning systems. However, the permitting and planning systems have not been successful in preventing the harmful impacts of excess nitrogen on biodiversity, ecosystems and public health.
- A need for **integrated regulation** and the guidance across air, water, climate and other will be needed to realise the potential co-benefits of each action and makes compliance and enforcement more complex for private actors and public agencies.
- The UK's exit from the EU left significant regulatory and enforcement gaps in terms of governance, principles and binding targets. There is further opportunity to learn from

the experience of our European neighbours as outlined in Section 3.1. However, developments in Denmark such as allocating the majority of their EU funded rural development programme to tackling pollution is not appropriate for the UK, as it goes against the subsidiarity and polluter pays principles. The UK is currently pursuing a more positive “public money for public goods” approach, with negative externalities primarily dealt with by regulation except where new issues arise (such as ammonia). In terms of reliance on technology (as discussed for Denmark), it can play a big part both in sustainable intensification and agroecology, but is unlikely to be sufficient on its own to achieve necessary change.

3.5 Options for integrated approaches and targets

As we have seen in Section 3.1.1, the policy/regulatory landscape for N is evolving quickly in recent years. A further key development is the push towards Net Zero and carbon neutrality in all sectors and in this regard agricultural practices have a key part to play as society moves in this direction. In this section we consider the implications of the UK’s 6th Carbon budget (see Section 1.5 and 2.3) for the agricultural sector.

3.5.1 Implications of the Sixth Carbon Budget for GHG Emissions from Agriculture

As set out in Section 1.5 of this report, the Sixth Carbon budget’s Balanced Pathway to Net Zero requires 9% of agricultural land for actions to reduce emissions and sequester carbon by 2035 with 21% needed by 2050. The CCC (2020b) considers five measures that can potentially release land covering societal changes and improvements in agricultural productivity:

1) Diet change (the most significant measure, Section 1.5. for details);
2) Food waste;
3) Improving crop yields;
4) Managing stocking rates for livestock;
5) Moving horticulture indoors.

These five measures, together with low carbon agricultural measures and fuel /agricultural machinery improvements, are estimated to reduce sectoral emissions from 54.6 MtCO_2e in 2018 to 39 in 2035 and 35 by 2050 under the balanced pathway. The 15.6 MtCO_2e reduction by 2035 it is estimated to be split by: low carbon farming practices (4 MtCO_2e), fuel /agricultural machinery (2 MtCO_2e), and the rest through a combination of the following land release measures diet change (7 MtCO_2e), food waste reduction (1 MtCO_2e), and productivity improvements related to crop yield and stocking rates (1 MtCO_2e) (Figure 3.5.1; Table 3.5.1).

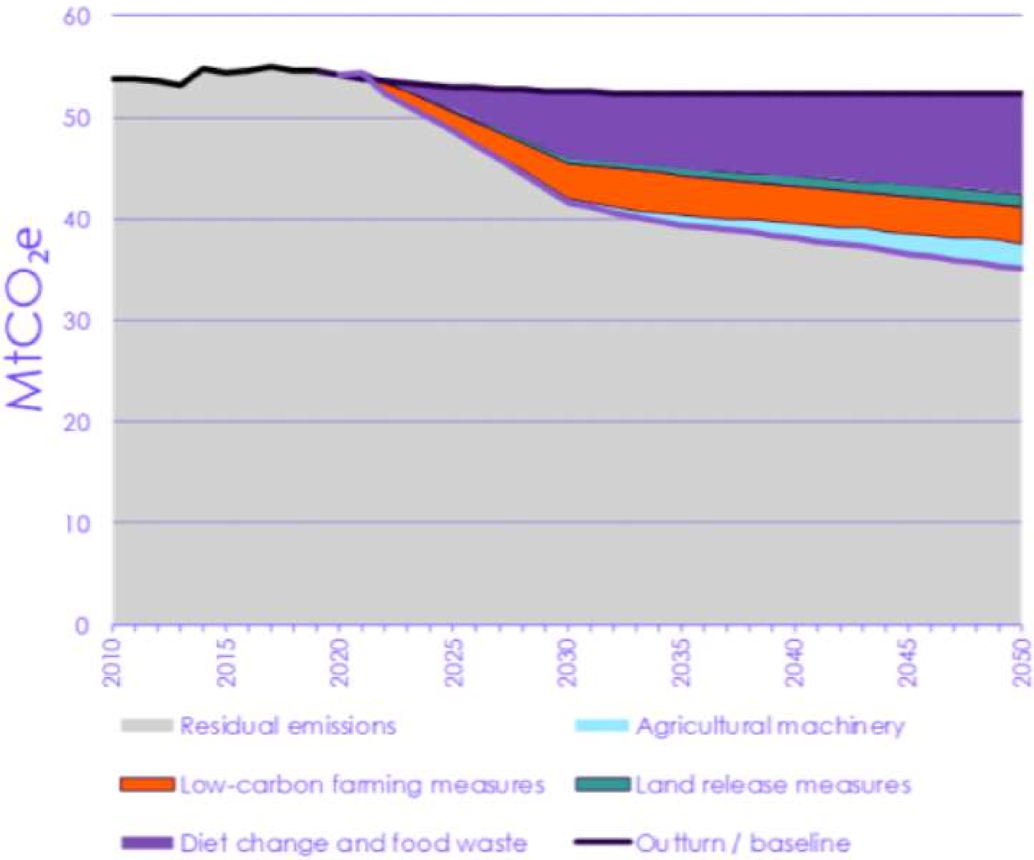

Figure 3.5.1 Sources of abatement in the Balanced Net Zero Pathway for the agricultural sector. Source: BEIS (2020) Provisional UK GHG National Statistics 2019; Eory et al. (2020): CCC analysis (CCC ,2020b)

The Balanced Pathway involves a 20% shift away from meat and dairy products by 2030, with a further 15% reduction of meat products by 2050, with meat substituted with plant-based options. The Balanced Pathway means reduction in livestock numbers and grassland area to deliver annual abatement of 7 $MtCO_2e$ by 2035, rising to nearly 10 $MtCO_2e$ by 2050 (see Table 3.5.1). The CCC also assumes food waste is halved across the supply chain by 2030 in line with the Waste and Resources Action Programme's (WRAP) UK Food Waste Reduction Roadmap, reducing UK emissions by almost 1 MtCO2e in 2035. Under the Widespread Engagement scenario (Table 3.5.2) a greater shift away from meat and dairy (e.g. a 50% switch by 2050) and a greater willingness to act on food waste results in additional GHG savings of 2 $MtCO_2e$ in 2035 (Table 3.5.1).

***Table 3.5.1**: Potential abatement under Balanced Pathway (in **bold)** achieves reduction from 54.6 $MtCO_2e$ in 2018 to 39 $MtCO_2e$ in 2035, and to 35 $MtCO_2e$ by 2050 (CCC, 2020b). (a) Land release measures; (b) Low Carbon measures; (c) Fuel /agricultural machinery measures; and various estimates/ expert judgement for potential reduction in N losses (Total N, N_2O, NH_3 and NO_3), CH_4 /GHG emission reductions. See table footnotes, text and Section 3.5.2 for further details.*

Measure	Abatement ambition – Balanced Net Zero Pathway assumptions	Potential Abatement – Balanced Net Zero Pathway, $MtCO_2e$ by 2035 (Effect of Widespread Engagement Scenario in brackets)	CH_4	N_2O	NH_3	NO_3 leaching and run-off	N Loss
Diet Change (a)	Medium level: 20% cut in meat and dairy by 2030, rising to 35% by 2050 for meat only. All replaced with plant based;	**7** (+2[1])	~ -35% (EU)[2]	~ -31% (EU)[2]	~ -43% (EU)[2]	~ -35% (EU)[2]	~-42% (EU)[2]
Food waste (a)	Medium Level; 50% cut in food waste by 2030, 60% by 2050	**1** (+2[1])	Especially from landfill				~-17%[3]
Productivity improvements (crop & livestock) (a)	Medium level for increasing average crop yields, livestock stocking rates on grassland, and shifting horticulture indoors	**1**[4] (+1)	Enteric fermentation responsible for over half of emissions from agricultural sector	Yes - amount emitted depends on livestock waste and crop management (see Section 2.3)			
Fuel /agricultural machinery (c)	Mix of electrification, hydrogen and later phase out of biofuels	**2** (+0[5])					(yes NOx)
Low-carbon measures (b)	Lower uptake: 50-75% for both behavioural measures and innovation measures	**4** (+1[6])	Especially from livestock	Yes - amount emitted depends on livestock waste and crop management (see Section 2.3)			

[1] Under the Widespread Engagement scenario, a greater shift away from meat and dairy (e.g. a 50% switch by 2050) and a greater willingness to act on food waste results in additional GHG savings of 2 $MtCO_2e$ in 2035; [2]According to a 50% reduction in all meat and dairy (Greening scenario) with no reduction in food waste (Westhoek et al, 2015); [3] for halving of food waste in Europe (from 30-15% waste) does not include farm level improvements or diet change (energy or protein reduction or type of foods) but this is not only food waste, it is also better waste management (use of sewage etc.) (for details see 'Improved Scenario' in Corrado et al 2020); [4]Higher livestock stocking densities on permanent grassland releases around 0.8 million more hectares of land out of agricultural production under the Widespread Innovation and Tailwinds scenarios compared with the Balanced Pathway could result in 1 $MtCO_2e$ additional GHG savings in 2035; plus 30% of the higher level of diet shift is towards lab-grown meat rather than plant-based alternatives could result in 5 $MtCO_2e$ additional GHG savings by 2035; [5]While the mix of technologies differ across the pathways (Table 3.5.2), they all achieve the same level of abatement by 2050; [6]Measures associated with changing farming practices (e.g. planting cover crops, livestock health measures and feeding cattle a high starch diet) is highest in the

Widespread Engagement scenario, and take-up of more innovative options (e.g. 3NOP additives, GM cattle, and breeding) is highest in Widespread Innovation CCC (2020b). However, there is relatively little difference in emissions savings across these scenarios, which vary from 4 $MtCO_2e$ in Balanced Pathway scenarios to 5 $MtCO_2e$ under Widespread Innovation by 2035.

Measures to increase agricultural productivity in the Balanced Pathway including improving crop yields without the need for additional inputs achieved through improved agronomic practices, technology and innovation could reduce emissions by 1 $MtCO_2e$ in 2035 and 2050 (Section 1.5). The Balanced Pathway assumes wheat yield increases from an average of 8 tonnes/hectare currently to 11 tonnes/hectare by 2050 (with equivalent increases for other crops).

Table 3.5.2 Summary of key differences in the agriculture sector scenarios (CCC, 2020b)

	Balanced Net Zero	Headwinds	Widespread Engagement	Widespread Innovation	Tailwinds
Behaviour change and demand reduction	**Medium level: 20% cut in meat and dairy by 2030, rising to 35% by 2050 for meat only. All replaced with plant-based; and** **Medium level: 50% cut in food waste by 2030, 60% by 2050.**	Low level: 20% shift away from all meat types and dairy products to all plant-based by 2050; and Low level: 50% fall in food waste by 2030, with no further reduction.	High level: 50% less meat and dairy by 2050. All replaced with plant-based; and High level: 50% fall in food waste by 2030, 70% by 2050.	High level: 50% less meat and dairy by 2050 with 30% of meat replaced with lab-grown meat. Medium level: 50% cut in food waste by 2030, 60% by 2050.	Diet change aligned to Wider Innovation. Food waste reduction aligned to Widespread Engagement.
Other land release measures	**Aligned to Headwinds.**	Medium level for increasing average crop yields, livestock stocking rates on grassland and shifting horticulture indoors.	Medium level for increasing average crop yields and shifting horticulture indoors. Low level for increasing livestock stocking rates on grassland.	High level for increasing average crop yields, livestock stocking rates on grassland and shifting horticulture indoors.	Aligned to Widespread Innovation.
Low-carbon farming practices	**Aligned to Headwinds.**	Lower uptake: 50-75% for both behavioural and innovation measures.	High uptake of behavioural measures 60-80%; and lower uptake 50-75% for innovative measures.	High uptake of innovation measures 60-80%; and lower uptake 50-75% for behavioural measures.	Aligned to Widespread Innovation.
Agricultural machinery	**Aligned to Headwinds.**	Mix of electrification, hydrogen and later phase-out of biofuels.	Focus on electrification and biofuels.	Hydrogen, electrification and biofuels.	Aligned to Widespread Innovation.

CCC (2020b) estimate that crop breeding or gene editing for new cultivars /traits could lead to higher yields (e.g., to 13 tonnes/hectare for wheat by 2050). Higher livestock stocking densities on permanent grassland releases around 0.8 million more hectares of land out of agricultural production under the Widespread Innovation and Tailwinds scenarios resulting in 1 $MtCO_2e$ additional GHG savings in 2035. This assumes that there is scope to sustainably increase stocking rates for livestock through improving productivity of grasslands and management practices such as rotational grazing. A shift of 10% of horticulture production indoors under a controlled environment would also reduce the carbon, nutrient, land and water footprint.

In the Widespread Innovation scenario, the Sixth Carbon Budget assumes that availability of lab-grown meat will account for 30% of the diet shift rather than plant-based alternatives (CCC, 2020b). This results in 5 $MtCO_2e$ additional GHG savings by 2035. In the Headwinds scenario a 20% shift away from meat and dairy products is achieved by 2050 instead of 2035 with a 6 $MtCO_2e$ lower GHGs saving in 2035 than the Balanced Pathway.

Currently 18 TWh of fossil fuels are used in agricultural vehicles, buildings and machinery, resulting in emissions of 4.6 $MtCO_2e$ (CCC 2020b). Actions for reduced use are similar to those in other sectors such as surface transport, off-road machinery and commercial buildings including electrification, biofuels, hydrogen and hybrid vehicles. The Balanced Pathway assumes biofuels and electrification are adopted from the mid-2020s and hydrogen from 2030, reducing emissions to 2 $MtCO_2e$ in 2035 (Table 3.5.1). The technologies will differ across sectors, but will achieve similar abatement by 2050.

The SRUC assessed the abatement potential from measures to reduce emissions from soils (e.g., grass leys and cover crops; see Section 2.3.6.2), livestock (e.g., diets and breeding; see Section 2.3.3.2) and waste and manure management (e.g., anaerobic digestion; see Section 2.3.4.2) and the CCC (2020b) estimate that these reduce agricultural emissions by 4 $MtCO_2e$ in 2035 (Table 3.5.1). This reduction interacts with other actions, notably diet change, which reduces the abatement potential of these measures over time. The Sixth Carbon Budget assumes changing farming practices (e.g., cover crops, livestock health measures, high starch cattle diet) is highest in the Widespread Engagement scenario, and take-up of more innovative options (e.g., 3NOP additives, GM cattle, and breeding) is highest in Widespread Innovation (CCC, 2020b). However, there is relatively little difference in emissions savings across these scenarios, which vary from 4 $MtCO_2e$ in Balanced Pathway scenarios to 5 $MtCO_2e$ under Widespread Innovation by 2035 (which is aligned to Tailwinds scenario for the low carbon measures as shown in Table 3.5.2).

3.5.2 Implications for CH_4 and N losses to the environment of Net Zero plans

This report clearly shows that efforts to achieve Net Zero have considerable implications for reductions of N losses to the environment (including N_2O, NOx, NH_3 and NO_3), and N use efficiency, related to the consumption and production of food products in the agricultural sector in the UK (See Section 2 for details). In the absence of a detailed quantification of the implications for N losses of the Net Zero plans (SRUC and Defra have so far only conducted their own qualitative assessments (see Section 3.5.3)), Table 3.5.1 shows available figures from the literature to give an indication of how the Sixth Carbon Budget is linked to N losses into the environment.

For dietary change, the Widespread Engagement scenario is equivalent to the Greening Scenario of Westhoek et al. (2015) that estimates that halving of meat and dairy intake (also known as the “demitarian” scenario) in the EU could reduce total N loss by 42%, NH_3 and N_2O emissions by 43% and 31% respectively, N leaching and runoff by 35%, and total GHG emission (i.e. including CH_4) by 42% (Table 2.1.1). These estimates assume no change in food waste, with calorie intake maintained.

Research at the Joint Research Centre for the European Commission (Corrado et al., 2020) estimates that halving of food waste in Europe (from 30-15% waste) could potentially reduce N losses to the environment by 17% (Table 3.5.1). This estimate does not include farm level improvements or diet change (energy or protein reduction or type of foods), and is not only food waste, it is also better waste management (use of sewage etc.). Reducing food waste will have co-benefits for N losses and methane emissions from landfill sites and farming operations. It has recently been estimated that globally 1.2 billion tonnes of food is lost on farms each year, on top of the 931 million tonnes wasted at retail and consumption, which is at a much higher rate than previously reported, with approximately 40% of all the food grown going uneaten (WWF, 2021a).

Quantitative estimates are also not available for the N loss reduction potential associated with the reduction in GHG emissions of 4 $MtCO_2e$ in 2035 from low carbon measures in agriculture under the Balanced Pathway scenario (5 $MtCO_2e$ under the Widespread Engagement Scenario). Below, the analysis of interventions carried out in Section 2.3, using the qualitative guidance on magnitude of effect from UNECE (2021), is used to highlight where there are important linkages between GHG measures and N loss to the environment that merit further investigation and quantification.

In terms of GHG mitigation, the greatest abatement potential is provided by the use of ruminant feed additives (956.4 $MtCO_2e$ under Balanced Pathway, Table 2.3.2), which is a more innovative option that has highest uptake in the Widespread Innovation scenario (see Section 3.5.1). Section 2.3.3.3 shows that there is no evidence for trade-offs between addition of enteric methane inhibitors (3NOP and nitrate) and Nr emissions and that adding nitrate to the diet can replace some of the non-protein nitrogen or high-protein feeds such as soy meal in the diet, which in turn can reduce the amount feedstuffs imported. Such GHG measures therefore have significant potential for synergistic effects with N loss to the environment.

Measures associated with changing farming practices (e.g. planting cover crops, livestock health measures and feeding cattle a high starch diet) are most prominent in the Widespread Engagement scenario, and increased animal and herd-level productivity through better health, breeding and diet is a key theme of GHG mitigation from livestock (Section 2.3.3.2). In Section 2.3.3.3 we described how breeding and health-related (non-dietary) measures can achieve a high feed conversion ratio, which tends to reduce CH_4 emissions from enteric fermentation, volatile solid and N excretion. And, that measures that reduce N excretion, such as breeding, animal diet and health and increased milking frequency, can have 'major' co-benefits for air quality and water quality. However, to avoid tradeoffs the full N use chain needs to be considered when pairing high digestibility with a low crude protein content (as in the case of maize silage; see Section 2.3.3.3).

For crops and land use measures, grass and legumes have the greatest GHG abatement potential (524.4 $MtCO_2e$ under Balanced Pathway, Table 2.3.8) and a small to medium magnitude effect reducing NO_3, N_2O and total N losses, potential to increase biodiversity and a negligible to small effect on ammonia emissions depending on management and use of fertilizers (Table 2.3.7). Overall reductions of these N species also depend on appropriate timing of cover crop establishment and incorporation into the soil.

For waste management, anaerobic digestion of cattle manure has the greatest GHG abatement potential (424.6 $MtCO_2e$ under Balanced Pathway, Table 2.3.5) and a medium to large magnitude effect on reducing NH_3, NO_3, N_2O and total N losses, no direct impact on biodiversity and a small to medium effect on N_2O emissions (Table 2.3.3). However, currently less than 10% of manure is treated in this way in the UK. The feedstocks used for anaerobic digestion are also a crucial factor (see Section 2.3.4) e.g., the use of maize where the crop is fertilized has been linked to nitrate pollution of rivers as it is usually harvested late in the year when soils are often wet and susceptible to run-off, particularly on slopes.

3.5.3 Tradeoffs and Synergies with UK Environmental Policies

The Sixth Carbon Budget scenarios include a high take-up of low-carbon farming practices, which could deliver benefits to biodiversity and soil quality, while there could also be some risks (CCC, 2020b). Based on a review of evidence from Defra's on-going 'Delivering Clean Growth through Sustainable Intensification' project (not yet published), the wider environmental considerations of the 18 low-carbon measures in the Balanced Pathway have been assessed (Table 3.5.3).

Table 3.5.3 Qualitative assessment of wider environmental benefits from low carbon practices; CCC analysis based on the Defra Sustainable intensification project (CCC, 2020b)

Low-carbon farming practices	Water quality	Air quality	Biodiversity	Soil
Breeding measures				
Genomics	Minor	Minor	-	-
Current breeding	Major	Major	-	-
Low methane	Minor	Minor	-	-
GM cattle	Major	Major	-	-
Increase milk frequency	Major	Major	-	-
Livestock diets				
High sugar grasses	Major	-	-	-
Nitrate additives	Minor	Minor	-	-
Precision feeding	-	Major	-	-
High starch diet	Negative	-	-	Negative
3NOP	Minor	Minor	-	-
Livestock health				
Cattle	Major	Major	Minor	-
Sheep	Major	Major	Minor	-
Soil measures				
Grass legume mix	Major	-	Major	-
Grass leys	-	-	-	Major
Cover crops	Major	Major	-	Major
Waste management				
AD pigs	-	Negative	Negative	-
Ad cattle	-	Negative	Negative	-
Cover slurry tanks	Major	Major	-	-

Air and water quality are the most improved with nine of 18 measures, including increasing milk frequency, improving livestock health and covering slurry tanks with impermeable covers, delivering major impacts. Biodiversity and soil quality show less benefit, with only three measures deemed to have a major impact (grass legume mix for biodiversity and grass leys and cover crops for soil). There are also negative trade-offs which could potentially worsen air quality (anaerobic digestion of pig and cattle manure) and water quality (adoption of high-starch diets).

Eory et al. (2017) report on the potential wider impacts of GHG mitigation in agriculture, land use, land use change and forestry. Their qualitative assessment provides an overview of the wider impacts of the GHG mitigation options and potential co-benefits and trade-offs (as described below and in Table 3.5.4).

Table 3.5.4 Overview of the wider impacts of the GHG mitigation options (Eory et al., 2017)

		WI1 Air quality: NH_3	WI2 Air quality: NOx	WI3 Air quality: PM	WI4 Air quality: other	WI5 Water quality: N leaching	WI6 Water quality: P	WI7 Water quality: other	WI8 Soil quality	WI9 Flood mgmt, water use	WI10 Land cover and land use	WI11 Biodiversity	WI12 Animal health and welfare	WI13 Crop health	WI14 Household income	WI15 Consumer and producer surplus	WI16 Employment	WI17 Resource efficiency	WI18 Human health	WI19 Social impacts	WI20 Cultural impacts
MO1	On-farm renewables	0	+	+	+	0	0	0	+/-	0	-	0/-	0	0	+	+/-	+	+	+/-	+	0
MO2	Precision farming	+	+	+	+	+	+	+	+	+	0	+	+/-	+	+	+	-	+	+	+/-	0
MO3	Optimal soil pH	+/-	0	0	0	+	+	+	+	+/-	0	+/-	+	+	+	+	0	0	+	0	0
MO4	Anaerobic digesters	-/0	-	-	0	+/-	-	0	+/-	0	0	0	0	0	+	+	+	++	+/-	+	0
MO5	Agroforestry	+	+	+	0	+	+	+	+	+	+	+	+	+	0	0	0	0	+	0	+
MO6	More legumes	+	+	+	0	-	0	0	+	0	+	+	0	+	0	0	0	+	0	0	0
MO7	Optimal mineral N use	+	+	+	0	+	+	0	0	0	0	0	0	0	0	0	0	0	+	0	0
MO8	Manure storage and application	++	0	+	+/-	+	+	+	+/-	0	0	0	+	0	+/-	0	+	+/-	+/-	0	0
MO9	Livestock health	+	0	0	0	+	+	-	0	0	0	-	+/-	0	0	0	0	+	+/-	0	0
MO10	Reduced livestock product consumption	+	0	0	0	+	+	-	+/-	+/-	+	+/-	+/-	0	+/-	+/-	+/-	+	++	+/-	+/-
MO11	Afforestation	++	++	++	+	+	0	+/-	+/-	++	+	+/-	-	0	+/-	+/-	+/-	+	+	+	+/-
MO12	Peatland restoration	0	0	+	0	-	-	+/-	++	+/-	+/-	++	+	-	+/-	0	0	0	+/-	+	+/-

Legend

++	Strong positive effect
+	Positive effect
0	No siginificant effect
+/-	Variable effect
-	Negative effect
--	Strong negative effect
	Weak evidence
	Moderate evidence
	Robust evidence

Table Notes: The scores show the direction and magnitude of impact (positive denoting favourable impact) and the colour scale provides an assessment of the robustness of the available scientific evidence (weak evidence refers to situations where there is limited availability of evidence and/or there are conflicting findings, while robust evidence refers to conclusive evidence). The majority of the wider impacts were positive or neutral, with also a high number of variable impacts (i.e. positive and negative impacts both possible), but there are no strongly negative impacts.

On-farm renewable energy, precision farming, anaerobic digestion, agroforestry, optimal mineral N use, livestock health, reduced livestock product consumption, afforestation and peatland restoration all show the potential for delivering co-benefits in policy. In agreement with the analysis in this report (Section 2.3) and reported above (Section 2.5.2), strong positive effects were found for AD (resource efficiency) and for low emission manure storage and application (NH_3 emissions). Strong positive effects were also found for reduced livestock product consumption on human health, afforestation on air quality and on flood management and peatland restoration on soil quality and biodiversity.

Adverse impacts were associated with eight mitigation options, although evidence on some of these was limited and careful planning and implementation are needed to minimize these effects (Eory et al. 2017). The negative impacts with moderate or strong evidence covered a range of actions. On-farm renewables can have a small negative impact on land use by removing land from other uses. Anaerobic digesters are linked to the production of air pollutants (NOx and PM) via the combustion of biogas. Improving livestock health might negatively affect biodiversity if habitats are altered to reduce vector borne diseases (e.g., field drainage to reduce mud snail populations, which act as a vector for liver fluke) and also from medications released via livestock excreta. Reduced animal consumption might lead to increased pesticide use due to higher vegetable consumption while afforestation could increase tick populations increasing the risk of tick-borne diseases in grazing animals. Finally, increased N and phosphorus leaching is possible in early years of peatland restoration.

Several impacts could have positive or negative impacts, thus requiring careful implementation (Eory et al., 2017). Most were associated with reduced animal product consumption, afforestation, low emission storage and application of manure and peatland restoration where the variable impact was a result of mitigation or an aggregation of varied impacts. For example, low emission storage and application of manure uses various technologies that have different effects on the environment (Section 2.3). In other cases, effects greatly depend on the specific context of implementation (i.e., location, species, management). For example, covering the digestate from AD can mitigate increased NH_3 emissions or the location of forest and peatland projects may result in a positive or negative cultural impact.

The most uncertain mitigation options, i.e. those supported only by weak or moderate evidence, were reduced livestock product consumption, livestock health, optimal soil pH, and, to a lesser extent, low emission storage and application of manure and legumes (Eory et al., 2017). Further research could help in closing these knowledge gaps including soil pH impacts on water quality, soil quality and biodiversity; the influence of improving livestock health on pesticide use and human health; and effects of reduced livestock product consumption on agricultural, particularly on soil quality, biodiversity, animal health and welfare, employment, social and cultural impacts.

Many mitigation options can have co-benefits that can be promoted by integrated approaches in relevant policy areas. Eory et al. (2017) report that the wider impacts that had the highest number of variable co-effects were soil quality, flood management and water use, household income and human health with policy integration of these areas with GHG mitigation being key in maximising the net benefits.

3.5.4 Options for combining Net Zero and N Targets

Sections 3.5.2 and 3.5.3 show that there is much potential to achieve large reductions in N losses to the environment in general, while maximizing co-benefits with climate policy and minimizing trade-offs with negative environmental consequences. In terms of targets, at global scale there are now policy aspirations to halve nitrogen waste by 2030 enshrined in the UN Colombo Declaration on Sustainable N management and reducing nutrient losses by at least 50% by 2030 under the Farm to Fork Strategy of the European Union's Green Deal (see Section 3.1.2). It is clear that there is considerable potential to address these targets while achieving significant GHG and N loss reduction co-benefits related to the key N impacts on Water, Air, GHGs, Ecosystems and Soils (WAGES), but care needs to be taken to avoid trade-offs and integrated approaches are required at various scales.

For example, Figure 3.5.1 shows the components in the Sixth Carbon Budget for GHG savings from measures to reduce agricultural and land use emissions and Figure 1.5.4 illustrates how some are related to reductions in emissions and some are related to uptake or sequestration of carbon (i.e. afforestation, agro-forestry and hedges and peatland restoration made possible by reduced agricultural land demand due to diet shift and waste reduction). How these measures are distributed across the landscape and interact with each other will determine how synergies and tradeoffs discussed above (e.g., see Table 3.1.2) play out with each other. The SRUC consider that as land is released some trade-offs may nearly disappear, but that the synergies are spatially variable, and maximising them requires careful land use planning (pers. comm.).

For the UK as a whole this will require policies affecting air, water, ecosystems/biodiversity and soils to be closely aligned. For example, the UK Progress on Reducing Nitrate Pollution (Environmental Audit Committee, 2018) concluded that 'leaving the EU offers an opportunity for a joined-up approach, which aligns water, air and soil quality regulations and regulators' and 'goes further than existing standards wherever possible'.

3.6 Policy Recommendations

In this section we attempt to pull together the policy recommendations that emerge from our assessment of N losses to the environment and its impacts (Part 1), the key interventions to reduce N losses (Part 2) and options for innovative policy/regulatory frameworks in the UK and the four devolved nations (Part 3). The recommendations are based on principles of action (Section 3.6.1) and cover: UK Government actions at international level (Section 3.6.2), national actions by devolved administrations & UK Government for England (Section 3.6.3), agriculture policy actions (Section 3.6.4), biodiversity policy actions (Section 3.6.5) and future research (Section 3.6.6.)

3.6.1 Principles of action:

- Adopt a full-cycle approach to quantifying N use and losses, including transboundary imports and exports embedded in food, feed and fertiliser, as well as transboundary pollution via air and water;
- Integrate action to reduce all forms of N losses to the environment, maximising the co-benefits and minimising trade-offs;
- Integrate action to reduce N losses with action to reduce environmental losses of carbon, methane, phosphorus, pesticides and other forms of pollution;
- Action should be taken at every stage of the food supply chain from primary production to consumers in order to share responsibility for reducing waste and negative impacts in an equitable way;
- Quantify and raise awareness of how reduction of nitrogen losses across **W**ater, **A**ir, **G**HG, **E**cosystems/Biodiversity and **S**oils (WAGES) contributes to achieving multiple Sustainable Development Goals, benefiting people, climate and nature;
- Use these co-benefits to make the case for new policy, legislation and investment. For example:
    - action to reduce ammonia emissions and fine particulate matter (PM) for public health provides powerful leverage for action which will also reduce greenhouse gas emissions and biodiversity loss;
    - a focus on N_2O and climate impacts alone may not provide sufficient justification for action; demonstrating the co-benefits for public health and biodiversity (from related reductions in NH_3 emissions and nitrate pollution) may provide this;
- Optimise available resources to focus on major N flows and priorities for action to avoid delay and use of disproportionate resource on items of lesser importance (“don’t let the perfect be the enemy of the good”);
- Government action needs to be a package of legislation, compliance and enforcement, financial support, fiscal measures, collaboration with industry, and specialist advice to farmers and other stakeholders. A mix of regulation and advice is required that is easily understood and applied, and which aligns financial support to achievement of targets/regulations (above minimum expected baseline).

3.6.2 UK Government actions at international level

- Provide active support for the **UN Inter-convention Nitrogen Coordination Mechanism** (INCOM) and identify a UK Government National Focal Point for the UNEP Nitrogen Working Group under the UNEP Committee of Permanent Representatives;
- Support delivery of the **UNEA-4 Resolution** on Sustainable Nitrogen Management (UNEP/EA.4/Res.14);
- **Collaborate through INCOM on developing N budgets** as a dynamic policy tool at country and international levels. This should encourage mutual learning between experience of carbon and N budgets and improved literacy in interpreting the policy implications of N budgets, and a mechanism for national governments to report their N budgets (through INMS and eventually through INCOM);
- Assess the **suitability of available UK data sources** for feeding into the international N assessment and N budgeting;
- Support the **#Nitrogen4NetZero** campaign and promote global action on nitrogen into the outcomes of **UNFCCC CoP26**;
- Ensure that nitrogen pollution is addressed as key driver of biodiversity loss at the **CBD CoP15** and as part of the **post-2020 Global Biodiversity Framework**;
- Support the establishment of global and national NO_x and NH_3 emissions reduction goals for 2030, 2040 and 2050 through the **UNECE Gothenburg Protocol** review process. This should include accounting for NO_x emissions from soils in the Protocol's inventories;
- Continue to provide active support for the **Towards INMS** project to ensure that the INMS is established on a sustainable basis.

3.6.3 National actions by devolved administration & UK Government for England

- Establish **legally-binding targets** for 2030, 2040 and 2050 for reducing all forms of N emissions to air and water. These could support the global target of halving N waste by 2030;
- Establish an **integrated and comprehensive strategy** for reducing N pollution through policy and legislation across government. This could support delivery of existing goals and targets, including for net zero GHG emissions, sustainable development, air and water quality and biodiversity, maximising co-benefits and minimising trade-offs. It should include mechanisms for monitoring, reporting and review;
- Commission **national N budgets** as a dynamic policy tool (similar to carbon budgets) to provide evidence of current N flows and impacts, to inform target-setting for reducing pollution and to shape future policy and strategy;
- Commission **independent analysis of the economic costs and benefits** of reducing N pollution at a national level and for farm businesses, exploring how a circular economy approach to resource use and fiscal measures can be applied to N resource management fairly and effectively;

- Establish a **cross-government working group** with representatives from relevant departments and teams to develop and deliver the national N strategy, maximising co-benefits and minimising trade-offs between different potential actions;

- Initiate further research and programmes of action on **awareness-raising and stakeholder engagement** on N. For example, public information on air quality often contains little or no reference to ammonia emissions or the impacts of air pollution on biodiversity. Increased awareness will also help build understanding of the importance of this issue within the food manufacturing and retail sectors, to help share the costs of action throughout the food supply chain and across public and private sectors;

- The full range of devolved and reserved policy levers must be used together (CCC, 2020d). Delivering the transition in the devolved nations will require effective collaboration between the devolved and UK governments, and a strong policy framework that works across all levels of government. For example, in Wales, policy areas relevant to decarbonisation that are partially or fully devolved to the Welsh Government include agriculture and land use, planning, transport, energy efficiency for new-builds, and waste.

3.6.4 Agriculture policy actions

- Policy and regulation to reduce N losses from agriculture should be consistent with the transition to more **environmentally-sustainable land management**, taking an integrated approach to improving air, water and soil quality, biodiversity and ecosystems. A land-sharing approach through, for example, more extensive livestock grazing on permanent semi-natural grasslands and less intensive field crop production is more consistent with this than the land-sparing/agricultural intensification approach;

- Agricultural policy actions must be integrated with a **national food and farming strategy** to ensure that sustainable food producers are supported and protected from unfair practices and trading rules throughout the supply chain;

- A **package** of legislation, financial support, fiscal measures, collaboration with industry, and specialist advice should be devised as appropriate to each devolved nation's agricultural system, including:
    - Integrated **baseline regulation** applicable to all farm businesses and other land managers. This should:
        - cover N losses to air (inc. GHGs) and water;
        - require nutrient management planning and use of low-emission techniques by all farm businesses;
        - require the use of low-emission livestock housing, slurry stores and other infrastructure on all new farm developments and (phased in over time) existing farm operations;

- Strengthened **environmental permitting** system for large and indoor livestock units and slurry/waste contractors including:
    - Regulation of wastes from intensive operations;
    - Lower thresholds for intensive pigs & poultry units;
    - Introduction of permit requirements for intensive beef & dairy units;
    - Introduction of permit requirements for slurry contractors (following the precedent from transport providers and emissions standards);
- Strengthened **spatial planning system** for new developments including on farms to ensure:
    - full compliance with all relevant legislation and regulation;
    - alignment with local development plans;
    - high standards of nutrient management planning and waste management planning;
    - assessment and control of cumulative impacts of pollution sources within the local area are taken into account;
    - protection of biodiversity, ecosystems and public health & wellbeing;
- Integrated, tailored **advice & training** to farm businesses on nutrient management planning to reduce nitrogen losses to air and water, and improve soil health;
- Adequate **government funding and political support** to ensure effective compliance and enforcement with regulation, to administer relevant schemes and to provide advice and training to local planning authorities, farmers and other stakeholders;
- **Financial support** from government and private sources (e.g., water companies) including:
    - payment for environmental services above and beyond regulatory requirements, such as the Environmental Land Management Scheme in England and Sustainable Farming Scheme in Wales;
    - grants for capital costs, such as precision technology, low-emission spreading equipment and more efficient livestock housing and fertiliser storage (e.g., the Slurry Investment Scheme in England);

- **Fiscal measures** – such as a tax or levy on artificial N fertiliser.

Such measures should be explored in an independent economic analysis as outlined above for national policy actions.

3.6.5 Biodiversity policy actions

- Address the impacts of N pollution by air and water at a strategic level in **biodiversity policy** and strengthen the capacity of statutory nature conservation and environmental agencies to take action;

- Introduce a **targeted site-based programme** to reduce emissions close to the most sensitive and vulnerable designated sites and other sensitive priority habitats, such as Site Nitrogen Action Plans and Diffuse Water Pollution Plans in England and Wales which have been developed at a small number of Special Areas of Conservation (SACs) to date;

- Extend and adapt the Local Air Quality Management (LAQM) system and measures such as Clean Air Zones to reduce ammonia emissions and address the impacts of air pollution on local biodiversity and ecosystems, as well as public health;

- Strengthen **environmental monitoring and development control for** pollution sources, in particular close to sensitive habitats. For example, there is currently no information about where, when or how much slurry, manure or litter is stored or spread on land near sensitive habitats;

- Integrate available data on atmospheric N concentration, deposition/diffuse pollution levels and impacts into the monitoring, assessment and management of sensitive habitats in **designated sites and other sensitive priority habitats**. This should then inform and enable site managers to:

    - Identify and monitor the sources and impacts of atmospheric and waterborne N input to the site;

    - Engage local land managers to help reduce N pollution through Site Nitrogen Action Plans (SNAPs);

    - Manage sensitive habitats to reduce the impacts on biodiversity, such as by:

        - controlling dominant species that are adapted to higher N levels;

        - removing excess N from the system;

        - techniques for achieving this include: grazing, cutting, burning, fertilisation, liming, hydrological management, scrub and tree management, and disturbance used to improve habitat suitability for vulnerable species. (However, many management techniques involve trade-offs with other conservation priorities (e.g., carbon emissions from burning), reinforcing the point that there is no substitute for emissions reduction at source);

    - Plan tree-planting schemes to help intercept N emissions before they reach species-rich habitats. This can be done at a farm level (e.g., tree belts around livestock housing), on the borders of nature conservation sites or at a wider strategic level. Trees can help to disperse, dilute and recapture atmospheric ammonia. However, this should be a secondary action in addition to emissions reduction at source;

- Control emissions from heavily-stocked grazing of cattle and evaluate the impact of grazing ruminants on land close to sensitive habitats including SSSIs;

- **Commission research** into the impacts of N pollution on biodiversity and ecosystems at a UK/country level, including ecosystem recovery following reduction of pollution and the impacts of terrestrial pollution on taxa other than wild plants and fungi, such as pollinators and birds.

3.6.6 Recommendations for future research

Considering the principles for action and policy recommendations above, studies are needed to:

- analyse the least integrated/effective parts of N management and regulation in the UK and devolved nations and make recommendations to replace them with something more coherent, maximising the co-benefits and minimising trade-offs;
- conduct a full-cycle approach to quantifying N use and losses, including transboundary imports and exports embedded in food, feed and fertiliser, as well as transboundary pollution via air and water;
- develop N budgets linked to impact thresholds to quantify and raise awareness of how reduction of N losses across Water, Air, GHG, Ecosystems/Biodiversity and Soils (WAGES) contributes to achieving multiple Sustainable Development Goals, benefiting people, climate and nature;
- identify the most likely suite of low carbon measures that will be taken up by farmers in the UK and devolved nations and assess qualitatively and quantitatively (where possible) the full chain implications for N losses and the required N management guidance, especially for manure management;
- conduct an independent economic assessment of the costs and benefits of N pollution, including circular economy considerations and options for action. Such an assessment should consider the range of possible fiscal measures in detail and make recommendations to government that spreads responsibility fairly across supply chains, including incentives for farmers to use more innovative practices;
- assess integrated actions to reduce N losses with action to reduce environmental losses of carbon, methane, phosphorus, pesticides and other forms of pollution for major farming types while protecting biodiversity, considering dependencies, including N, P, pesticides etc and housing gains lost in the field when manure is applied;
- Involve farmers in enhancing knowledge on how farm businesses can be made more environmentally and financially viable and disseminate knowledge widely using peer-to-peer networks. Including developing guidance on how to

minimize losses in different conditions and implications of moving to more agro-ecological practices.

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(Note: All urls accessed November 2021)

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