Chapter 2. Proposed approach to address nitrogen pollution

This chapter proposes a three-pronged approach to cost-effectively respond to nitrogen pollution. First, to better manage the risks of air, soil and water pollution and associated ecosystems through a detailed analysis of the nitrogen pathways, the so-called "spatially targeted risk approach". Second, address the steady increase in nitrous oxide concentrations in the atmosphere through a "global risk approach". Third, take into account the uncertainty of cascading effects and anticipate potentially significant long-term impacts through a "precautionary approach".

As we have seen in Chapter 1. , there are specific nitrogen pathways for each environmental medium and the relative impacts on water quality, air quality, greenhouse balance and ozone layer, ecosystems and biodiversity, and soil quality ("WAGES"). The nitrogen cascade described in Chapter 1 "superimposes" these media-specific pathways by including the possibility for a nitrogen atom to pass from a medium to the other, and from one impact to another. How should environmental policy handle this?

- Should environmental policy stick to media-specific impacts,[1] as it already does when managing environmental risks?
- Should environmental policy add a more "precautionary" dimension that takes into account the potential for and uncertainty of cascading effects across media and anticipates potentially large impacts in the longer term?

There is only one form of nitrogen that has a global impact and that is nitrous oxide (N_2O), which impacts on both global warming and the stratospheric ozone layer. While the impacts of accumulating N_2O will differ regionally, the problem can only be tackled globally since it is a long-lived and well-mixed gas in the atmosphere. The other forms of nitrogen are labile (i.e. they move, change form and combine easily with other pollutants); their impacts on air, soil and water quality will therefore tend to be more local or regional. Policies to tackle these different scales of impacts are therefore needed. As we will see in this section, N_2O management calls for global action because of the global coverage of its impacts whereas a more spatially targeted approach is more appropriate for other nitrogen risks.

On the other hand, the uncertainties of the cascade call for a complementary action (in part, following OECD, 2016). However, a cost-effective environmental policy requires prioritising the management of well-documented risks over the management of such uncertainties (Table 2.1). First, there is a need to manage risks by better understanding the pathways of nitrogen between sources and impacts, including the contributions of selected sources at different scales and times. Second, in the absence of strong evidence on pathways, there is a need to prevent nitrogen entering the environment by developing strategies addressed to all sources (on the basis of the most cost effective means) to reduce them. What differs between the two approaches is whether nitrogen pathways leading to specific impacts are sufficiently well understood (or predictable) or are largely uncertain.

Table 2.1. A three-pronged approach to address nitrogen pollution

Spatially targeted risk approach	Global risk approach	Precautionary approach
	Nitrogen forms	
All but N_2O	N_2O	All
	Pathways	
Impact-Pathway Analysis (IPA)[1]	Global exposure	Nitrogen cascade
	Focus	
Media-specific (air, soil, water)	Greenhouse effect, ozone layer	"Systemic" (all-media)
	Intervention points (scale)	
Risk specific	Global	National (based on monitoring the country's total nitrogen load)
	Policy effectiveness	
High (tailored to risk)	High (tailored to risk)	Low (only targets the nitrogen load)
	Policy priority	
According to policy objectives	According to policy objectives	When IPA is not possible

1. Impact-Pathway Analysis (IPA) is an evaluation of the pathways that generate an impact (including through modelling) to estimate the expected benefits of possible emissions changes.
Source: OECD Secretariat.

Section 2.1 will set out a proposal for how to implement the spatial management of environmental risks related to forms of nitrogen other than N_2O as well as a global approach to management of N_2O. Section 2.2 will examine how to approach precautionary management of uncertainty related to the nitrogen cascade.

2.1 The risk approach

2.1.1 The different nitrogen risks

Countries have set acceptable levels of health risks for nitrogen dioxide (NO_2) and nitrate (NO_3^-) concentrations in air and water, respectively, as well as for concentrations of ground-level ozone (GLO) and particulate matter (PM) (to both of which nitrogen is a precursor). Any emission reduction or any practice to achieve it arises from these acceptable levels of risk.[2]

To address ecosystem risks, critical loads and critical levels have been estimated for the terrestrial ecosystems (forests, wetlands) and are used to regulate nitrogen oxides (NO_x), ammonia (NH_3) and GLO emissions under the Convention on Long-range Transboundary Air Pollution (LRTAP) and its protocols. For example, in some countries Total Maximum Daily Load (TMDL) is used to calculate the maximum amount of nitrogen allowed to enter aquatic ecosystems (lakes, coastal areas) in which the acceptable level of water quality is not met.

As regards climate change, the Paris Agreement established a level of global warming risk[3] acceptable to the parties and, therefore, indirectly, an acceptable level of climate forcing due to greenhouse gases (GHGs), including N_2O. All other things being equal, to stop global temperatures increasing, emissions of long-lived GHGs must eventually fall to zero on a net basis. The Montreal Protocol on Substances that Deplete the Ozone Layer sets an acceptable risk of depletion of the ozone layer, but N_2O is not one of the ozone depleting substances covered by the Protocol at present.

Beyond these regulated risks, the risk of undermining the resilience (of exceeding the coping capacity) of nitrogen sinks such as terrestrial biomass change and marine sediments should also be addressed. For example, if the land sinks were to be saturated, the risk of nitrogen export from watersheds to coastal waters and the associated impacts would be much higher. So far, we do not fully understand the resilience of ecosystems to increased nitrogen loading. Although the data are highly uncertain, the storage of nitrogen in soils and trees[4] appears to represent only a small part of the annual nitrogen input into the land sinks, most of which seems to be denitrified (USEPA-SAB, 2011),[5] thus in part (i.e. the N_2O portion) contributing to global climate and ozone risks. There is an even greater uncertainty about the data when it comes to marine sediments. In particular, the diversity of possible cycling mechanisms have hindered the ability to quantify marine organic matter transformation, degradation and turnover rates (Walker et al., 2016).

Cost-benefit analysis (CBA) defines the risks to be managed as a priority (in part, following OECD 2008). Nitrogen risk management can make an important contribution to social welfare (for example by protecting the natural bases of production and improving human health). However, it can also entail significant economic costs. It is therefore important to carefully consider whether the additional benefits of environmental improvements, and the additional costs to society of achieving these improvements, are justified.

2.1.2 Deepening pathway analysis to better manage risks of nitrogen pollution

Current nitrogen management policies have often focused on a specific impact without consideration of biogeochemical pathways[6] contributing to this impact (USEPA-SAB, 2011). Yet, a better knowledge of pathways can improve the cost-effectiveness of risk management by better identifying the points of policy intervention (i.e. by better allocating emission reduction across different nitrogen sources). For example, the reduction of nitrogen impacts in estuaries may greatly benefit from a stricter control of atmospheric deposition in the airshed[7] as well as stricter runoff controls in the watershed. This is why significant efforts have been made to join-up air and water management in the Chesapeake Bay region (Linker et al., 2013).

The Impact-Pathway Analysis (IPA) is an evaluation of the pathways that generate an impact (including through modelling) to estimate the expected benefits of possible emissions changes (in part, following OECD, 2018). IPA differs from damage assessment, which assesses an impact at a given point in time, without explicitly considering how that impact was generated (ibid). IPA recognises that nitrogen can move between environmental media (air, water, soil and biota) as it travels along a pathway from one or more sources to a receptor.

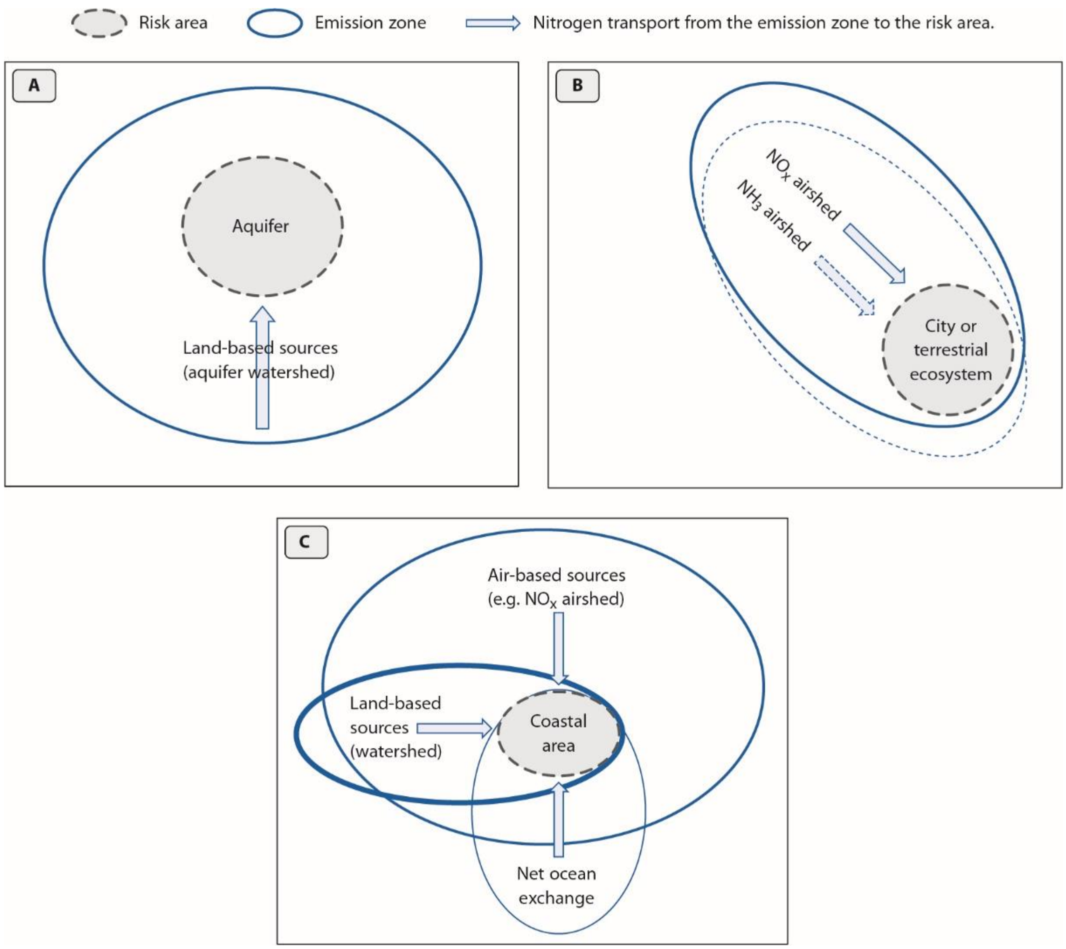
This section discusses the importance of deepening the analysis of pathways to adapt risk management to the specificities of nitrogen pollution. Chapter 3. presents case studies of IPA and its translation into policy-making.

A four-step approach is proposed to implement the IPA:

- First, identify the nitrogen sources of relevance to the impact and delineate the different nitrogen emission zones that converge towards the risk area
- Second, calculate the marginal abatement costs for reductions in release of the different nitrogen forms, for that, estimate the potential for new or additional emission reductions in each emission zone.
- Third, compare the cost-effectiveness of emission reductions for all sources of risk in the different emission zones
- Fourth, estimate the marginal ancillary benefits of reducing nitrogen emissions in the different emission zones (i.e. the avoided damages along the pathways of nitrogen to the risk area).

The first step is to delineate the different nitrogen emission zones that converge towards the risk area (Figure 2.1). The advance of knowledge on pathways will allow consideration of new nitrogen sources (e.g. located further back along the pathways) compared with the absence of IPA, where the only points of intervention to manage the risk of pollution are the sources whose link with the impact is already well established.

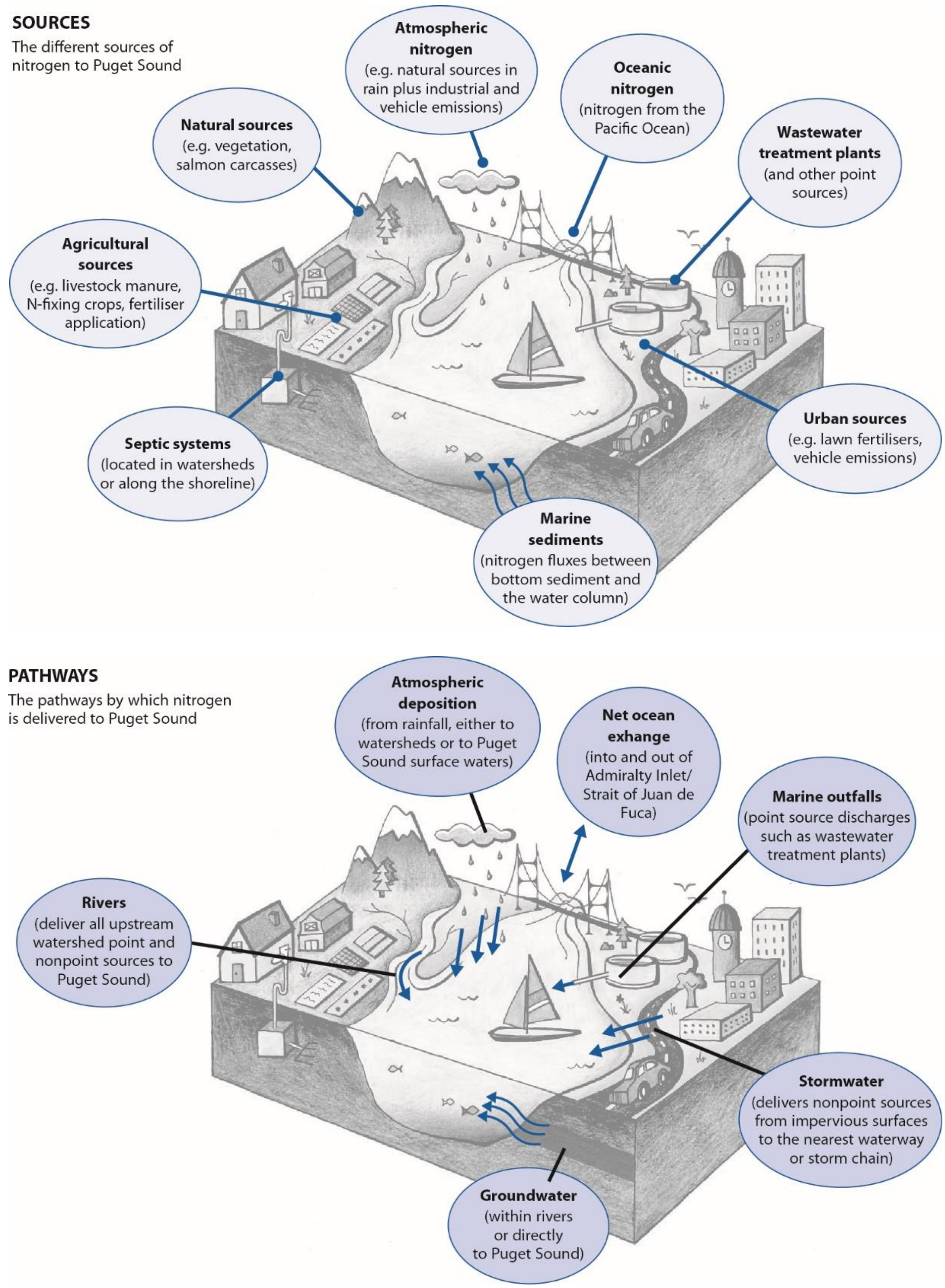
Figure 2.1. Nitrogen emission zones for different pollution risks



The three panels represent three types of risk:
(A) contamination of an aquifer by nitrate (NO_3^-)
(B) atmospheric pollution of a city or terrestrial ecosystem
(C) nitrogen pollution of a coastal area.
Source: OECD Secretariat.

The Puget Sound dissolved oxygen study is a good example of the delineation of nitrogen emission zones in the case of coastal zone nitrogen pollution. The study identified the different sources of nitrogen that could contribute to the low dissolved oxygen content in the Sound and assessed their relative contributions through pathway analysis (Figure 2.2).

Figure 2.2. Sources and pathways of nitrogen in South and Central Puget Sound



Source: Roberts and Kolosseus (2014).

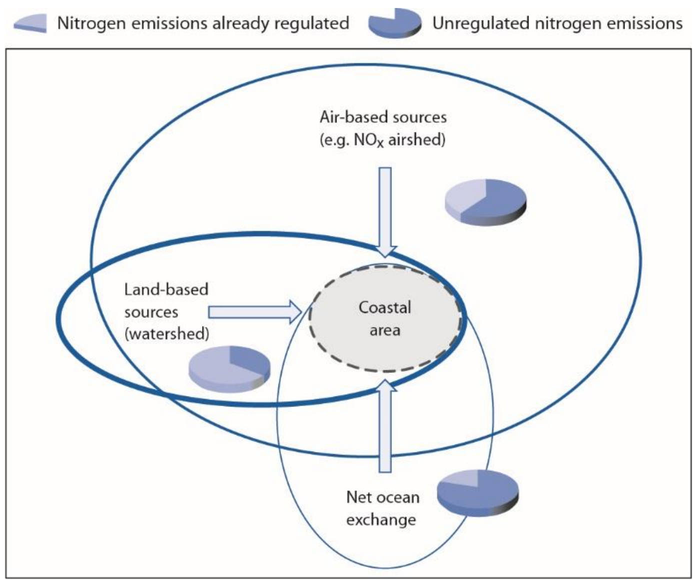
Another example relates to urban air pollution. In March 2014, Paris suffered a major peak of particle pollution. The scientific assessment revealed that half of the PM_{10} were ammonium nitrate particles formed by a combination of NO_x and NH_3. The NO_x airshed corresponds to the agglomeration (the main source being car traffic). The NH_3 airshed is much larger as emissions come from agricultural activity in north-western France and beyond (see Case Study 1 of Chapter 3.).

The management of air pollution risk in the Santiago Metropolitan Region of Chile is another good example of a policy that takes into account, at least in part, the emission zones of the various pollutants involved.[8] In January 2017, Chile begun implementing a tax on emissions of carbon dioxide (CO_2), PM, NO_x and sulphur dioxide (SO_2). The tax is levied on large stationary sources, particularly fossil fuel-based electricity plants, and takes into account the size of the population affected by the pollution, which is a positive feature. However, the formula considers only the population of the municipality where the source of the emission is located and not, as originally planned, the entire relevant airshed. Since air pollutants can be transported and deposited over relatively large areas, it would have been preferable to take the atmospheric dispersion effects (and the total population exposed) into account when calculating the tax for each polluter (OECD/ECLAC, 2016). Ideally, it would also have been useful to consider other impacts related to deposition, such as exposed ecosystems in the airshed.

IPA is also useful for delineating the risk area. For example, in the German Land of Baden-Württemberg high-resolution deposition analysis and actual monitoring on the ground allowed revealing exceedances of critical loads for eutrophication on major parts of the Land, which previous (lower-resolution and modelling) analysis did not (see Case Study 1 of Chapter 3.).

The second step is to estimate the potential for new or additional emission reductions in the different emission zones, taking into account ongoing emission reduction measures to avoid overlap (Figure 2.3). Cost-effectiveness analysis (CEA) should determine whether to continue (or intensify) existing emission reduction or whether it is preferable to intervene on other emission sources. Denmark is a good example of implementing CEA for managing the risk of eutrophication of coastal waters, lakes and watercourses from agricultural sources of nitrogen. The assessment includes not only an analysis of measures prior to implementation but also a mid-term and a final (ex-post) evaluation (Jacobsen, 2012). For example, the most cost effective measures in the second Danish Action Plan for the Aquatic Environment (1998-2003) were the requirements for catch crops and wetlands, increased utilisation of animal manure and improved feeding practices (Jacobsen, 2004). The least cost-effective measures were land set-aside and increased areas with grass as well as the requirement for reduced animal density (ibid).[9]

Figure 2.3. State of nitrogen regulation in emission zones



Note: The regulated and unregulated portion of pies is illustrative only.
Source: OECD Secretariat.

The third step in IPA is to compare the cost-effectiveness of emission reductions across all sources of risk (including those already regulated) in all emission zones. This step, which is essential for ensuring coherence of interventions, is still not widespread in the implementation of nitrogen policies. The previous example shows that nitrogen sources other than agriculture – such as industry, wastewater, atmospheric sources and net nitrogen exchanges with marine waters – have not been included in assessing the cost-effectiveness of eutrophication risk management measures in Denmark.

IPA seeks to link the pollution risk to its sources and ensure coherence with ongoing policies to reduce emissions in each source. For example, efforts to reduce NO_x emissions at the national level should be considered in risk areas subject to NO_x deposition. The aim is to select emission sources for which abatement is the most cost-effective, including a further reduction for already regulated nitrogen sources.

To a certain extent, the Baltic Sea Action Plan (BSAP) is a good example of seeking such policy coherence to tackle the risks of eutrophication. Based on scientific advice, the BSAP determines the maximum allowable input (MAI) of nitrogen that the Baltic Sea is thought to be able to tolerate. Each party is then allocated a nitrogen reduction target based on the MAI and according to its watershed area. Finally, atmospheric nitrogen reduction efforts under the LRTAP Gothenburg Protocol are deducted from each party target.

The cost-effectiveness of reducing emissions depends not only on the source of the emission and the form in which the nitrogen is emitted, but also on the place of emission (within the emission zone) and how nitrogen is transported to the risk area. The amount of nitrogen that actually contributes to the risk must be estimated. In fact, only a part of the nitrogen emitted in an emission zone may end up in the risk area.

Typically (according to the US Environmental Protection Agency (USEPA) definition), approximately 75% of the emissions in the airshed of a watershed are redeposited into the watershed, be it NO_x or NH_3. For land-based sources, analysis of nitrogen flows at the watershed level (Billen et al., 2013) suggests that only about 30% of net anthropogenic nitrogen inputs (or NANI)[10] reach the coastal waters. The remaining 70% is either retained by the terrestrial biosphere or denitrified en route, with the relative shares being partly controlled by climatic conditions (Howarth et al., 2012).[11]

The use of models of nitrogen transport in air, land and water can clearly inform cost effectiveness assessments and policy formulation. For example, the U.S. Geological Survey has developed a groundwater vulnerability and assessment (GWAVA) model to relate groundwater NO_3^- concentration observations to spatial attributes representing nitrogen sources and NO_3^- transport and attenuation. GWAVA predicted NO_3^- concentrations in groundwater for the entire United States.[12]

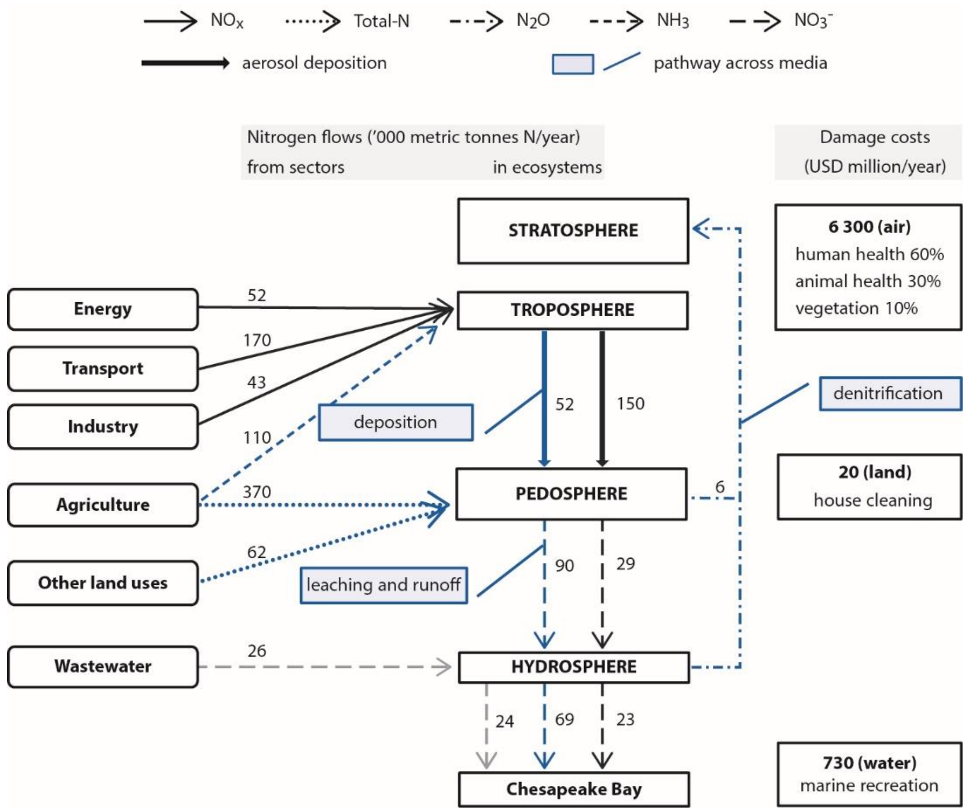
In Denmark, TargetEconN models the attenuation of nitrogen loss from the root zone to the coast with a view to better manage the diffuse pollution of water by nitrogen. This attenuation depends on hydrology, soil types, slope, vegetation and weather conditions. The model was developed to support watershed-scale decision-making. It integrates biophysical and economic modelling. The results clearly indicate that there are large differences in cost-effectiveness between uniform reduction measures and targeted measures, and that more targeted regulation taking account of heterogeneity in both abatement costs and pathways can enhance cost-effectiveness in addressing nitrogen risks (Hasler, 2016).

A third example is NTRADER, which was developed in New Zealand to model the transport of nitrogen from farmland to lakes, including groundwater lag times. NTRADER integrates the outputs of several models, at the farm scale (nitrogen leaching and attenuation model and economic model) and at the catchment scale (nitrogen transport model). It models both nitrogen generation and transport and the economics of “cap and trade” schemes. NTRADER has been useful in managing the risk of water pollution of New Zealand lakes by nitrogen from pastoral lands (Cox et al., 2013).

Case Study 2 of Chapter 3. analyses the management of nitrogen pollution in Lake Rotorua (New Zealand). It also refers to the different models used in the Chesapeake Bay Programme, such as the Community Multi-scale Air Quality model for atmospheric deposition, a watershed transport model, and the Water Quality and Sediment Transport Model for sediment transport.

The ultimate step of the IPA is to estimate the ancillary benefits (avoided damage) of reducing nitrogen emissions in the different emission zones (i.e. taking into account the nitrogen pathways towards the risk area). Indeed, evaluating the benefits of mitigating a tonne of nitrogen released needs to consider the damage avoided in all of the ecosystems through which that tonne would cascade (Moomaw and Birch, 2005). Figure 2.4 provides an example of such “economic cascade” in the Chesapeake Bay watershed.

Figure 2.4. Nitrogen flows and cascading damage costs in the Chesapeake Bay watershed

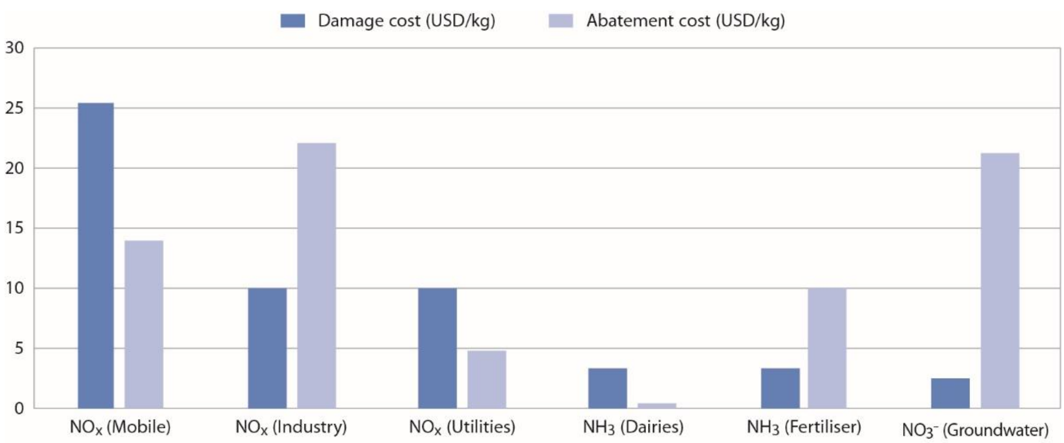


Note: Arrow colour indicates the origin of nitrogen flows: black for air, blue for land and grey for wastewater. Data are illustrative only. Recent estimates indicate that agriculture accounts for 42% of nitrogen mass flows emitted in the Chesapeake Bay watershed, followed by discharges from wastewater (34%) and atmospheric deposition from fossil fuel combustion (24%). These estimates are based on nitrogen monitoring of (i) agriculture, (ii) urban runoff, wastewater and combined sewer overflow, and (iii) atmospheric deposition to forest, non-tidal water and tidal water (www.chesapeakebay.net/indicators/indicator/reducing_nitrogen_pollution, accessed 20 March 2016).
Source: Adapted from Birch et al. (2011).

As can be seen in Figure 2.4, damage costs from air emissions are much larger in the watershed than those from the whole of land and water emissions (despite being smaller nitrogen flows). This reflects human health benefits from reduced air pollution. In addition, since some of the atmospheric nitrogen is found in soils and water, the damage costs are increased accordingly. Thus the reduction of Chesapeake Bay damages from nitrogen (including freshwater and estuarine impacts) may benefit more from a stricter control of air pollution than from stricter water pollution controls. Indeed significant efforts have been made to integrate air and water management in the Chesapeake Bay region (Linker et al., 2013).

Given that the estimated health benefits of cleaner air are much greater than the estimated environmental and health benefits of cleaner water, reducing NO_x and NH_3 emissions will often prove more beneficial than reducing other forms of nitrogen emissions. This is confirmed, for NO_x, by a recent study of nitrogen flows in California's San Joaquin Valley (Figure 2.5). However, this study does not consider the role of NH_3 in the formation of secondary particles ($PM_{2.5}$), which has a significant impact on people's health.

Figure 2.5. Costs of damage and reduction of different forms and sources of nitrogen in the San Joaquin Valley, California



Source: Horowitz et al. (2016).

Assessment of damage avoided in emission zones could consider willingness to pay to improve health and the social cost of carbon. This is what Keeler et al., 2016 did to estimate the social costs of applying a 1kg of fertiliser in Minnesota (Box 2.1). Important assumptions for such valuation are the choices of the value of statistical life (e.g. for premature deaths due to fine particles ($PM_{2.5}$) formation) and the social cost of carbon (for climate related damages). There is also a need to consider the willingness to pay retrospectively (and not only preventively) as it will probably be higher for those who have been drinking contaminated water or breathing polluted air without knowing it.

It can be assumed that each ecosystem responds differently to changes in nitrogen load, depending on the type of ecosystem and local conditions, and that the lag time between emission and impact is specific to each ecosystem. The question arises as to whether a discount factor should be introduced to account for these lag times, as they can significantly influence estimates of damage costs in the IPA (in part, following OECD, 2018). The monetised benefits could be significantly lower, depending on the length of the lag and the discount rate used. Conversely, assuming that the lags do not exist, the value of the benefits of nitrogen reduction will generally be overestimated, perhaps significantly (and critically from the point of view of CBA). For example, Cox et al., 2013 estimate that it takes between 0 and 127 years for nitrogen emitted into the Lake Rotorua watershed to reach the lake via groundwater, depending on whether the source is near or far from the lake (see Case Study 2 of Chapter 3.).

Box 2.1. The social costs of a kilogram of nitrogen fertiliser

A recent study demonstrated that the social cost of a kilogram of nitrogen fertiliser applied in Minnesota varies between less than a tenth of a cent to more than USD 10 depending on the site, the nitrogen form and so-called "end points of interest", that is, whether the impact is about GHG emissions (N_2O), air pollution ($PM_{2.5}$, and indirectly its precursors NO_x and NH_3), or groundwater contamination (NO_3^-) (Keeler et al., 2016).

Following Keeler et al., 2016, it is possible to estimate the social cost of NO_3^- groundwater contamination caused by nitrogen fertiliser. Cost is obtained by multiplying the number of known and predicted contaminated wells by the population using these wells and an average cost of well contamination per household. The latter is estimated on the basis of a survey of well owners facing excessive nitrogen in their wells; it includes the costs of building a new well, buying bottled water or investing in an NO_3^- disposal system. The authors have prepared risk maps for NO_3^- contamination in the various counties of Minnesota by combining, at the county level, data on the three risk factors of agricultural expansion (likelihood), soil characteristics (vulnerability), and population relying on groundwater (exposure).

Keeler et al., 2016, also estimated the social cost associated with NH_3 and NO_x emissions from nitrogen fertiliser, based on their contribution to premature deaths due to $PM_{2.5}$ formation. The cost is obtained by multiplying the number of deaths due to $PM_{2.5}$ downwind of NH_3 and NO_x emissions by an average cost of premature death. An emissions-to-health impact model for $PM_{2.5}$ – the Intervention Model for Air Pollution (InMAP) – calculates the former, while the latter reflects the willingness to pay of people in the United States for a reduction in their mortality risk. InMAP simulates the transport, transformation, and removal of emissions and then calculates mortalities based on resulting $PM_{2.5}$ concentrations, epidemiological information and population census data. NH_3 and NO_x emissions were derived by applying emission factors – 0.08 for NH_3 and 0.005 for NO_x – to the reported on-farm nitrogen inputs in each county. The authors have prepared risk maps for NH_3 and NO_x emissions by modelling damages that occur downwind of their emissions, even beyond the borders of Minnesota, and then allocating the damages back to the county where the NH_3 and NO_x emissions took place.

Finally, Keeler et al., 2016 evaluated the social cost associated with N_2O emissions from nitrogen fertilisers, based on climate related damages. The cost was obtained by converting N_2O into equivalent CO_2 emissions and applying an estimate of the social cost of carbon. This amounts to multiplying the estimated social cost of carbon for CO_2 by 395 (the long-term radiative forcing difference between CO_2 and N_2O). N_2O emissions were derived by applying an emission factor of 0.01 to agricultural nitrogen inputs in each county. For all nitrogen forms (NO_3^-, NH_3, NO_x and N_2O), the social cost per unit of nitrogen was obtained by dividing its total social cost by the agricultural nitrogen inputs in each county.

Beyond IPA, an important question to be clarified for cost-effective management of pollution risk is whether the issue of nitrogen to be addressed is a single-pollutant issue or a multi-pollutant problem. Many of the nitrogen-related impacts fall into the second category and the management of non-nitrogen pollutants may be a priority in some cases. For example, policy to reduce algal blooms may need to address phosphorus (P) first in freshwater lakes, where P limitation is the norm. In these P-limited lakes, it is more effective to initially control the algal growth limiting factor (P intakes) than to seek to reduce excessive nitrogen pollution. The opposite is true in coastal systems, where nitrogen is generally the limiting factor rather than P. The reason is that there is a large amount of denitrification in the coastal zone, so that the equilibrium often passes to an excess of P. Management of both nutrients may sometimes be necessary. This is the case in freshwater lakes, for example, when the reduction of nitrogen inputs improves the composition of algae by reducing harmful cyanobacteria. On the other hand, the reduction of nitrogen inputs in freshwater lakes is ineffective when it leads to an increase in nitrogen-fixing organisms (such as cyanobacteria). Thus, the decision to manage one or both nutrients must be determined on a case-by-case basis.

Focusing on nitrogen pollution only, governments would run the risk of incorrect policy recommendations. IPA is pollutant-specific. It does not seek prioritisation of policy action between nitrogen and other pollutants or precursors. Such prioritisation should ideally entail undertaking pathway assessment(s) for the other pollutant(s) or precursor(s). On the other hand, IPA helps to prioritise between environmental policies, such as between water policy and air policy to manage nitrogen pollution in coastal waters (see Chapter 3. ,Section 3.2.1).

2.1.3 Feasibility of the risk approach

Beyond the criteria of economic efficiency, the "feasibility" of IPA is of course essential for implementation and effective operation of the risk approach (see Chapter 5. for a detailed analysis of the feasibility criteria). This is particularly the case for public acceptability, including the agreement of stakeholders to delineate risk areas and emission zones. For example, in the United States, it would have been appropriate to delineate the entire Willamette River Valley (as a hydrogeological entity) to manage the risk of groundwater NO_3^- contamination in the Willamette Basin, given the high connectivity between the river and shallow groundwater. In practice, only the southern part of the valley could be declared a risk area in the name of the general interest (area used by public systems for the abstraction of drinking water) (see Case Study 2 in Chapter 3.).

Administrative feasibility issues may also arise. Indeed, the nitrogen pathways do not follow the administrative boundaries, far from it. For example, few countries have put in place governance at the level of the airshed (this is more the case for watersheds). The management of emissions in atmospheric basins that have been delineated on a scientific basis can therefore be problematic, as shown by the above example of the Chilean tax on CO_2, PM, NO_x and SO_2 emissions. The Chilean tax takes into account the population of the municipality where the source of the emission is located and not the entire population affected by the pollution (that is, the population of the airshed).

Air and water pollution can extend beyond national borders. In this case, watershed-level or airshed-level emissions management may be problematic in the absence of an international convention or agreement governing transboundary pollution such as LRTAP, for example. IPA has an important role to play in facilitating the adoption of new international provisions to manage transboundary nitrogen-related pollution. The responsibilities and risks of each party could indeed be better identified by the IPA's identification of the emission zones and risk areas.

The feasibility of IPA also raises the issue of its own cost. The preparation of an IPA can involve complex analytical work and require collaboration with a wide range of actors/public authorities, and thus be extremely costly. As a general principle, the level of IPA sophistication should match the expected level of nitrogen pollution risk. When major impacts are at stake, a precise and detailed IPA is required. On the other hand, when risk levels are low, a basic IPA can be used.

2.1.4 The case of nitrous oxide (N_2O)

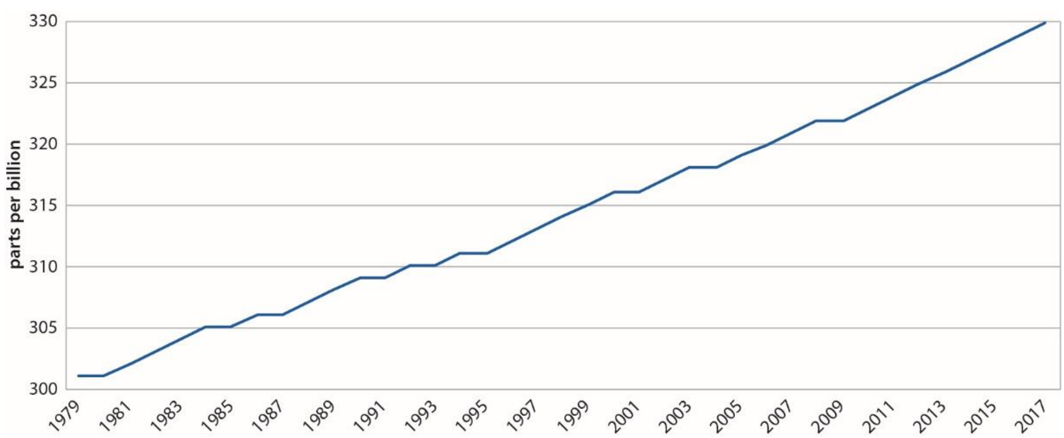
IPA does not apply to N_2O. As we saw, for a given risk (risk area), the IPA identifies the sources of emission to be managed in priority among a given number of sources, those which are located in the emission zones. In the case of N_2O, the risk area (be it the risk of a greenhouse effect or the risk of depletion of the ozone layer) is global[13] and delineating emission zones, if at all possible, would be of no use for the purposes of CEA (or CBA). Indeed the more sources to compare, the more likely it is to find the one for which emission reduction is the most cost-effective. In other words, as many sources of N_2O as possible should be identified, wherever they are in the country, so as to monitor them and compare the costs – and if possible the ancillary benefits – of reducing their emissions. This requires a global approach.

Such global approach aims to manage only the sources of N_2O unlike a precautionary approach, as described in Section 2.2, which covers all nitrogen forms. As we have seen (Chapter 1.), agriculture accounts for the majority (about 2/3) of anthropogenic emissions of N_2O and N_2O emission data by source are available (see FAOSTAT for example). However, because of the complexities of farm biological and management systems, accuracy of N_2O emission data is limited (PCE, 2016). For example, current research in New Zealand is looking at improving a modelling tool[14] to enable more accurate measurement of emissions from individual farms and thus better estimation of the impacts of different farm management practices on N_2O emissions (ibid).

It is therefore necessary to better understand the nitrogen pathways in the soil to better identify the sources of N_2O and the level of their emissions.[15], This involves estimating the share of incomplete denitrification (that is to say the ratios N_2O/dinitrogen) (see Section 1.4.5). The difficulty of obtaining accurate N_2O emission data arises in part from the complexity of the biological systems involved. Denitrification involves more than 150 known species of bacteria (see Annex A). Although it is well established that soils are a dominating source for N_2O, researchers are still struggling to fully understand the complexity of the underlying microbial production and consumption processes (Butterbach-Bahl et al., 2013).[16] The processes of N_2O formation in the oceans are no less complex. For example, recent research revealed that large amounts of N_2O are produced in regions of the Atlantic Ocean with little oxygen[17] (Grundle et al., 2017).

Applying a global approach does not necessarily imply that governments should seek a goal of reducing N_2O emissions. The climate change mitigation policy does not set an individual reduction target for each GHG. Instead, N_2O is part of a GHG basket under the UN Framework Convention on Climate Change (UNFCCC) and countries can decide how to prioritise GHG emission reductions in their own nationally determined contributions. Achieving the temperature goals of the Paris Agreement will require net zero (or lower) emissions of long-lived GHGs in the second half of the century (see for example, Rogelj et al., 2015). The implications for mitigation of N_2O are as yet unclear and, as with other GHGs, also depend on the potential for natural and artificial carbon sequestration. Meanwhile, N_2O concentrations in the atmosphere continue to increase (Figure 2.6).

Figure 2.6. Global average atmospheric concentrations of nitrous oxide (N_2O)



Source: The National Oceanic and Atmospheric Administration (NOAA) Annual Greenhouse Gas Index (AGGI), Spring 2018, https://www.esrl.noaa.gov/gmd/aggi/aggi.html, accessed 9 July 2018.

2.2 The "precautionary" approach

The increasing amount of nitrogen on Earth has enhanced the speed of nitrogen cycling in the environment, i.e. both the rate at which nitrogen has been added to but also lost from the environment has increased (Müller and Clough, 2014). The uncertainties associated with such an acceleration of the nitrogen cycle and their implications for the nitrogen cascade pathways raise the question of the precautionary principle (PP).

There is no definitive statement of the PP, but there is a reasonable consensus about what it says, at least among its proponents (in part, following Saunders, 2010). When an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically. In other words, the principle is to be applied when (a) there is scientific evidence for a threat to the environment or to health, but (b) the evidence, while sound, is not conclusive. According to the European Commission, the PP applies "where preliminary objective scientific evaluation indicates that there are reasonable grounds for concern" (EC, 2000). Under the PP, it is (somehow) deemed that there is not enough science to conduct an IPA. What is crucial, however, is that there must be a prima facie scientific case for a threat. If this is not the case, the application of the principle is not justified in any way.[18]

According to Battye et al., 2017, the rapid increase in human production of nitrogen – by almost five-fold in the last half-century – "has removed any uncertainty about the importance of human-produced nitrogen on the overall nitrogen cycle". Battye et al., 2017, questions whether denitrification can continue to keep pace with the increase in human production of nitrogen, which seems to be the case so far. For example, if the habitats of denitrifying bacteria, such as marshes and wetlands, were to be reduced, a major imbalance in the nitrogen cycle could occur. Alteration of the denitrification process could lead to a chain reaction on nitrogen pathways with unpredictable but potentially disastrous consequences on health and the environment. If not, if denitrification can meet the growing demand, then the environmental consequences are predictable and are a matter of risk management (for example, risk of increasing the greenhouse effect and depleting the ozone layer).

A second key criterion for triggering the implementation of the PP is that the proponent of an activity, rather than the public, must bear the burden of proof. It is for the scientific community to demonstrate prima facie that the acceleration of the nitrogen cycle will result in negative effects that are not already taken into account by environmental policy and that cannot be because they are not linked to one but to all environmental media. In other words, it is necessary to demonstrate that there is a systemic effect linked to the nitrogen cascade.

A third factor to consider is that the PP, and therefore the management of uncertainty, must be viewed within the overall framework of risk management (Box 2.2) According to the European Commission, the PP "forms part of a structured approach to the analysis of risk, as well as being relevant to risk management" (EC, 2000). Thus, a poorly defined PP may "direct resources toward attempts to control poorly understood, low-level risks using resources that could be more effectively directed toward the reduction of well-known, large-scale risks" (Majone, 2010).

Box 2.2. Differentiating between risk and uncertainty

A common distinction between risk and uncertainty derives from Knight's (1921) observation that risk is uncertainty that can be reliably measured. Thus, risk describes the likelihood and consequence of an uncertain event of which the probability of occurrence can be reliably estimated. Uncertainty describes situations where the probability of occurrence is not known and perhaps cannot be known. The difference between risk and uncertainty can be understood as a spectrum, where uncertainty is an expression of the degree to which a value or relationship is unknown.

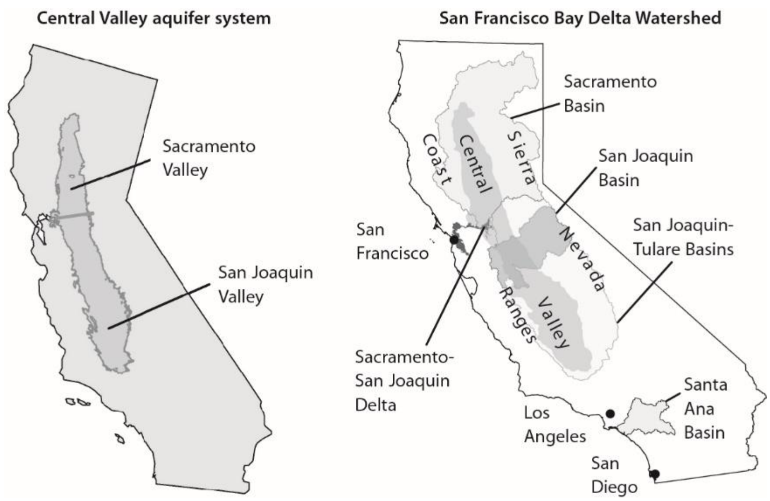
Pollutants have been defined by Holdgate, 1979, as substances causing damage to receptors in the environment (in part, following IEEP, 2014). The pollutant may be emitted from a “source” into the environment, through which it travels along a “pathway” till it reaches a receptor and creates an “impact”. It follows from this definition that if the pollutant reaches no receptor in damaging quantities – because it has been rendered harmless along the cascade either by being transformed into another substance or into a form where it cannot affect the receptor or because it has been diluted to harmless levels – then there has been no pollution. It also follows that as the mere emission of a potential pollutant to the environment does not necessarily constitute pollution, managing uncertainty does not necessarily require a reduction in all emission sources. It requires consistency with risk management. The PP should therefore be closely associated with the risk approach as part of a dual approach to managing human impacts on the nitrogen cycle. It must complement the risk approach and not replace it. It must aim to limit the total nitrogen load entering the system and, where appropriate, propose measures in addition to and in line with risk management measures.

This leads to the question of cost-effectiveness, the fourth criterion to be taken into account in applying the PP. This criterion was already embedded in Principle 15 of the Rio Earth Summit Declaration on Environment and Development, which emphasised "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (UNGA, 1992). Managing uncertainties of the nitrogen cascade for a given country means covering the entire economy. In comparison to environmental policies that target sources according to impacts, the precautionary approach offers flexibility in the choice of nitrogen sources to manage (for example, among agriculture, transport, energy, industry, wastewater). While such flexibility is synonymous with potential cost-efficiency gains, this is not the case in terms of environmental effectiveness.

The scientific community is calling for reductions in nitrogen emissions and improved efficiency of nitrogen use (see for example the concept of planetary boundaries discussed below). The USEPA's Science Advisory Board estimated that approximately 7 million tonnes of nitrogen could be reduced each year (nearly 25% of current nitrogen emissions in the United States) by disseminating technologies available in precision agriculture, fertilisers, NO_x control, wetland creation and sewage treatment (USEPA-SAB, 2011). Jörß et al., 2014 and Döhler et al., 2011 see significant reduction potentials in the German energy and agriculture sectors; SRU, 2015 estimated the overall nitrogen reduction potential in the German agricultural sector at around 40% of current emissions. Sutton et al., 2013 and Tomich et al., 2016 suggest improving the efficiency of nitrogen use along food and energy supply chains. Seitzinger and Phillips, 2017 wonder whether nitrogen use efficiency could not constitute a kind of "heuristic device", a proxy to guide policy making. However, pointing out potentials of reduction or gain in efficiency in the use of nitrogen is not sufficient. Policies must be designed to achieve these reductions or efficiencies in the use of nitrogen in a cost-effective manner, i.e. not just in terms of cost but also in relation to environmental impacts.

For example, reducing nitrogen emissions in the San Joaquin river watershed can be cost-efficient in reducing the total amount of nitrogen emitted in the state of California, given the importance of this “specific high-nitrogen region" for nitrogen emissions in California.[19] But this is certainly not the most effective way to improve the quality of groundwater in the Central Valley aquifer system and of coastal waters in the San Francisco bay (Figure 2.7) not to mention air quality in the City of Los Angeles. These risks must be managed at different geographical scales by specific risk measures and not by untargeted actions whose sole purpose is to reduce the overall nitrogen load in California. In other words, risk management and precautionary measures are not mutually exclusive, they must be complementary.

Figure 2.7. Aquatic systems at risk of nitrogen pollution in central California



Left map. Intense demand for water in the Central Valley of California and related increases in groundwater nitrate (NO_3^-) concentration threaten the sustainability of the groundwater resource (Ransom et al., 2017). The Central Valley of California, including the Sacramento Valley and the San Joaquin Valley, must be considered when assessing groundwater contamination risk in the region.
Right map: The San Francisco Bay Delta Watershed consists of several major waterways, including the Sacramento and San Joaquin Rivers and their tributaries. Where these two large rivers meet near Sacramento, a great inland Delta is formed where the river waters collect before passing into San Francisco Bay. The Sacramento and San Joaquin River Basins and the Sacramento-San Joaquin Delta should be considered when assessing the risk of nitrogen pollution of the coastal areas into which the San Francisco Bay Delta Watershed is draining, including the San Francisco Bay, San Pablo Bay, Suisun Bay, and the Golden Gate Strait.
Source: Ransom et al. (2017) (Left map) ; Kratzer et al. (2011) (Right map).

Fifth, the economic and policy implications of using the PP need to be carefully analysed. Use of the principle with respect to the nitrogen cycle can influence production decisions (for example, farmers switching to crops with low nitrogen inputs)[20] and trade (e.g. through restricting imports of foods produced with high inputs of nitrogen). Applying the PP to the nitrogen cycle can foster the development of new technologies, the dissemination of which is itself put into question by the principle, as is the case of biotechnology (genetically modified organisms). For example, it could lead to an increase in genetic engineering research to develop new nitrogen-fixing crops[21] to increase crop productivity while reducing fertiliser use.[22]

Precautionary management to cope with the uncertainties of the nitrogen cascade finally raises the thorny question of the limit to be set, the level acceptable to society, in terms of the net nitrogen balance of a country, or even of the planet. To help decision-making, some studies have estimated “boundaries” or “tipping points” for each form of nitrogen beyond which “critical limits” of their respective impacts would be exceeded (Box 2.3). As these studies acknowledge, such estimates should not lead to setting global emission reduction limits, given the spatial (and temporal) variability of nitrogen impacts on air, water, biodiversity and soils and, for N_2O, the possibility of mitigating global warming by acting on other GHGs. The boundary concept is even less conducive to setting a single threshold for nitrogen in all its forms. Indeed, there is no such thing as a single value of the damages caused by nitrogen – or a fixed exchange rate between the emissions of different forms of nitrogen – since the impacts of nitrogen are multiple and are specific to the site and form of nitrogen.[23]

Box 2.3. Estimating boundaries for different nitrogen forms

The concept of planetary boundaries

The concept of planet-wide environmental boundaries, or "tipping points", has recently been introduced in an attempt to respond to the multiple pressures on the Earth's system caused by the acceleration of human activities (Rockström et al., 2009).[24] The Planetary Boundaries (PB) framework identifies acceptable levels of anthropogenic perturbations below which the risk of destabilisation of the earth system is likely to remain low – a "safe operating space" for global societal development. Beyond the boundaries, the earth system is at risk, and with it humanity. In this way, PBs provide a science-based analysis of the risk to destabilise the earth system at the planetary scale.

Rockström et al., 2009 proposed nine PBs, namely climate change, biodiversity loss, increased nitrogen and phosphorus cycling, stratospheric ozone depletion, ocean acidification, global freshwater use, change in land use, chemical pollution and atmospheric aerosol loading. Three PBs were identified as having already exceeded their tipping points – climate change, biodiversity loss and human interference with the nitrogen cycle. On the latter, the safe operating space for anthropogenic nitrogen fixation was roughly estimated at 35 million tonnes N per year, or 17% of its current value as estimated by Fowler et al., 2013 (see Table 1.1 in Chapter 1). Rockström et al., 2009 admitted the roughness of this estimate, implying the need for an update.

First criticism

The concept of PB appeared to be controversial as it was deemed too low to feed the current world population (Nordhaus et al., 2012). Since PBs aim to ensure a "safe operating space" for human development, the human need for nitrogen should be considered as well the environmental impacts.

In response to this first criticism, de Vries et al., 2013 estimated global food production needs at ~ 52-80 million tonnes N per year and losses to the environment deemed 'acceptable' (i.e. below critical thresholds for health and the environment) at 20-133 million tonnes N per year[25] (Table 2.2). Although it is two to six times higher than the previous estimate (72 to 213 million tonnes N per year compared to 35 million N tonnes per year), this new estimate of PB remains below or close to the current rate of anthropogenic fixation estimated by Fowler et al., 2013 at 210 million tonnes N annually. In other words, according to de Vries et al., 2013 even if an optimal allocation of nitrogen (and phosphorus) could be achieved across the planet, it is likely that the PB for nitrogen is lower than the current fixation.

Table 2.2. Planetary boundaries for anthropogenic nitrogen fixation

Million tonnes of N per year

Indicator	Critical limits	Current nitrogen losses	Planetary boundaries
NH_3 concentration in air[1]	1 μg per m^3	24.9	89
	3 μg per m^3	32.1	115
Radiative forcing of N_2O[2]	1 W per m^2	0.8	20
	2.6 W per m^2	5.3	133
NO_3^- concentration in drinking water[3]	25 mg per litre	30.0	83
	50 mg per litre	36.9	111
Nitrogen in surface water[4]	1.0 mg N per litre	5.4	62
	2.5 mg N per litre	7.2	82
Phosphorus in water[5]	6.2 million tonnes P per year	-	73
	11.2 million tonnes P per year	-	132
Global needs of nitrogen for food production[6]	Current NUE	-	80
	25% increase in NUE	-	52

1. Upper and lower limits for impacts on lichens and higher plants, respectively
2. Using the present share of N_2O to the sum of CO_2, methane (CH_4) and N_2O and excluding the effects of nitrogen on CO_2 sequestration
3. World Health Organisation (WHO) drinking water limits
4. Critical limits in terms of surface water eutrophication (dissolved inorganic nitrogen)
5. Critical limits in terms of phosphorus fertiliser input (Carpenter and Bennett, 2011)
6. For 9 billion people consuming food at the recommended level, thus avoiding both overconsumption and malnutrition, assuming either current Nitrogen Use Efficiency (NUE) in agriculture or a 25% increase in NUE

Source: After De Vries et al. (2013); Steffen et al. (2015).

Second criticism

The second, more fundamental, criticism of the PB concept is the irrelevance of a global threshold due to spatial variability (Lewis, 2012; Nordhaus et al., 2012). First, many impacts occur at a regional level (e.g. terrestrial biodiversity decline due to nitrogen deposition; eutrophication of fresh and marine waters due to nitrogen runoff). Second, the supply of nitrogen (and phosphorus) fertilisers is very unevenly distributed between OECD countries and the rest of the world (Vitousek et al., 2009).

Some scientists have argued that this spatial variability of nitrogen impacts and nitrogen availability would be taken into account by establishing regional boundaries (RBs), citing as an example of RB the critical loads of nitrogen deposition. According to Vries et al., 2001 and Erisman et al., 2001, the setting of RBs does not preclude assessing boundaries on a larger scale, as was done in the Netherlands; exceeding these "upper boundaries" would signal potential regional problems unrelated to the spatial allocation of nitrogen.

Overall assessment

A PB for N_2O has little policy relevance; instead, any radiative forcing boundary must refer to all GHGs as the reduction of N_2O emissions can be exchanged for CO_2 or CH_4 reductions. RBs are already part of nitrogen risk management (such as the risks of acidification and eutrophication in the example of critical loads of deposition on terrestrial ecosystems). Pending more work on the relevance of establishing PBs or RBs, as in the framework of the International Nitrogen Management System (INMS), a national nitrogen budget could be a starting point for any precautionary approach (see below).

It is too early to discuss any limit or level of efficiency improvement desirable in the use of nitrogen or its losses. The ongoing work of the INMS is expected to shed light on this issue by 2021 (Box 2.4). In the meantime, as a first step, countries could establish an economy-wide nitrogen balance and begin to monitor trends. This would involve assessing the total amount of nitrogen introduced into the environment from all sources and monitoring these sources in order to report – both by source and overall – the amount of nitrogen released each year, also taking into account denitrification. This could be undertaken in parallel with risk-based efforts to manage specific nitrogen impacts.

Box 2.4. The International Nitrogen Management System

The Global Environment Fund (GEF) has pledged close to USD 6 million to set up the INMS (via the so-called "Toward INMS" process). The four-year timeframe for Toward INMS is 2017-21. Coverage is global. UNEP is the GEF 'Implementing Agency' (i.e. policy customer), while the UK Natural Environment Research Council, Centre for Ecology and Hydrology (CEH) is the GEF 'Executing Agency' (i.e. project coordinator). UNEP contribution to Toward INMS is through providing the Secretariat function of the Global Partnership on Nutrient Management (GPNM), which was established in 2007 to steer dialogue and actions to promote effective nutrient management.

INMS will provide a combination of analyses to support nitrogen policy making. The Toward INMS architecture includes four components: (i) tools for understanding and managing the global nitrogen cycle; (ii) quantification of nitrogen flows, threats, benefits; (iii) regional demonstrations; and (iv) awareness raising and knowledge sharing.

Seven regional demonstrations have been set up to show the benefits of joining up nitrogen management at the regional/catchment scale:

- Western Europe (Atlantic coast, including France, Spain and Portugal)
- East Asia (including China and Japan)
- South Asia (including India, Maldives and Sri Lanka)
- Eastern Europe (Dnieper catchment, including Russia and Ukraine)
- Latin America (La Plata catchment, including Brasil)
- East Africa (Lake Victoria catchment, including Uganda)

Monitoring of nitrogen mass flows (e.g. via an economy-wide national nitrogen balance) would provide an early-warning indicator of whether policies are easing the overall situation or are only shifting the problem from one location to another

(Salomon et al., 2016). In other words, if the nitrogen released into the environment has not been significantly reduced by the policies in place, this may indicate that not all impacts have been satisfactorily managed (i.e. that nitrogen policies have not all been effective) or that new risks are incurred. Such “headline indicator” would also raise policy makers’ awareness to the systemic dimension of the nitrogen issue and would improve policy coherence (ibid).

Some countries have undertaken such comprehensive monitoring of their nitrogen mass flows, such as Denmark, Germany, The Netherlands, Switzerland and the United States. The European Nitrogen Assessment (Sutton et al., 2011) provides estimates for the EU-27 and the California Nitrogen Assessment estimates a state-level economy-wide nitrogen balance (Tomich et al., 2016). In 2012, the OECD Working Party on Environmental Information (WPEI) initiated discussions on the methodology for measuring nitrogen flows with a view to developing a country-wide nitrogen indicator. The WPEI proposed a tiered measurement framework and a reporting template that could form the basis for defining nitrogen indicators for inclusion in the OECD sets of green growth and environmental indicators (OECD, 2013 and 2014).

Notes

[1] What Keeler et al., 2016 call the "end points of interest".

[2] This report does not detail the acceptable levels of risks set by environmental policy.

[3] In December 2015, the twenty-first session of the Conference of the Parties (COP 21) to the UN Framework Convention on Climate Change (UNFCCC), held in Paris, set a long-term goal of keeping the increase in global average temperature to well below 2°C above pre-industrial levels.

[4] As estimated by using carbon (C) stock information in the national inventory of GHG emissions and sinks. Nitrogen stock change was determined by simply assigning a molar C/N ratio of 12 for soils and 261 for trees and making the appropriate conversions from C to N.

[5] The literature review of Battye et al., 2017 leads to the same conclusion: changes in terrestrial biomass and marine sediments, which are the main nitrogen sink after denitrification, would capture only 9 and 13 Tg N per year respectively, compared to 400 Tg N per year for denitrification.

[6] Biogeochemical pathways are referred to in this report as “pathways”.

[7] As defined by the USEPA, an airshed defines the geographic area that contains the emissions sources that contribute 75% of the nitrogen form (e.g. NO_x) deposited in a particular watershed, city or protected ecosystem.

[8] Air pollution is considered the biggest environmental challenge in Chile, particularly fine particles ($PM_{2.5}$) pollution in the Metropolitan Region of Santiago, two thirds of which are secondary particles with NO_x as one of the precursors.

[9] Other benefits are not included in the calculation, and this is the main reason why area-related measures generally have the lowest cost efficiency.

[10] The net input term of NANI includes deposition of NO_y (the sum of all oxidised nitrogen forms), nitrogen fertiliser use, nitrogen fixation by crops and net nitrogen in import and export of food and feed. The output term is riverine export (Hong et al., 2013).

[11] The share remitted back to the atmosphere is significantly higher than the share absorbed by terrestrial plants, according to USEPA-SAB, 2011.

[12] GWAVA consists of two nonlinear regression models: one for shallow groundwater (typically < 5 m deep, and which may or may not be used for drinking) and the other for deeper supplies used for drinking.

[13] No matter where N_2O is emitted, it will contribute to climate change and ozone layer depletion effects globally (see Chapter 1).

[14] The modelling tool, named Overseer, was originally designed for managing nitrogen fertiliser; it can also be used to estimate N_2O emissions by combining GHG emission factors used in the national inventory with farm data.

[15] A better understanding of the pathways would also be required for estimating the share of N_2O emissions that contribute to ozone layer depletion.

[16] For example, current biochemical models hold that inorganic hydroxylamine is the only intermediary formed when nitrifying bacteria convert ammonium (NH_4^+) into dormant nitrite (NO_2^-). In a new study, chemists found that hydroxylamine is converted into another intermediary – nitric oxide (NO) - which under normal soil conditions acts as the

chemical prelude to NO_2^-; but under imperfect soil conditions, NO is converted into N_2O (Caranto and Lancaster, 2017).

[17] Extreme low oxygen concentrations occur in the Atlantic in ocean eddies of up to 100 kilometres in diameter.

[18] So far, there has been a narrow and selective application of the PP. There are only two cases where it has been invoked: climate change and biotechnology products, including the use of genetically modified organisms (GMOs) to reduce the use of herbicides (Saunders, 2010).

[19] The California Nitrogen Assessment has chosen to study the San Joaquin river watershed "because of the mix of intensive agricultural production combined with large urban areas" and a "history of air and water pollution associated with excess nitrogen" - a so-called "specific high-nitrogen region" (Tomich et al., 2016). Groundwater pollution is described as the main nitrogen pollution problem.

[20] For example, grasses such as chicory, plantain, red clover or white clover may be substituted for perennial ryegrass as forage in temperate regions, with perennial ryegrass having a very high need for nitrogen to support its growth (Gilliland et al., 2010).

[21] Either through a transgenic strategy (i.e. by transferring the nitrogenase gene from the bacteria to the plant) or by extending the symbiotic fixation of nitrogen to non-legumes (i.e. by letting the bacteria fix the nitrogen, but by making it happen inside the plant instead of the soil).

[22] Nitrogen fixed by legumes responds more to the needs of the host plant than spreading chemical fertilisers, no matter how precise.

[23] Reasoning by analogy with the possibility of setting a carbon price based on the global warming potential of different GHGs, Socolow, 1999 and 2016 proposes to use a single price of nitrogen to promote an efficient use of nitrogen.

[24] The acceleration of human activities has been so dramatic that a new geological epoch, the Anthropocene, has been proposed (Crutzen, 2002; Crutzen and Steffen, 2003).

[25] Steffen et al., 2015 estimated a PB for phosphorus (P) based on the need of P in plant growth, using an average N:P ratio in growing plant tissue of 11.8.

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