Defra project AQ0834 - Identification of Potential "Remedies" for Air Pollution (nitrogen) Impacts on Designated Sites (R.A.P.I.D.S.)

Appendix 9. Timescales of intervention and recovery, and evidence of success

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Summary

- Impacts of reactive nitrogen (N) pollution can be detected in chemical and biological aspects of soil, vegetation and the freshwater environment, and are usually unfavourable for nature conservation targets.
- Interventions to reduce or mitigate the effects of N emissions can be achieved over timescales of 20-30 years for agricultural infrastructure, 10-20 years to establish tree belts that can increase interception of N, or as little as two years to achieve local reductions in road transport emissions. More rapid reductions in agricultural N emissions could be achieved if there is willingness to adopt alternative approaches for manure or fertilizer use.
- Recovery from impacts occurs across different timescales, and is primarily dependent on four controlling variables: the accumulated N in the plant-soil system from previous N deposition, the amount by which N deposition declines, the level it declines to and the composition of the ecological community. Timescales of recovery are affected by these variables in different ways.
- Recovery in 1-4 years is possible for some sensitive bryophytes and lichens, declines in nitrate leaching, and declines in plant tissue N content. Early indications of recovery in 5-20 years may include some improvements in species composition, plant growth, soil processes such as reductions in available N, and chemical and biological recovery of freshwaters from acidification. However, because N can persist within ecosystems for centuries, effects of excess N may still be visible after 50 years, and full recovery may take many decades or may not be achievable.
- Ongoing damage from previous N pollution will prevent the fulfillment of the EU target of "Halting the loss of biodiversity … by 2020". Nevertheless, substantial decreases in N pollution would

considerably reduce both current and future damage to biodiversity, even if deposition remains above the critical load.

- Detecting N impacts and recovery on a particular site is most effectively done by monitoring the flora using permanently-sited quadrats, for changes in: the occurrence of sensitive species (particularly mosses and lichens), the ratio of the abundance of grass species to forb species, the species richness, and the mean Ellenberg N score. Measurement of other plant, soil and water variables is also useful, but requires additional technical expertise.

9.1 Timescales of intervention and recovery

9.1.1 Timescales for implementation of measures

Many of the agricultural mitigation measures, and particularly those relevant to manure application, can be implemented immediately and therefore give immediate emission reductions. However, for some of the livestock housing and manure storage measures, retro-fitting of new technologies to existing facilities is often prohibitively expensive and these measures are therefore only generally applicable to new build situations (a potential exception would be review of IED permit adjacent to an SAC where there is UK precedent for requiring retro-fitting). The anticipated livestock housing lifetime will depend on design and build parameters, and may vary from 10 to 50 years. Assuming a conservative estimate of mean lifetime of 30 years would give an annual replacement rate (and therefore potential mitigation measure uptake rate) of 3.3%. For manure storage a 20 year lifetime, or 5% replacement rate, may be more applicable. However, the actual replacement rates will depend on the economic health of the sector and be strongly influenced by market and legislative drivers. For example, where animal welfare legislation requires a change in animal housing methods, this can provide the opportunity to require that replacement approaches also decrease, but at least not increase, ammonia emissions.

In theory, large pig and poultry farms that have to comply with the IED, (accounting for >90% of broilers, >60% of layers and c. 30-40% of pigs) will already have implemented measures to reduce emissions from housing and on-site manure storage. In practice, there is a lack of robust evidence on current uptake of specific mitigation measures on pig and poultry farms. The assumption made in current NH_3 emission projections is that in 2006 there was 0% compliance and that by 2020 there will be 100% compliance, with implementation in housing of best available techniques (BAT) giving, on average, a 30% reduction in emission over baseline. It should be noted that there is a range of techniques recognised as BAT, with a range of performance. It is therefore also possible to specify techniques that exceed the basic BAT requirements (sometimes termed 'BAT+' or 'BAT++').

Considering that N emissions from road transport are predominantly governed by advances in technology, reductions in emissions are likely to correspond to the extent and speed in which future technologies are adopted. However, the London Congestion Charge has shown that changes in emissions can be made more rapidly. The Congestion Charge has reduced vehicle numbers in the capital and significant improvements to air quality were observed within 2 years (reductions in NO_x and PM_{10} of 12 % and CO_2 by ~20% after 2 years; Beevers and Carslaw, 2005).

Use of landscape structures to reduce N exposure to designated sites will have variable implementation times. For example, buffer zones requiring low emission application of manure and urea-based fertilisers could provide immediate benefits. By contrast, strategies based on tree belts (either adjacent to road or farm sources or adjacent to nature areas) take 10-20 years to become effective, depending on the type of trees used (Bealey et al., 2013).

9.1.2 Nitrogen impacts on habitats

"Receptors" is a general term for different aspects of habitats that are affected by N. Below we list the general receptors that are damaged by N, and that might show recovery at different timescales in response to declines in N deposition. These are broad ecosystem components or processes for terrestrial and freshwater systems, and for each it is possible to select specific indicators that can be monitored to reflect change in status due to increasing or decreasing N deposition, as discussed in section 9.2 below.

Terrestrial habitats

Plant receptors

Lower plants, in the context of N deposition assessment, are mosses and liverworts (together termed bryophytes) and lichens. The key feature in this context is that such plants have little or no rooting capability, making them highly dependent on the atmosphere for nutrient inputs[1]. Bryophyte and lichen species are not universally sensitive to N, with species ranging from highly sensitive low-N adapted species through to N-loving (nitrophile) species. Lower plants can respond rapidly to changes in N since they take up the majority of their N directly through their leaves and exposed surfaces, rather than from the soil. This means that although there may be some short-term storage of N (for a few years) within plant tissue such as moss mats, most species are in limited contact with the accumulated pools of soil N, and so can recover more rapidly than other receptors (e.g. Jones 2005; Mitchell et al., 2004). Epiphytic species not affected by plant canopies are likely to recover the fastest, while recovery of epiphytes below tree canopies and recovery of terricolous species may be slower due to continued cycling of N via the canopy and due to light competition from higher plants respectively. Lower plant characteristics responsive to N include **tissue N content, tissue N:P ratio, growth rates and biomass, and species composition and abundance**.

Higher plants include grasses, forbs[2], shrubs and trees. As with lower plants these exhibit a range of N tolerance, but most species are more tolerant of N than the lower plants. The most sensitive species are those associated with habitats characteristic of extreme nutrient limitation (oligotrophic), such as moorlands and peatlands which are key habitats for the UK. Changes to assemblages of higher plant species are typically caused by competition from faster growing nitrogen-loving species, or may be mediated by secondary factors interacting with eutrophication such as climate stress (drought, severe frost), disease and herbivory (Jones et al. 2014; Dise et al. 2013). Characteristics responsive to N include **tissue N content, tissue N:P ratio, growth rates and biomass, and species composition and abundance**.

Soil receptors.

Soil processes. A wide range of processes in soil may be affected by N deposition. **Mineralisation** reflects the rate of N turnover (i.e. the rate of processing or breakdown of soil organic matter). Generally, mineralisation rates are higher under elevated N deposition. **Nitrate leaching** occurs when available N in the soil exceeds the biological demand from plants and microbes, and is subsequently leached out through the soil profile. Nitrate leaching occurs once soils start to become saturated with N. In soils approaching N saturation, leaching typically occurs in winter when biological demand for available N is low but mineralisation is still producing N. In completely N-saturated soils leaching may occur all year round.

Soil C and N pools. Although the overall soil N pool is likely to increase with N deposition, this increase can be hard to detect. This is because most soils have a large existing stock of (mainly inert) N bound up as organic material which is not available for uptake by plants until it is slowly mineralised, so it is difficult to measure the relatively minor concentration changes due to the addition of more reactive and ecologically significant N. Changes in the total amount of N in soil are also hard to detect, because it is difficult to sample a comparable soil layer each time. Changes in soil C:N ratio are in principle easier to detect, but it should be noted that N pollution can lead to no change or an increase in soil C/N ratio by stimulating the production of fresh litter (Dise et al. 1998; Jones et al. 2004). **Soil pH and leachate pH** reflect acidification caused by N processing and by N leaching. A decrease in soil pH indicates a

[1] ferns are strictly also lower plants, but have rooting capability, so in this context are grouped with higher plants
[2] a herbaceous plant that is not a graminoid, i.e. not a grass, sedge or rush.

reduction of the buffering capacity of the soil as a result of acidification caused by N. The likely decrease depends on how base-rich the soil is. For example, calcareous soils on limestone or chalk are highly buffered and are unlikely to acidify much due to N deposition. By contrast acidic or neutral soils may acidify rapidly and take a long time to recover.

Animal and bird receptors

In principle, wildlife can be adversely impacted by atmospheric nitrogen deposition. In most cases, there is little known direct sensitivity to high N deposition or atmospheric concentrations, although as with humans wildlife may be expected to be adversely affected by the airborne particles and ozone concentrations associated with nitrogen air pollution. With regard to N deposition itself, the main mechanism of change is expected to be the alteration of food quality / availability or nesting conditions. Where the nesting conditions or food source of a bird or animal with very specific requirements are altered by nitrogen deposition, then knock-on adverse effects may be expected in the bird or animal population. For example, red-backed shrike populations in the coastal dunes of continental Europe have declined due to N deposition, mediated via changes in the size and species of prey items as a result of N-induced increases in grass cover (Dise et al. 2011). Evidence for widespread changes in practice, however, is only circumstantial at present, indicating the need for further research (Dise et al., 2011). By contrast, animals that are generalists in terms of their feeding and breeding habitat requirements are expected to have little sensitivity to nitrogen deposition.

Freshwater habitats

Chemical receptors

Chemical parameters include key factors in freshwater systems that are directly affected by N deposition. **Surface water nitrate concentrations** in upland aquatic ecosystems in the UK can reflect the amount of atmospheric N deposition that is received within the catchment. Nitrate is an important nutrient, and in some aquatic ecosystems, inputs of nitrate via atmospheric N deposition can stimulate the growth of some algal and aquatic plant species at the expense of others, thereby altering the ecology of the aquatic system. Surface water acidity increases substantially as a result of atmospheric N deposition in UK upland waters that have a low acid neutralising capacity or alkalinity. Therefore, reductions in the **pH** or **Acid Neutralizing Capacity** (ANC, the balance between base cations and acid anions) of surface waters can reflect acidification impacts of atmospheric N deposition. In contrast to freshwater habitats in upland areas, where atmospheric N deposition effects can be detected, nitrate levels in lowland freshwater ecosystems are more typically dominated by nitrogen inputs from agriculture and sewage related sources.

Biological receptors

Biological parameters in freshwater habitats include aquatic organisms that are sensitive to changes in chemical parameters resulting from atmospheric N deposition. These include **micro-algae** and **macrophytes** (plants that live in aquatic ecosystems). Changes in the absolute or relative abundance of some sensitive species of micro-algae and macrophytes can be indicative of the degree of acidification in the aquatic ecosystem. Other species of micro-algae and macrophytes are N-loving and increase in abundance with higher inputs of atmospheric N to the aquatic system. Aquatic **macroinvertebrates** include insects, crustaceans, snails, worms and their relatives that live in aquatic ecosystems. The abundance of some **macroinvertebrate indicator species** is linked to the degree of acidification of some ecosystems due to their clear pH tolerance thresholds.

9.1.3 Nitrogen impact timescales – concepts

Effects of N pollution occur over different timescales, as summarised in the previous section. The effects of progressive N pollution have been characterised using the concept of saturation stage (Aber et al., 1998; Emmett, 2007) (e.g. Figure 9.1). Although such conceptual diagrams are useful for illustrating that different effects are observed as N pollution becomes progressively worse, the concept of "saturation stage" is not well defined. The effects of a short period with large amounts of N deposition are not distinguished from those of a longer period with lower deposition, and the diagrams could give the impression that reducing deposition results in an immediate movement towards a less saturated stage. In fact, different receptors (aspects of the habitat that are affected by N) will recover at different rates.

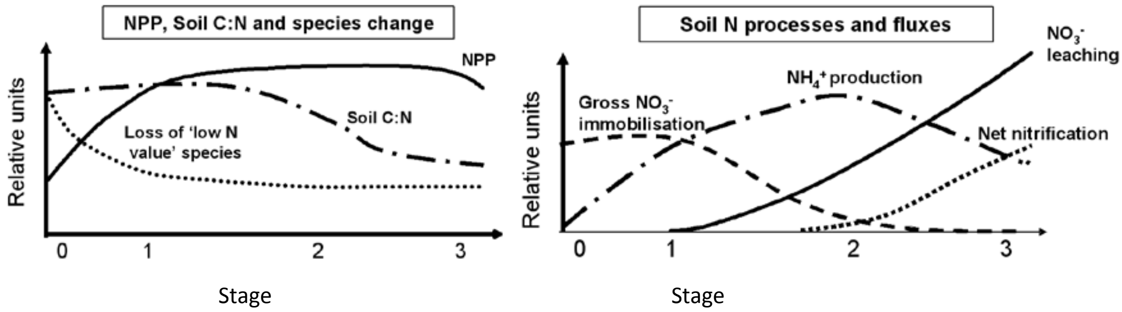

Figure 9.1. Timing, in relation to continued N deposition and progressive N saturation of an ecosystem, of changes in soil C/N ratio, net primary productivity (NPP), occurrence of plant species associated with low N availability, gross microbial nitrate immobilisation, ammonium production and nitrate leaching. From Emmett (2007).

Sensitive epiphytes such as bryophytes and lichens may be lost rapidly due to ammonia toxicity, but can recover rapidly following a decrease in N deposition rate (Mitchell et al., 2004). These rapid responses may relate to the limited N storage capacity in epiphytes and the thin substrates they grow on. In ecosystems with larger pools of N in vegetation and soil organic matter, pollutant N tends to persist and accumulate. Of the extra N that enters through atmospheric deposition, a comparatively small proportion is likely to be lost through processes such as leaching or denitrification, and the remainder will build up in vegetation and in soil organic matter. Much of the damage to habitats results from soluble N, whether this is leached, causing acidification, or taken up by plants, causing an increase in productivity and the loss of less-competitive species. From this perspective, the storage of N in organic matter could be seen as comparatively undamaging. However, soil organic matter is likely to decompose in the medium- to long-term, releasing a steady trickle of soluble N and resulting in persistent and ongoing eutrophication.

A dynamic model of N transfers within ecosystems can be used to illustrate the importance of these temporal processes (Figure 9.2), reproduced from Rowe et al. (2013)). Two N deposition scenarios are shown, one in which deposition onto a wet heath is maintained at the current level of around 20 kg N ha^{-1} yr^{-1}, and one in which deposition is decreased to the upper end of the Critical Load range i.e. 10 kg N ha^{-1} yr^{-1}. This represents a change in **pressure**. The effect of this change has to be placed in the context of a long history of elevated N deposition at this site, which has likely caused an accumulation of over 100 kg N ha^{-1} within slowly-turning-over organic matter (N will also accumulate in organic matter pools with faster and even-slower turnover rates; these pools are not shown). If deposition continues at the current rate, this accumulation will continue, as shown by the still-increasing solid line in the central graph. However, even with an abrupt decrease in deposition, it is likely to take very many decades for even half of the accumulated N to be released. This store of N can be seen as representing a **risk** to the ecosystem, causing a chronic increase in N availability. The immediate cause of **damage**, however, is mainly the exposure of plants and other organisms to available, soluble N. With an abrupt decrease in N deposition, there is an immediate decline in plant-available N, since much of this comes from current deposition. However, the effects of the N that has accumulated within the soil are shown by the continued release of N into soluble form, for many decades following a decrease to the Critical Load.

These dynamic model outputs provide a good illustration of the way chemical delays can affect the impacts of changes to deposition rate. Nitrogen inputs in excess of the ecosystem's processing capacity (which is broadly equivalent to the Critical Load) are likely to accumulate. The greater the deposition rate, and the longer the period for which excess N is deposited, the larger an amount of N is likely to accumulate in soil organic matter, creating the risk of long-term damage. When N deposition is decreased, this is likely to cause an immediate decline in N availability, and also allow slow turn-over pools of stored N to begin to decrease. However, the most persistent passive pool, which represents the majority of the stored N in the soil, will actually continue to increase over timescales of 100s of years (Figure 9.3) due to continued transfers of N from the faster pools. Models suggest that this may start to decline ever so slightly after a few centuries, but effectively this pool of passive N remains in the system and will not decrease within a relevant timeframe for management purposes.

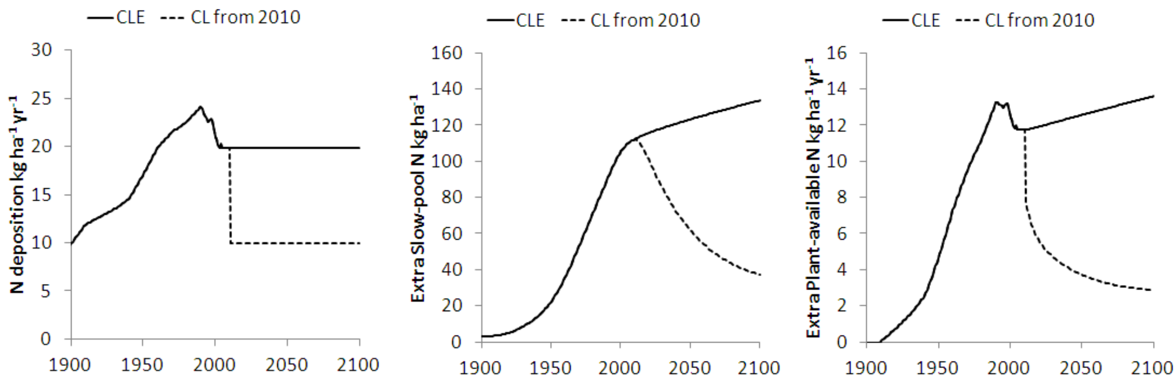

Figure 9.2. Comparison of effects of two N deposition rate scenarios ('pressure') on 'risk' (slow-pool soil N*) and 'damage' (plant-available N*), in a wet heath site (Migneint) as simulated by the MADOC dynamic model. Solid line = Current Legislated Emissions (CLE) scenario for N deposition. Dashed line = decrease to upper end of critical load range for blanket bogs (10 kg N ha^{-1} yr^{-1}) from 2010. Reproduced from Rowe *et al.* (2013). * = Extra in comparison to pre-industrial conditions.

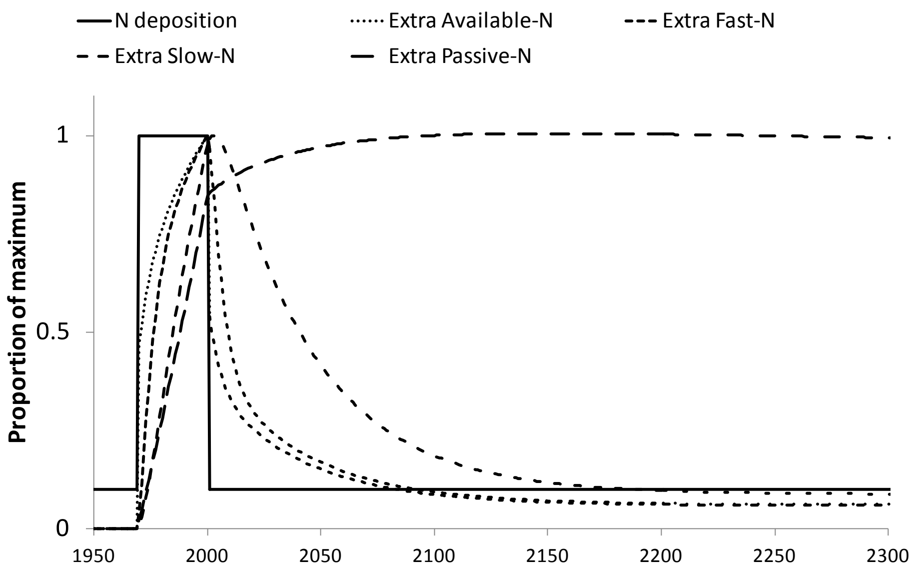

Figure 9.3. Increases in soil N pools (extra over constant low deposition scenario) with different turnover rates to a hypothetical increase in N deposition from 2 kg ha^{-1} yr^{-1} to 20 kg ha^{-1} yr^{-1} for the period 1970-2000, as predicted by the MADOC model for a peatland system.

The persistence of N in ecosystems might be seen as a reason for not taking mitigation action, since recovery will be partial and many aspects of the ecosystem will not recover for a lengthy period. However, even marginal decreases will be beneficial in terms of reducing both N availability and reducing the further accumulation of N in soil organic matter. Together with evidence that biological responses are progressive, with different species being put at risk at different stages of N saturation (see section 9.2.3), this understanding of chemical delays makes it clear that any decreases in deposition are likely to decrease the risk to habitats in both the short and long term. Reducing pollution is most effective where it has not been elevated by much or for long, benefitting sites and receptors on the edge of the critical load (Payne et al. 2013).

9.1.4 Evidence of recovery from N pollution

Terrestrial systems

Different ecosystem components (receptors, see section 9.1.2) differ in sensitivity to N impacts, and their rate of recovery may not be the same as the rate at which they are impacted. Compared with the reasonably extensive literature on N impacts, there have been relatively few studies looking at evidence for recovery. This is partly due to the difficulty of experimentally reducing levels of N deposition in areas with high background levels. The literature comprises three main sets of evidence: revisiting or monitoring experiments where high levels of N had been applied and then ceased (e.g. Mountford et al. 1996; Strengbom et al. 2001; Power et al. 2006); transplant studies usually involving reciprocal transplants between areas of high and low N deposition (e.g. Mitchell et al. 2004); and experimental manipulations of N to below ambient levels using some form of controlled environment such as recovery roofs (e.g. Boxman et al. 1998) or mesocosms within controlled conditions (e.g. Jones 2005). We summarise the evidence for rates of recovery, drawing on recent syntheses in Rowe et al. (2012), and ROTAP (2012), as well as primary sources.

The faster response of lower plants outlined above applies both to increases in N deposition, and in response to declines in N deposition. There is evidence from recovery experiments and transplant experiments that species abundance and species composition, tissue chemistry, moss mat thickness and growth rates of N-sensitive mosses can show at least **partial recovery** within timescales as short as 1-4 years for *Racomitrium lanuginosum* in alpine heath (Armitage et al. 2011) and *Racomitrium lanuginosum*, *Rhytidiadelphus loreus, Hypnum jutlandicum* and the lichen *Cladonia furcata* in acid grassland (Jones 2005). Similarly, in a transplant experiment of epiphytic bryophytes, tissue N and growth recovered in *Frullania tamarisci* within a year, with similar but non-significant trends in two other species (Mitchell et al. 2004). However, in other experiments, lower plants have taken longer to recover, or have not shown any recovery. In calcareous grassland mesocosms the bryophytes *Rhytidiadelphus squarrosus* and *Pseudoscleropodium purum* showed no response to declines in N deposition after 7 years (Jones 2005), while in heathland recovery experiments, there was no change in bryophyte and lichen abundance after 7 years following a reduction in N deposition for an upland heath (Edmonson et al. 2013), while no consistent change in lichen abundance was observed after 8 years reduction in deposition for a lowland heath (Power et al. 2006). In a boreal forest experiment N effects on bryophyte abundance were still observable after 47 years of recovery (Strengbom et al. 2001). One possible reason for the delayed response in canopy-dominated systems such as heathland and forest is that cycling of the stored N pool continues via litter fall, thus acting as an additional indirect source of N to lower plants otherwise largely isolated from the soil.

Higher plants generally show a wide range in response times under recovery from previous high nitrogen deposition loads, ranging from relatively short timescales for physiological measures in some studies to little or no recovery of species composition in the majority of experiments so far. Changes in plant productivity or tissue chemistry have been noted over relatively short recovery periods. For example in an upland heathland, shoot extension as a measure of plant productivity of *Calluna vulgaris* (heather) decreased after only two years of recovery (Edmondson et al. 2013), while in a lowland heath, effects on canopy height recovered after eight years (Power et al. 2006). However, effects on bud burst and flowering and drought injury on *Calluna vulgaris* in the lowland heath were still observable at eight years of recovery. In a hay meadow only small decreases in biomass were observed 15 years after reducing N inputs (Stevens et al. 2012). Nitrogen effects on higher plant community composition appear to be very persistent. One experiment in a hay meadow has shown relatively rapid recovery in species richness, although differences were still apparent after 18 years (Královec et al. 2009). However, most experiments show no clear recovery in higher plant species richness or composition. Jones (2005) showed no change in higher plant composition of acid grassland or calcareous grassland mesocosms after seven years of recovery. Stevens et al. (2012) reported only small decreases in mean Ellenberg N score (a measure of fertility or productivity) after 15 years of reducing N deposition in a hay meadow. Strengbom et al. (2001) showed persistent differences in ground flora composition of boreal forest after 47 years of recovery from experimental N additions.

Soils also show a wide range in response times. Although nitrate leaching generally increases only after considerable accumulation of N in ecosystems, a decrease in nitrate leaching is often observed quickly after a decrease in N deposition. Once the level of N deposition drops below the point at which the biological demand and chemical binding capacity of the soil are exceeded, nitrate leaching has been shown to decrease substantially within a few years in pine forest and heathland (Boxman et al. 1998; Williams et al. 2004). There can also be recovery of other soil processes. In an upland heath, soil available N and total soil N concentrations declined, and soil C:N ratios increased after seven years (Edmondson et al. 2013). However, there was no change in soil N pools or soil microbial biomass in lowland heath after eight years of recovery. In a hay meadow experiment there is evidence of reductions in the soil N pool after 15 years (Stevens et al. 2012).

Freshwater systems

Recovery of water receptors varies between chemical and biological parameters, with the majority of studies focusing on recovery from acidification. The chemical parameters, surface nitrate concentration and acid neutralising capacity, typically have more rapid recovery times relative to the biological parameters such as algal and macroinvertebrate abundance and species composition (Battarbee et al. 2014; Murphy et al. 2014). Chemical recovery of freshwaters from acidification may begin approximately 5 – 15 years following decreases in acid deposition, while biological recovery from acidification occurs between 10 – 20 years following decreases in acid deposition. Lags in biological recovery from acidification may be partly due to insufficient recovery of chemical parameters as well as a degree of ecological resistance to the re-establishment of acid-sensitive species in the aquatic ecosystem. For example, 17 UK upland freshwater sites monitored for 20 years by the UK Upland Waters Monitoring Network (UWMN) have showed chemical recovery, while only 10 of these sites have shown biological recovery in terms of their macroinvertebrate populations (Murphy et al. 2014). Freshwaters should also respond to changes in diffuse N pollution loading from agricultural and other sources.

The recovery of aquatic ecosystems from the nutrient-enrichment effects of N deposition needs to be examined through long-term monitoring programmes, as current evidence is limited. However, based on work on nutrient reductions in other types of freshwater ecosystems, chemical and biological recovery might be expected to begin roughly 5 – 20 years following reductions in N concentrations within the freshwaters.

Summary

Sources of information on the timescales and evidence of recovery in terrestrial systems are summarised in Table 9.1 below, reproduced from Rowe et al. (2013), supplemented by additional information. However, other information and data are available in the grey literature which, if collated and analysed, might improve our understanding of the timescales of recovery of different processes, and the consistency of responses across habitats. This information is summarised graphically, together with recovery timescales in aquatic systems in Figure 9.1, to illustrate the response of each receptor group discussed above.

Table 9.1. Summary of observations in experiments where nitrogen deposition rate was decreased. Adapted from Rowe et al. (2013).

Reference	Habitat type	Years since decrease	Recovery observed
(Armitage et al., 2011)	alpine moss-sedge heath	2	Partial decrease in moss tissue N
(Edmondson et al., 2013)	Heathland	2	Decreased *Calluna* shoot extension
"	"	7	Decreased litter N, extractable N, litter C/N. No recovery of bryophytes and lichens.
(Power et al., 2006)	Heathland	8	Canopy height recovered. Differences remain in e.g. lichen frequency, total soil N.
(Jones, 2005)	Acid & calcareous grasslands	1	Some recovery of *Racomitrium* cover
"	"	4	Some recovery of cover of other moss and lichen species.
(Clark et al., 2009)	grassland	12	Partial decrease in N mineralisation rate
(Královec et al., 2009)	Meadow	18	Some differences still apparent
(Mountford et al., 1996)	Grassland	3-9	Recovery of cover of individual species (period depends on previous N application rate)
(Olff and Bakker, 1991)	Mown grassland	14	No decrease in productivity on sandy soil; decrease in productivity and increased species-richness on peat soil
(Stevens et al., 2012)	Hay meadow	15	Recovery of mean Ellenberg N only in highest treatment of 25 kg N ha^{-1} yr^{-1}; dominant species persist.
(Mitchell et al., 2004)	Oak woodland epiphytes	1	Decreased tissue N in one species
(Strengbom et al., 2001)	Boreal forest ground flora	19	Differences in species composition still apparent
"	"	47	Bryophytes not recovered

In summary, the range of response times varies from 1 year to more than 47 years, depending on the receptor and the habitat. Additional factors are also highly likely to play a part in the likelihood of recovery, which include the more obvious ones such as by how much the N deposition has declined, and from what annual dose. A major factor is how long the site had been receiving high N deposition, prior to the decrease in N deposition examined. This will determine the amount of N that will have accumulated in the plant and soil system, and will therefore influence recovery time. Less obvious factors include the loss rates of accumulated N from the system via biomass removal, fire (in heathlands), nitrate leaching or denitrification emissions of N_2 or N_2O gas. These loss factors are discussed in more detail in Jones et al. (2012) and Rowe et al. (2013).

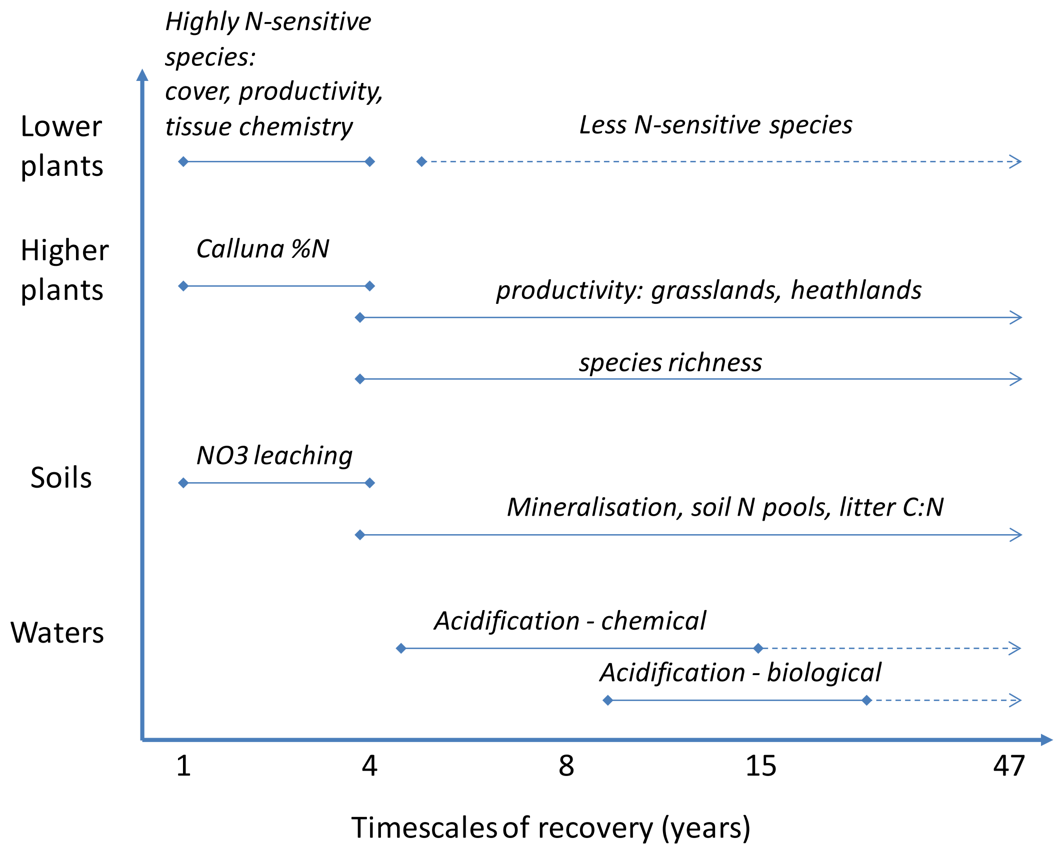

Figure 9.4. Summary of timescales of recovery for different terrestrial and freshwater receptors.

9.1.5 Timescales in relation to biodiversity targets

The targets agreed in the Nagoya / Aichi meeting of the Convention on Biological Diversity (CBD) are currently a major focus for international biodiversity conservation efforts (Secretariat of the Convention on Biological Diversity, 2011). The aim for most of these targets is that they are met by 2020. The "Aichi" targets that are particularly relevant for N impacts on biodiversity are:

Target 2: By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.

Target 4: By 2020, at the latest, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.

Target 8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.

Target 12: By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.

This date of 2020 is also reflected in strategy documents produced by the European Commission (EC) and by UK conservation bodies. The headline target of the EC biodiversity strategy is: "Halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss" (European Commission, 2011). In the UK, a strategic framework was provided the 2011 Natural Environment White Paper, and the Government and devolved administrations have developed strategy documents which also run up to the 2020 date. The strategy for England, for example, explicitly addresses air pollution concerns with a Priority Action: "Reduce air pollution impacts on biodiversity through approaches at national, UK EU and international levels targeted at the sectors which are the source of the relevant pollutants (nitrogen oxides, ozone, sulphur dioxide, ammonia).”

The EC strategy also looks further in the future, with a vision for 2050: “By 2050, European Union biodiversity and the ecosystem services it provides – its natural capital – are protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided.” However, the date of 2020 is the main timeframe that is currently being considered.

It will be clear from the discussion in the previous sections of habitat recovery timescales and chemical and biological delays that, even if N deposition were to be immediately reduced to pre-industrial levels, complete recovery of habitats by 2020 is not feasible. There have been considerable changes in ecosystems as a result of chronic elevated N deposition, and just as these have occurred over many decades, complete recovery will take a long time. Even the prevention of further biodiversity loss after 2020 may be unachievable, as the effects of long-term elevated N deposition above the critical load for a habitat may continue to cause the loss of species that are currently approaching the limit of N exposure that they can tolerate. In this context it should be noted that while European Union NO_x emissions are projected to decrease by a further 30% from 2010 to 2020 under the Gothenburg Protocol, at this scale NH_3 emissions are only projected to decrease by 2% over the same period (Sutton et al., 2013). With these modest commitments for 2020, it may be that the focus of attention increasingly shifts to the more ambitious reduction targets for 2030 in the European Commission’s proposal for revision of the National Emissions Ceilings Directive (EC, 2013).

The inability to simply turn off the effects of N pollution should not be taken as an excuse for inaction. It is equally clear that decreases in deposition rate have immediate benefits (for example, rapid partial recovery of epiphytic communities), as well as potentially reversing the long-term increase in stored N and thus reducing the risk of further species loss in future. From this perspective, it can be seen that reducing N emissions and N impacts on biodiversity represent long term processes, where both the societal changes in emissions and improvements in ecological condition can typically take several decades.

9.2 Evidence needed to demonstrate success

9.2.1 Indicators

In section 9.1.2, different aspects of a habitat that may be affected by N deposition were summarised. In principle, many of these aspects could be used as indicators of recovery from N pollution, but these receptors vary in how much and how quickly they respond to reduced N deposition, and in how easy these changes are to detect. In this section we review the indicators of N damage and recovery that have been proposed in reports such as Leith *et al.* (2005), Stevens *et al.* (2009) and Rowe *et al.* (2013) and assess which of these should be targeted when monitoring the success of measures aimed at decreasing N deposition.

Simple measurements such as pH or vegetation height can be useful in assessing N impacts. Plants also make good indicators and integrators of site conditions, and floristic records are extremely valuable. Measurements are particularly useful if they are repeated consistently over time, and the use of permanently-sited quadrats for botanical recording is strongly recommended. Methods suitable for on-site monitoring will be assessed in the next section. Methods for interpreting floristic changes will be discussed in section 9.2.3.

9.2.2 On-site monitoring

Evidence to demonstrate success can take various forms. Success can be demonstrated in the first instance by measuring the uptake or level of implementation of measures by the agricultural community, by industry or the number and extent of schemes set up by government. Secondly, it may be possible to demonstrate or to model changes in N deposition as a result of emission control measures being implemented. This can take two forms. At the site level it is possible to monitor changes in the pollution pressure on a site by measuring ammonia concentrations or N deposition over time. Ammonia and nitrogen dioxide concentrations can be monitored using passive samplers e.g. ALPHA badge-type passive diffusion samplers for ammonia (Tang et al. 2001) to provide average concentrations over approximately monthly periods, or via more intensive monitoring campaigns of shorter duration, using continuous sampling. Monitoring total N deposition at a site would require measurements of the nutrient chemistry and volumes of rainfall, in addition to ammonia and nitrogen dioxide measurements in order to obtain more accurate measures of N deposition than the modelled estimates available from the UK Air Pollution Information System (APIS, www.apis.ac.uk). From site-level up to national scale, it is possible to model the likely changes in deposition as a result of emission control measures being implemented. Site scale assessments using tools such as SCAIL (http://www.scail.ceh.ac.uk/) would involve estimating reductions in emissions from particular sources and modelling the resulting deposition, although this is subject to considerable uncertainty and would ideally be backed up by site-based measurements to confirm this. However, emission reductions may not be immediately apparent from model outputs, depending on the level of detail available in the model input data. For example, UK-scale modelling often uses national average data where no further detailed information is available. This is a limitation of using models to assess success. Thirdly, and more tangible to the site manager, is to measure local biological or biogeochemical measures which may demonstrate recovery.

On-site monitoring of biological or biogeochemical indicators may be used to demonstrate success over time. Plant parameters that could be measured include sward height, growth rates of individual species (e.g. shoot extension), or N content of foliage. Soil parameters that could be measured include N mineralisation rates or amounts of plant-available N in the soil. Measuring soil total N stocks are less useful since changes are difficult to detect (see section 9.1.2). Similarly, the soil total C:N ratio may increase or decrease as a result of N pollution, and is not a consistent indicator of recovery. Soil pH is likely to increase with recovery from the acidification caused by N, although this increase may be slow. Freshwater evidence of recovery from eutrophication includes a reduction in nitrate concentrations in lakes and streams, and an increase in pH or Acid Neutralising Capacity as evidence of recovery from acidification.

Measurement of species occurrence and abundance is relatively easy for many site managers, and is covered in more detail in the next section.

9.2.3 Interpreting floristic change

The effects of N pollution on plant and lichen species are very clear at a regional scale. For example, an analysis of the large floristic datasets that exist for the UK showed significant responses to N in the cover and presence of 91 plant and lichen species, including two BAP priority species and four species mentioned in Annexes of the Habitats Directive (Emmett et al., 2011). The main mechanisms are:

- direct toxic effects on sensitive bryophytes, lichens and higher plants
- stimulated growth and litter production, which shade out the shorter-growing species
- soil acidification, with reduced pH causing a loss of acid-sensitive species

Nitrogen pollution has caused decline and significant loss of species richness and diversity (Maskell et al., 2010; Stevens et al., 2004). Short-growing plant species are more likely to either have a greater threat status (i.e. nationally rare or scarce), or already be locally extinct (Hodgson et al., 2014). However, some species are nitrogen-loving and are therefore favoured by increased N deposition. These are mostly fast-growing species such as nettles, bramble or some grasses, which then out-compete slow-growing species. A number of floristic changes have been shown to be more likely on N-polluted sites:

- decreases in species richness

- loss of particular species
- increases in grass cover at the expense of forbs and heathland shrubs
- changes in average value for traits such as height, or Ellenberg indicator value for nutrients

It is recommended that a combination of these indicators is used rather than a single indicator, as not all may be apparent at N-polluted sites. Emmett et al. (2011) summarise for particular habitats where these changes have been shown to be significant at different rates of N deposition. But despite these clear patterns in large-scale survey data, it can be difficult to attribute changes at a site to N pollution. This is partly because effects of N on the competitive balance among plant species are similar to those from reduced management intensity, which is more easily identified as a problem, and partly because the loss of species has been gradual. Gradual impoverishment of habitats is difficult to observe (Miller, 2005), and species are likely to have already been lost due to the effects of N pollution over several decades.

The long-term nature of N pollution presents difficulties for all types of site monitoring. Current N deposition conditions usually reflect the site's history of N pollution, but other factors such as climate or site management history may also be important influences on site condition. Furthermore, changes in current deposition are inevitably small in relation to the amount of N that has accumulated in an ecosystem, so decreasing the N deposition rate might not result in immediate changes. Epiphytic bryophytes and lichens are an exception, and a small set of lichen species has been used to develop a simple index of current air quality (CEH et al., 2013).

Changes in the terrestrial flora are more difficult to attribute exclusively to N pollution, but several indicators are strongly correlated with N deposition rate and are a useful focus for monitoring.

a) Species-richness

Vegetation species richness (e.g. the number of plant species found by searching a defined area) has been shown to be negatively correlated with N deposition in many habitats. Species richness is widely seen as reflecting conservation value and may already be monitored.

b) Occurrence and cover of N-sensitive species

Nitrogen pollution drives species loss in order of species sensitivity. The species that are currently under threat from N pollution will therefore depend on the history of N deposition at the site and the current deposition, as the more sensitive species will already have moved from local abundance, to threatened, to eventual local extinction. Ideally, species that are currently vulnerable would be targeted in monitoring schemes since they are likely to be the most sensitive to changes in N deposition rate. If the current deposition rate is known, then the indicators that might be expected to change at different levels of N deposition could be identified by reference to Table 9.2, reproduced from Emmett et al. (2011). More detail is provided in Emmett et al. (2011) for each habitat on the particular response observed at each level of N deposition. Plant niche models such as GBMOVE (Smart et al., 2010) could also be used to identify species that are close to their environmental limit due to the N status of the site. Payne et al. (2013) show that in acid grassland, the abundance of individual species starts to change at the lowest measureable N deposition, i.e. changes in community composition are occurring below the current empirical critical load.

In the case of epiphytic lichens an approach has been taken based on the cover of different species groups, including nitrophytes (nitrogen-loving) and acidophytes (preferring more acidic substrates, which can be due to enhanced N deposition). The difference between these indices (acidophyte score minus nitrophyte score) provides a composite index of net eutrophication, in which a negative value indicates the site is dominated by nitrophytes and a positive score, by acidophytes. This index has proved highly sensitive to atmospheric ammonia concentrations (e.g. Wolseley et al., 2005; Sutton et al., 2009) and has also recently been extended to integrate interactions with bark pH and atmospheric NO_2 concentrations (CEH, NHM and University of Nottingham, 2013).

Table 9.2. Nitrogen deposition loads at which changes in trait-based indicators are likely, for four major UK habitats. Reproduced from Emmett et al. (2011).

Habitat	Upland or Lowland	Dataset[1]	Functional variable[2]	Slope of relationship	N deposition where a functional change is observed (20% change[3])	N deposition where a major functional change is observed (50% change[3])	Max change [3] observed (%)	N deposition where max % change[3] observed
Calcareous Grassland	Upland	VPD	Canopy Height	Positive	5-10	15-20	58%	20-25
Calcareous Grassland	Upland	VPD	Ellenberg N	Positive	10-15	35-40	upper end point	
Calcareous Grassland	Lowland	VPD	Ellenberg N	Humpback	Not Observed[4]	Not Observed	9%	25-30
Calcareous Grassland	Lowland	VPD	SLA	Humpback	Not Observed	Not Observed	4%	25-30
Calcareous Grassland	Lowland	LCS	Ellenberg N	Positive	10-15	Not Observed	upper end point	
Acid Grassland	Lowland	VPD	Canopy Height	Positive	30-35	45-50	upper end point	
Acid Grassland	Lowland	VPD	Ellenberg N	Positive	5-10	10-15	upper end point	
Acid Grassland	Upland	VPD	SLA	Humpback	Not Observed	Not Observed	5%	20-25
Acid Grassland	Lowland	LCS	Ellenberg N	Positive	30-35	Not Observed	upper end point	
Heathland	Lowland	VPD	Canopy Height	Positive	5-10	5-10	1500%	20-25
Heathland	Upland	VPD	Ellenberg N	Positive	15-20	Not Observed	upper end point	
Heathland	Lowland	VPD	Ellenberg N	Positive	15-20	Not Observed	upper end point	
Heathland	Lowland	LCS	Canopy Height	Negative	5-10	5-10	71%	10-15
Heathland	Lowland	LCS	Ellenberg N	Positive	5-10	10-15	upper end point	
Bog	Upland	VPD	Ellenberg N	Positive	15-20	Not Observed	upper end point	

[1] VPD - Vascular Plant Database; LCS – Local Change Survey
[2] SLA – Specific Leaf Area
[3] Change from baseline
[4] Not observed – 20% or 50% change not observed

c) Cover of functional groups, e.g. grass/forb cover ratio

The grass/forb cover ratio is a useful indicator for grasslands, since high values are associated with higher N deposition rates. Nitrogen has also been shown to affect the cover of *Sphagnum* species, which are important in bogs, and sub-shrubs, which are important in heathlands. However the grass/forb ratio can also be affected by site management such as grazing type and intensity.

d) Mean scores for plant traits

All plant species have distinctive characteristics and environmental requirements. For most UK species these characteristics and requirements have been measured and given scores in databases such as PlantAtt (Hill et al., 2004) and BryoAtt (Hill et al., 2007). The average value of the measurements/scores for the species that are present at a site can be useful and responsive indicators of site conditions. The following trait-means are particularly useful in assessing N impacts:

- **Ellenberg N score**. An indicator of the productivity of sites where a species typically grows.
- **Specific Leaf Area** (SLA, cm^2 leaf per g of leaf biomass. An indicator of whether the species is relatively competitive (large SLA) or stress-tolerant (small SLA).
- **Grime height score**. An indicator of the typical height of the species.

For all three of these trait means, a higher value implies greater N impacts. These trait means were referred to as “functional variables” in Emmett et al. (2011): Table 9.2 summarises the habitats and N deposition rates where significant changes in trait means have been observed.

In the vast majority of cases N pollution reduces the conservation value of sites, although a few species with conservation designations (*Platanthera bifolia* and several *Sphagnum* species) show increased prevalence in areas with higher N deposition (Emmett et al., 2011). It is therefore useful to distinguish two goals in assessing changes in the frequency and cover of plant and lichen species:

- to indicate whether N pollution is affecting the site
- to demonstrate changes to the nature conservation interest

For many species and habitats, damage from N pollution has developed gradually and chronically over many years, and it is likely that large areas of the UK have been significantly affected through an increase in ecosystem N status (Aber et al. 2003), and in reduced abundance of sensitive species. Demonstrating the extent to which observed changes in species or habitats are due to N pollution (whether ongoing, increased or decreased) requires an array of tools including long-term monitoring, experimental manipulations, targeted surveys along N deposition gradient with locations selected to control for co-varying factors such as climate, and paired studies which examine habitats in locations that receive different N inputs but similar climates. Such studies are beyond the scope of most site managers, but monitoring of N-sensitive indicators (e.g. grass/forb ratio, species richness, mean Ellenberg N score, and plant chemistry and soil-based indicators, discussed above) can provide circumstantial evidence that N is adversely affecting a site.

The extent to which these impacts reduce the conservation value of the site depends on the importance attached to these and other habitat changes. A recent consultation with habitat specialists at the Statutory Nature Conservation Bodies showed that they make extensive use of positive indicator species when assessing the value and condition of conservation sites (Rowe et al., 2014). Based on this insight, metrics are currently being developed to interpret changes in a set of species directly in terms of conservation value.

Key to demonstrating the success of remedies at individual sites is to establish monitoring of the baseline prior to implementation of any measures, in order to document improvement as a result of the intervention. This requires a consistent monitoring methodology, repeated at regular intervals over time, and preferably at the same locations, for example by installation of permanently marked vegetation quadrats. When quadrat locations are not fixed, as for example in the National Vegetation Classification (NVC) methodology which suggests placement in “homogenous stands”, changes are harder to detect, and conclusions may well be conservative (i.e. not detecting changes due to N pollution) because quadrats are placed to avoid transitional habitats. Therefore, permanently marked vegetation quadrats with repeat monitoring over time are strongly recommended. In terms of demonstrating success at a national scale, the same principles apply that there is a need for monitoring using a consistent methodology with repeat measurements over time at the same locations. At national scale this evidence can come from the Countryside Survey (e.g. Emmett et al, 2010), or from freshwater monitoring programmes such as the Acid Waters Monitoring Network (Monteith and Shilland, 2007). In fact, partial soil recovery from acidification effects has already been demonstrated in Countryside Survey data. Caution needs to be applied, as the Countryside Survey approach may reveal recovery in the broader UK landscape, but does not cover designated sites.

At designated sites, the only current national scale tool with the potential to demonstrate recovery from changes in N deposition is Common Standards Monitoring (CSM) and subsequent reporting to EU under Article 17. However, the current CSM methodology is designed for rapid assessment, and is not designed to detect gradual trends in species composition nor to attribute the causes of these changes, whether they are due to N deposition, climate change, site management or new emerging threats (Williams 2006). This is because CSM assessment typically records pass/fail against specified thresholds rather than recording values of e.g. cover of indicator species, the CSM positive and negative indicator species are not necessarily appropriate for monitoring changes due to N deposition as they were not selected solely for this purpose, and the CSM guidance suggests monitoring locations should be re-locatable but in practice it is not clear how frequently this is implemented. In theory, CSM could be supplemented by additional measurements, ranging in sophistication and complexity. The simplest might be incorporate recording of grass:forb ratio[3] at specified monitoring locations on-site. More powerful methods to detect change would involve setting up permanent monitoring quadrats to detect changes in the indicators discussed above. At sites where there is a particular interest in monitoring N impacts it would be necessary to set up multiple permanent quadrats. However, if the purpose is solely to monitor country-wide changes over time, only a few permanent quadrats might be needed per site, with the statistical power coming from analysing changes across sites. A further alternative is to identify sites or habitats with historical and re-locatable survey data and use this as a baseline for repeat surveys in future. As with any indicator, there is a basic need for high-quality repeat data over time. The same point applies when considering site condition in relation to local sources of atmospheric N deposition. In these cases, assessment of the condition of a site can benefit substantially from comparison with a clean 'reference site' under similar climate conditions to provide a baseline to compare with the levels of deposition and species occurring at the target site (e.g. Sutton et al., 2011).

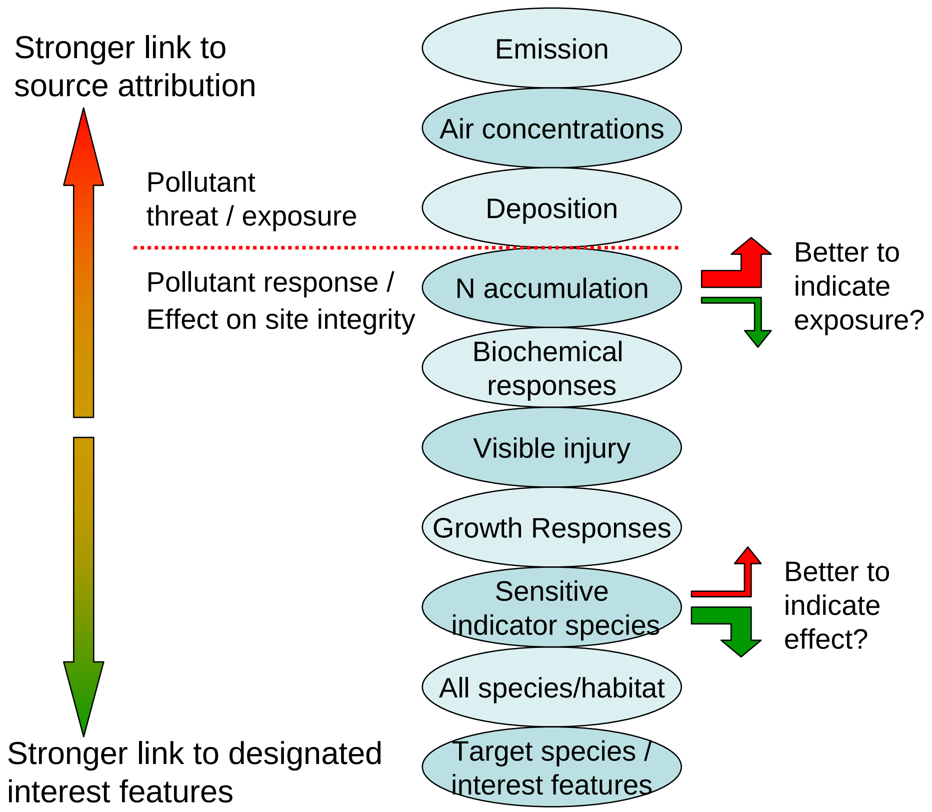

Figure 9.5. Overview of the "biomonitoring chain" showing how different indicator measurements may be ordered from pollutant source to ultimate pollutant impacts. Measurements closer to emission show a stronger link to source attribution, but weaker link to effects on designated interest features. Conversely, species-based measurements show a close link to the designated interest features, but a weak link to source attribution. A comprehensive robust program of biomonitoring should therefore combine measurements distributed along the biomonitoring chain. Dark shaded ellipses show the typically most practicable approaches (Leith et al., 2005; Sutton et al., 2011).

[3] Ideally separately record the percentage cover of 3 categories: graminoids, forbs and remaining species. This allows calculation of several different grass:forb metrics.

In building up a robust body of evidence to demonstrate success of remedies it is useful to consider the concept of the 'biomonitoring chain' (Figure 9.5, Ch 16 in Leith et al., 2005; Sutton et al., 2011; Hall et al. 2012). This recognises that the indicators most closely related to features of interest are better to indicate effect, but may be influenced by factors other than N emissions and deposition. Conversely, the indicators most closely linked to source attribution are not necessarily linked to biological change. The concept of the biomonitoring chain therefore links key indicators across the series from emission to deposition to species response in order to build a robust package of observations that can demonstrate the success of measures to reduce N threats. The dark shaded ellipses in Figure 9.5 are those that are easier to implement, and fortunately are distributed relatively evenly along the biomonitoring chain.

References

Aber J.D., McDowell W., Nadelhoffer K.J., Magill A., Berntson G., Kamakea M., McNulty S., Currie W., Rustad L., Fernandez I. (1998) Nitrogen saturation in temperate forest ecosystems: hypotheses revisited. Bioscience 48, 921–934.

Aber J. D., Goodale C. L., Ollinger S. V. et al. (2003) Is nitrogen deposition altering the nitrogen status of northeastern forests? BioScience, 53, 375–389.

Armitage H.F., Britton A.J., Woodin S.J., van der Wal R. (2011) Assessing the recovery potential of alpine moss-sedge heath: Reciprocal transplants along a nitrogen deposition gradient. Environmental Pollution 159, 140-147.

Battarbee R. W., G. L. Simpson et al. (2014) "Recovery of UK lakes from acidification: An assessment using combined palaeoecological and contemporary diatom assemblage data." Ecological Indicators 37, Part B(0): 365-380.

Bealey W.J., Braban C.F., Famulari D., Dragosits U., Dore A.J., Nemitz E., Tang Y.S., Twigg M., Leeson S., Sutton M.A., Loubet B., Valatin G., Wheat A., Helfter C., Coyle M., Williams A., Sandars D,L. (2013) Agroforestry for ammonia abatement summary report. CEH/Defra

Beevers S.D. and Carslaw D.C (2005) The impact of congestion charging on vehicle emissions in London. Atmospheric Environment 39(1), 1-5.

Boxman A.W., van der Ven P.J.M., Roelofs J.G.M. (1998) Ecosystem recovery after a decrease in nitrogen input to a Scots pine stand at Ysselsteyn, the Netherlands. Forest Ecology and Management 101, 155-163.

CEH, NHM, University of Nottingham (2013) Guide to using a lichen based index to nitrogen air quality. OP158. Field Studies Council.

Clark C.M., Hobbie S.E., Venterea R., Tilman D. (2009) Long-lasting effects on nitrogen cycling 12 years after treatments cease despite minimal long-term nitrogen retention. Global Change Biology 15, 1755-1766.

Dise N. B., Ashmore M. R., Belyazid S., Bobbink R., De Vries W., Erisman J. W., Spranger T., Stevens C. and van den Berg L. (2011) Nitrogen as a threat to European terrestrial biodiversity. In Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., Grinsven, H. V., & Grizzetti, B. (Eds.), The European nitrogen assessment . Cambridge: Cambridge University Press.

Dise N.B., Matzner E., Gundersen P. (1998) Synthesis of nitrogen pools and fluxes from European forest ecosystems. Water, Air and Soil Pollution, 105, 143-154.

EC (2013). http://ec.europa.eu/environment/air/pollutants/ceilings.htm

Edmondson J., Terribile E., Carroll J.A., Price E.A.C., Caporn S.J.M. (2013) The legacy of nitrogen pollution in heather moorlands: Ecosystem response to simulated decline in nitrogen deposition over seven years. The Science of the total environment 444, 138-144.

Emmett B.A. (2007) Nitrogen saturation of terrestrial ecosystems: Some recent findings and their implications for our conceptual framework. . Water Air and Soil Pollution: Focus 7, 99-109.

Emmett B.A., Reynolds B., Chamberlain P.M., Rowe E., Spurgeon D., Brittain S.A. et al. (2010) Countryside Survey: Soils report from 2007. NERC/Centre for Ecology & Hydrology, 2010, pp. 192.

Emmett B.A., Rowe E.C., Stevens C.J., Gowing D.J., Henrys P.A., Maskell L.C., Smart S.M. (2011) Interpretation of evidence of nitrogen impacts on vegetation in relation to UK biodiversity objectives. JNCC Report 449. JNCC, Peterborough., p. 105.

European Commission (2011) Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Communication from the Commission (COM(2011) 244).

Hall J., Field C., Stevens C., Leith I., Sheppard L., Bealey B., Caporn S. and Sutton M. (2012) Air Quality Risk Assessment and SSSI Survey project: To refine the understanding of air quality risk assessments (based on modelling) and site specific responses to atmospheric ammonia pollution. Final report to Natural England ITT_183-23875, Sept 2012.

Hill M.O., Preston C.D., Bosanquet S.D.S., Roy D.B. (2007) BRYOATT - attributes of British and Irish mosses, liverworts and hornworts with information on native status, size, life form, life history, geography and habitat. Centre for Ecology and Hydrology.

Hill M.O., Preston C.D., Roy D.B. (2004) PLANTATT: Attributes of British and Irish plants: Status, size, life history, geography and habitats. Centre for Ecology and Hydrology. Available from http://nora.nerc.ac.uk/9535/.

Hodgson J.G., Tallowin J., Dennis R.L.H., Thompson K., Poschlod P., Dhanoa M.S., Charles M., Jones G., Wilson P., Band S.R., Bogaard A., Palmer C., Carter G., Hynd A. (2014) Leaf nitrogen and canopy height identify processes leading to plant and butterfly diversity loss in agricultural landscapes . Functional Ecology doi: 10.1111/1365-2435.12253.

Jones M.L.M., Wallace H., Norris D.A., Brittain S.A., Haria S., Jones R.E., Rhind P.M., Williams P.D., Reynolds B. and Emmett B.A. (2004) Changes in vegetation and soil characteristics in coastal sand dunes along a gradient of atmospheric nitrogen deposition. Plant Biology 6, 598-605.

Jones M.L.M. (2005) Nitrogen deposition in upland grasslands: Critical loads, management and recovery. PhD Thesis, University of Sheffield.

Jones L., Curtis C., Tipping E., Hall J. and B, E. (2012) Impacts of climate change on critical loads for nutrient nitrogen, acidity and heavy metals. Report to Defra as component of UKREATE consortium.

Jones L., Provins A., Harper-Simmonds L., Holland M., Mills G., Hayes F., Emmett B.A., Hall J., Sheppard L.J., Smith R., Sutton M., Hicks K., Ashmore M., Haines-Young R. (2014) A review and application of the evidence for nitrogen impacts on ecosystem services. Ecosystem Services 7, 76–88. http://dx.doi.org/10.1016/j.ecoser.2013.09.001

Královec J., Pocová L., Jonášová M., Petr M., Larel P. (2009) Spontaneous recovery of an intensively used grassland after cessation of fertilizing. Applied Vegetation Science 12, 391-397.

Leith I.D., van Dijk N., Pitcairn C.E.R.,Wolseley P.A., Whitfield C.P. and Sutton M.A. (2005) Biomonitoring methods for assessing the impacts of nitrogen pollution: refinement and testing (eds). Report 386. Joint Nature Conservation Committee, Peterborough, 290 pp. www.jncc.gov.uk/default.aspx?page=3886

Maskell L.C., Smart S.M., Bullock J.M., Thompson K., Stevens C.J. (2010) Nitrogen deposition causes widespread loss of species richness in British habitats. Global Change Biology 16, 671-679.

Miller J.R. (2005) Biodiversity conservation and the extinction of experience. Trends in Ecology & Evolution 20, 430-434.

Mitchell R.J., Sutton M.A., Truscott A.M., Leith I.D., Cape J.N., Pitcairn C.E.R., Van Dijk N. (2004) Growth and tissue nitrogen of epiphytic Atlantic bryophytes: effects of increased and decreased atmospheric N deposition. Functional Ecology 18, 322-329.

Monteith D.T., Shilland E.M. (2007) The United Kingdom Acid Waters Monitoring Network: Assessment of the First 18 Years of Data. Data summary annexe accompanying research project final report. Report to Defra (contract EPG 1/3/160). ENSIS Ltd., London.

Mountford J.O., Lakhani K.H., Holland, R.J. (1996) Reversion of grassland vegetation following the cessation of fertilizer application. Journal of Vegetation Science 7, 219-228.

Murphy J. F., J. H. Winterbottom, et al. (2014) "Evidence of recovery from acidification in the macroinvertebrate assemblages of UK fresh waters: A 20-year time series." Ecological Indicators 37, Part B(0): 330-340.

Olff H., Bakker J.P. (1991) Long-term dynamics of standing crop and species composition after the cessation of fertiliser application to mown grassland. Journal of Applied Ecology 28, 1040-1052.

Payne R.J., Dise N.B., Stevens C.J., Gowing D.G., and BEGIN Partners (2013). Impact of nitrogen deposition at the species level. PNAS 110, 984–987.

Power S.A., Green E.R., Barker C.G., Bell J.N.B., Ashmore M.R. (2006) Ecosystem recovery: heathland response to a reduction in nitrogen deposition. Global Change Biology 12, 1241-1252.

RoTAP (2012) Review of transboundary air pollution: Acidification, eutrophication, ground level ozone and heavy metals in the UK. Contract Report to the Department for Environment, Food and Rural Affairs. Centre for Ecology & Hydrology.

Rowe E.C., Ford-Thompson A., Monteith D., van Hinsberg A., Smart S., Henrys P., Ashmore M. (2014) A biodiversity metric for interpreting outputs of models of atmospheric nitrogen pollution impacts on habitats. Final report on Defra project AQ0823, "Research to define a biodiversity metric to inform the UK's response to the CCE 'Call for Data 2012-2014'. CEH project NEC04988. Centre for Ecology and Hydrology, ECW, Deiniol Road, Bangor, LL57 2UW., p. 90.

Rowe E.C., Jones L., Stevens C.J., Vieno M., Dore A.J., Hall J., Sutton M.A., Mills G., Evans C.D., Helliwell R.C., Britton A.J., Mitchell R.J., Caporn S.J., Dise N.B., Field C., Emmett B.A. (2013) Measures to evaluate benefits to UK semi-

natural habitats of reductions in nitrogen deposition. Final report on REBEND project (Defra AQ0823; CEH NEC04307). Centre for Ecology and Hydrology, ECW, Deiniol Road, Bangor, LL57 2UW, UK., p. 73.
Secretariat of the Convention on Biological Diversity, 2011. Strategic plan for biodiversity 2011–2020 and the Aichi Targets., p. 4.
Smart S.M., Scott W.A., Whitaker J., Hill M.O., Roy D.B., Critchley C.N., Marini L., Evans C., Emmett B.A., Rowe E.C., Crowe A., Le Duc M., Marrs R.H. (2010) Empirical realised niche models for British higher and lower plants - development and preliminary testing. Journal of Vegetation Science 21, 643-656.
Stevens C.J., Caporn S.J.M., Maskell L.C., Smart S.M., Dise N.B., Gowing D.J. (2009) Detecting and attributing air pollution impacts during SSSI condition assessment. JNCC, Peterborough. JNCC Report No. 426.
Stevens C.J., Dise N.B., Mountford J.O., Gowing D.J. (2004) Impact of nitrogen deposition on the species richness of grasslands. Science 303, 1876-1879.
Stevens C.J., Mountford J.O., Gowing D.J.G., Bardgett R.D. (2012) Differences in yield, Ellenberg N value, tissue chemistry and soil chemistry 15 years after the cessation of nitrogen addition. Plant and Soil 357, 309-319.
Strengbom J., Nordin A., Nasholm T., Ericson L. (2001) Slow recovery of boreal forest ecosystem following decreased nitrogen input. Functional Ecology 15, 451-457.
Sutton M.A., Wolseley P.A., Leith, I.D. van Dijk N, Tang Y.S., James P.W., Theobald M.R. and Whitfield, C.P. (2009) Estimation of the ammonia critical level for epiphytic lichens based on observations at farm, landscape and national scales. Chapter 6, in: Atmospheric Ammonia: Detecting emission changes and environmental impacts (eds. M.A. Sutton, S. Reis and S.M.H. Baker), pp 71-86, Springer.
Sutton M.A., Leith I.D., Bealey W.J., van Dijk N., Tang Y.S. (2011) Moninea Bog: A case study of atmospheric ammonia impacts on a Special Area of Conservation. In: 'Nitrogen Deposition and Natura 2000: Science & practice in determining environmental impacts' (Eds: W.K. Hicks, C.P. Whitfield, W.J. Bealey, and M.A. Sutton), pp 59-71. COST Office, Brussels.
Sutton M.A., Reis R., Riddick S.N., Dragosits U., Nemitz E., Theobald M.R., Tang Y.S., Braban C.F., Vieno M., Dore A.J., Mitchell R.F., Wanless S., Daunt F., Fowler D., Blackall T.D., Milford. C., Flechard C.F., Loubet B., Massad R., Cellier P., Coheur P.F., Clarisse L., van Damme M., Ngadi, Y., Clerbaux C., Skjøth C.A., Geels C., Hertel O., Wichink Kruit R.J., Pinder, R.W., Bash J.O., Walker J.D., Simpson D., Horvath, L., Misselbrook, T.H., Bleeker A., Dentener F. &Wim de Vries V. (2013) Toward a climate-dependent paradigm of ammonia emission & deposition. Phil. Trans. Roy. Soc. (Ser. B). 368: 20130166. http://dx.doi.org/10.1098/rstb.2013.0166
Tang Y.S., Cape J.N., Sutton M.A. (2001) Development and types of passive diffusion samplers for monitoring atmospheric NO2 and NH3 concentrations., Proceedings of the International Symposium on Passive Sampling of Gaseous Air Pollutants in Ecological Effects Research. TheScientificWorld, pp. 513-529.
Williams D., Emmett B.A., Brittain S.A., Reynolds B., Stevens P.A. & Benham D. (2004) The GANE Roof Project: the impact of reduced N and S deposition and experimental warming in an acid grassland. Water Air & Soil Pollution: Focus 4(6), 187-196.
Williams J.M. (ed.) (2006) Common Standards Monitoring for Designated Sites: First Six Year Report. Peterborough, JNCC.
Wolseley P.A., James P. W., Theobald M. R. and Sutton M.A. (2006) Detecting changes in epiphytic lichen communities at sites affected by atmospheric ammonia from agricultural sources. The Lichenologist 38(2): 161-176.