Nitrogen mitigation

A review of nitrogen deposition impacts and mitigation potential in Scottish semi-natural ecosystems

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Summary

- Deposition of reactive nitrogen from anthropogenic sources greatly exceeds natural fluxes and, in Scotland, a large proportion of semi-natural ecosystems (i.e., communities of native species, modified to varying degrees by human activities) currently experience nitrogen deposition loads in excess of the thresholds at which adverse impacts have been reported.

- This review provides a summary of current knowledge on the thresholds for adverse nitrogen impacts in Scottish terrestrial semi-natural ecosystems, ecosystem responses to excess nitrogen deposition, the scope for mitigating the impacts of nitrogen deposition through habitat management and metrics which could be used to monitor ecosystem recovery.

- The recent (2022) review of empirical critical loads of nitrogen for Europe was used to identify critical load thresholds and responses to excess nitrogen for terrestrial semi-natural ecosystems present in Scotland and to determine knowledge gaps.

- At present, critical loads of nitrogen are known for around 60% of Scottish habitats. The remaining 40% for which we have no knowledge of thresholds for nitrogen impacts, include many habitats of significance for biodiversity in Scotland especially communities of rocky areas, alpine habitats, scrub, and wetlands.

- Most knowledge on ecosystem responses to excess nitrogen is focussed on the response of plant communities (particularly vascular plants) and plant and soil chemical parameters. There are significant knowledge gaps around the responses of belowground biodiversity and above ground fauna.

- A Web of Science literature search was used to identify the current state of knowledge on the rate and completeness of semi-natural ecosystem recovery from excess nitrogen deposition and the scope to improve outcomes through habitat management. The search produced 1593 studies, of which, after screening of titles, abstracts and full texts, 130 were identified as being relevant to natural recovery from or mitigation of nitrogen impacts in Scottish semi-natural ecosystems.

- Evidence on the rate and extent of natural recovery was found for bog, grasslands, alpine moss heath, dwarf-shrub heath and forest ecosystems. In general, the evidence suggests that some chemical parameters including soil pH, nitrogen availability and leaching of excess nitrogen to surface waters may recovery fairly quickly (timescale of months to years) when nitrogen deposition declines. Bryophyte or lichen tissue chemistry may also show relatively rapid recovery (months to years). Nitrogen stocks in soils and vegetation often remain elevated in the long term however, and diversity and composition of plant, fungal and animal communities can also be slow to recover (limited recovery over decadal timescales).

- 60% of Scottish semi-natural ecosystems are not actively managed and for these ecosystems natural recovery is generally the best or only option for recovery from nitrogen pollution. Action to improve air quality and reduce nitrogen deposition loads will be essential to improve the long-term outcomes for these ecosystems.

- Evidence on the effectiveness of habitat management interventions to improve or accelerate recovery from excess nitrogen deposition was found for dunes, fens, grasslands, alpine moss heaths, dwarf-shrub heaths, and forests. In many ecosystems the number of studies was limited and only a small number of potential management options and ecosystem responses had been studied. Most studies were from the southern UK or from Europe, with few studies conducted under Scottish climate and soil conditions.

- There was evidence that some management activities could be used to help mitigate nitrogen impacts. Liming was able to reduce acidification in grasslands and forests and grazing or mowing improved sward structure and helped to maintain species richness in grasslands. Management options were best explored in heathlands with burning, grazing, mowing and turf cutting all showing potential to reduce nitrogen stocks.

- Management interventions could also have undesirable negative side-effects, for example increased nitrogen leaching following vegetation/soil disturbance, loss of carbon stocks, or reduction in habitat suitability for fauna. Detailed assessment of benefits and trade-offs for nitrogen mitigation in heathlands concluded that there was a risk that trying to solve one problem could create another. Any potential mitigation managements require a clear understanding of impacts under Scottish conditions before use.

- Monitoring of ecosystem recovery in response to declining nitrogen deposition or mitigation action requires suitable nitrogen impact indicator metrics and progress with developing such metrics is briefly reviewed.

- Although many measurements have been used to demonstrate impacts of excess nitrogen deposition on ecosystem structure and function, few have been tested and developed for use as indicator metrics suitable for use at single sites.

- Average Accumulated Exceedance (the amount of nitrogen deposited in excess of the critical load) is widely accepted as metric of nitrogen deposition pressure and is used for UK-level biodiversity indicator reporting but does not inform on nitrogen impacts.

- Potential indicators of impact include simple chemical measures such as moss or plant tissue nitrogen content, or more complex botanical measurements such as species richness, graminoid:forb ratio or lichen diversity. Initial testing of these potential indicators shows that not all indicator metrics will be effective in all habitats. Since relationships between biodiversity and nitrogen deposition are influenced by a variety of local factors, it is suggested that the best approach would be to develop a ‘basket’ of metrics for each habitat, with inference of nitrogen impacts being based on a weight of evidence approach rather than relying on the pass/fail of a single metric. All metrics will need benchmarking data to allow interpretation of results for single sites within the national context, but this is currently lacking for most potential indicators.

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Background

Deposition of reactive nitrogen from anthropogenic sources now greatly exceeds that from natural processes, impacting on biodiversity and functioning of ecosystems. Deposited nitrogen accumulates in plants and soils and any which is not retained by the ecosystem is transported downslope and can leach into ground and surface waters. Increased availability of nitrogen alters species' physiology and growth and changes the interactions between species with consequences for biodiversity. In Scotland, a large proportion of semi-natural ecosystems (i.e., communities of native species modified to varying degrees by human activity) receive nitrogen deposition loads above the thresholds (known as critical loads) at which negative impacts have been reported. Efforts to reduce nitrogen emissions have had some success in controlling emissions of oxidised nitrogen which have shown a downward trend, but emissions of reduced nitrogen have not declined. Deposition of excess nitrogen (i.e., amounts of nitrogen above the critical load threshold) remains a widespread problem in Scotland, widely impacting the condition of semi-natural habitats and the supply of associated ecosystem services.

Emissions reduction must remain the policy priority. However, many Scottish semi-natural ecosystems are managed through a combination of grazing, mowing, burning and other activities. This provides a potential opportunity to mitigate the effects of excess nitrogen deposition through habitat management including physical removal of nitrogen-containing materials or alteration of vegetation structure to promote biodiversity. Identification of the success or otherwise of such mitigation measures and of efforts to reduce excess nitrogen deposition to allow natural recovery requires development and selection of appropriate metrics which can be readily implemented over a large scale to monitor effects on both protected areas and in the wider countryside.

Objectives

The objectives of this review were:

1. To provide a summary of the thresholds for adverse impacts of nitrogen deposition in Scottish semi-natural terrestrial ecosystems and key ecosystem responses to excess nitrogen deposition, and to identify those ecosystems for which knowledge of thresholds and impacts is lacking.

2. To review current knowledge on the potential of habitat management to mitigate the effects of nitrogen deposition on Scottish semi-natural terrestrial ecosystems and the timescales for 'natural recovery' in the absence of mitigation action.

3. To make recommendations for appropriate metrics which could be used to monitor impacts of, and recovery from, excess nitrogen deposition in Scottish semi-natural ecosystems.

To address the first objective, we drew on the recently published review and revision of empirical critical loads of nitrogen for Europe (Bobbink *et al.*, 2022). This document presents an up-to-date review of empirical evidence for nitrogen impacts and ecosystem responses in European semi-natural ecosystems. We produced a table summarizing the evidence for thresholds and impacts in ecosystems relevant to Scotland and used this to identify those ecosystems in Scotland for which evidence is currently lacking. For the second objective we conducted a review of the literature on mitigation of nitrogen impacts through habitat management and timescales for natural recovery in Scottish terrestrial ecosystems and their close analogues in cool temperate and boreal zones worldwide. Based on the results of the first two objectives we then identified potential metrics of nitrogen impact and recovery which could be used to monitor the success of mitigation actions both in protected areas and in the wider countryside.

Section 1: Critical loads and response thresholds for nitrogen impacts in Scottish terrestrial ecosystems and key ecosystem responses

Introduction

Nitrogen is an essential element for plant growth and, under pristine conditions, limitation in its supply can be a strong constraint on ecosystem productivity (LeBauer & Treseder, 2008), particularly in the naturally nutrient-poor habitats which predominate in Scotland. Inert nitrogen gas in the atmosphere is naturally converted to reactive nitrogen (which can be utilised by plants and other organisms) by lightning and by nitrogen fixing bacteria. However, these natural fluxes are now greatly exceeded by anthropogenic inputs of reactive nitrogen from industrially manufactured fertilisers, combustion processes and agricultural sources (Galloway *et al.*, 2008). Anthropogenic nitrogen emissions can be transported far from their original sources by atmospheric processes and are deposited on semi-natural ecosystems, greatly increasing nitrogen availability. This increased nitrogen supply impacts terrestrial ecosystems through a variety of mechanisms including eutrophication, acidification, altered sensitivity to secondary stressors (e.g. drought, frost, pests), and direct toxicity (Bobbink *et al.*, 2010; Dise *et al.*, 2011; Stevens *et al.*, 2020).

Critical loads are defined as *'a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge'* (Bobbink *et al.*, 2010). Empirical critical loads of nitrogen are set based on observations of changes in the structure or functioning of ecosystems, either from field or mesocosm studies where nitrogen deposition loads have been experimentally manipulated, or from survey-based studies where ecosystem properties are examined over a range of nitrogen deposition levels. In Europe, empirical critical loads of nitrogen are set through a process of expert scientific working groups which meet to assess all the available evidence from both peer-reviewed and 'grey' literature and assign both a critical load range and an indication of confidence ('expert judgement', 'quite reliable' or 'reliable') for each habitat. To be included in the critical loads review process, studies must meet certain criteria, including that any nitrogen treatments must have a duration of at least two years, that background deposition rates in field experiments should be accurately estimated and that nitrogen treatments should preferably include low-level additions close to the critical load. Long term experiments in low background deposition areas of Europe are particularly useful. Critical loads are set as a range, e.g., 5-15 kg N ha^{-1} yr^{-1}, which is intended to capture the uncertainty in the exact threshold for impacts due to factors such as intervals between nitrogen addition levels in experiments, within ecosystem variation in response and uncertainty in estimated total nitrogen deposition levels.

Critical loads are reviewed on an approximately 10-yearly cycle and any new evidence arising since the last review is used to refine the critical loads ranges and confidence, or to add new critical loads for additional habitats. Over time, as evidence for nitrogen impacts on different ecosystems accumulates and a wider range of ecosystem attributes is investigated, empirical critical loads for most habitats have been lowered and uncertainty ranges have narrowed. The most recent review of empirical critical loads for nitrogen took place during 2020-2022 and reviewed all new evidence available up to the summer of 2021 (Bobbink *et al.*, 2022).

In this section of the report, we draw on the 2022 review and revision of empirical critical loads of nitrogen for Europe (Bobbink *et al.*, 2022) to determine the critical loads or thresholds for nitrogen impacts in Scottish semi-natural terrestrial ecosystems and the key ecosystem responses to excess nitrogen. We also explore where the key gaps in knowledge on Scottish ecosystem response to nitrogen lie and the priorities for additional research.

Methods

To determine the coverage of critical loads information for Scottish ecosystems, we first compiled an inventory of terrestrial semi-natural habitats present in Scotland based on Strachan (2017). Habitats were described according to the European Nature Information System (EUNIS) level 2 and 3 categories as updated according to Chytry *et al.* (2020) and the 2021/22 EUNIS revision (https://eunis.eea.europa.eu/habitats.jsp). Appendix 1 gives the equivalent British National Vegetation Classification (NVC) and Natura 2000 habitat codes for reference. We then used the recently completed 2022 revision and update of empirical critical loads for Europe (Bobbink *et al.*, 2022) to identify for each habitat a) if a critical load had been assigned and, b) the currently identified threshold for nitrogen impacts. For those habitats for which no critical load has yet been set, we further considered if there was an appropriate similar habitat which could be used to approximate the likely threshold for nitrogen impacts. We then used the habitat evidence summaries in the critical loads report to identify what changes in ecosystem structure, properties or function have been associated with excess nitrogen deposition in each of the habitats.

Evidence for critical loads and thresholds for nitrogen impacts in Scottish ecosystems.

Of the 76 EUNIS level 2 or 3 terrestrial semi-natural habitats present in Scotland, 45 have sufficient evidence on thresholds for nitrogen impacts for a critical load to be assigned (Table 1). These habitats include coastal dunes and heaths, bogs, mires and fens, most grasslands, temperate and alpine heaths and both coniferous and deciduous forests. For 19 of these habitats, including alpine heaths and scrub, bogs and mires, and some forests, critical loads are set for the broader EUNIS level 2 category and do not distinguish between individual level 3 habitats. This is usually because there is insufficient data to determine whether the level 3 habitats within a group might exhibit different responses to nitrogen; additional studies would therefore be useful for these habitats to further resolve the critical loads values.

Although more than half of Scottish terrestrial semi-natural habitats are covered by critical loads, 31 habitats currently have no critical load for nitrogen assigned (Table 1). These include coastal habitats of cliffs, littoral rocks and shingle, helophyte beds (wetland vegetation fringing lakes and rivers), woodland fringe communities, scrub habitats, snowbeds and a wide range of rocky habitats including cliffs, scree and fell field. Most of these communities are likely to be responsive to nitrogen deposition, and this represents a significant gap in knowledge. Future research should prioritise filling these knowledge gaps.

Ecosystem responses which have been associated with exceedance of critical load thresholds are summarised in Table 2. Many studies focus on a particular aspect of ecosystem response to nitrogen such as plant community composition or nitrogen stocks and fluxes, and so evidence of exceedance impacts tends to develop over time as new studies fill knowledge gaps. Many ecosystems exhibit broadly similar responses to critical load exceedance and changes which are commonly observed across a range of ecosystems may have potential as indicators of nitrogen impact and recovery. Responses of vegetation (particularly vascular plants) and soils have been most commonly studied. For soils, increase in nitrogen leaching was the most commonly reported response, along with increases in nitrogen mineralisation rates and decreased soil C:N ratio. Decreases in root biomass and changes in mycorrhizal fungal and soil fauna communities were also reported across a number of habitats, but in general below ground biodiversity is much less studied than above ground.

In terms of vegetation, change in species composition was the single most widely reported response to excess nitrogen deposition, reflecting the large number of studies which have investigated plant community responses. Increases in vegetation biomass and nitrophilous species, and decreases in species richness, oligotrophic species and positive indicator species were also widely reported. The vascular plant community had the greatest number of nitrogen responses reported. The most frequently observed changes in the vascular community were increased graminoid cover, increased vascular plant cover or biomass (particularly tall species) and increased plant tissue nitrogen content. Fewer aspects of bryophyte and lichen communities have been investigated but decreased cover of both bryophytes and lichens was one of the most widely reported responses to nitrogen, along with decreased lichen richness, altered bryophyte community composition and increases in tissue nitrogen of both groups.

Table 1. *Empirical Critical Loads (CL_{emp}) for nitrogen in Scottish terrestrial semi-natural ecosystems. For some habitats critical loads are set at EUNIS level 2 because there is insufficient evidence to define critical loads for separate level 3 habitats. *In MA22 (saltmarsh), a higher critical load of 20-30 kg N ha^{-1} yr^{-1} is assigned to pioneer marshes and a lower critical load of 10-20 kg N ha^{-1} yr^{-1} to mid-low and upper-mid marshes.*

EUNIS code	EUNIS 2022 habitat name	CL_{emp} available Y/N?	2022 CL_{emp} (kg N ha^{-1} yr^{-1})	If no CL_{emp} is there a close analogue habitat?
M	**Marine benthic habitats**			
MA1	***Littoral rock***	**N**	**NA**	**N**
MA2	***Littoral biogenic habitat***			
MA22	Atlantic littoral biogenic habitat (saltmarsh)	Y	10-20 or 20-30*	
N	**Coastal habitats**			
N1	***Coastal dunes and sandy shores***			
N11	Atlantic, Baltic and Arctic sand beach	N	NA	N
N13	Atlantic and Baltic shifting coastal dune	Y	10-20	
N15	Atlantic and Baltic coastal dune grassland (grey dune)	Y	5-15	
N18	Atlantic and Baltic coastal *Empetrum* heath	Y	10-15	
N19	Atlantic coastal *Calluna* and *Ulex* heath	Y	10-15	
N1A	Atlantic and Baltic coastal dune scrub	N	NA	N
N1H	Atlantic and Baltic moist and wet dune slack	Y	5-15	
N2	***Coastal shingle***			
N21	Atlantic, Baltic and Arctic coastal shingle beach	N	NA	N
N23	Shingle and gravel beach with scrub	N	NA	N
N24	Shingle and gravel beach forest	N	NA	N
N3	***Rock cliffs, ledges and shores, including the supralittoral***			
N31	Atlantic and Baltic rocky sea cliff and shore	N	NA	N
N24	Atlantic and Baltic soft sea cliff	N	NA	N
Q	**Wetlands**			
Q1	***Raised and blanket mires***	**Y**	**5-10**	
Q11	Raised bog	level 2	5-10	
Q12	Blanket bog	level 2	5-10	
Q2	***Valley mires, poor fens and transition mires***	**Y**	**5-15**	
Q21	Oceanic valley mire	level 2	5-15	
Q22	Poor fen	level 2	5-15	
Q24	Intermediate fen and soft-water spring mire	level 2	5-15	
Q25	Non-calcareous quaking mire	level 2	5-15	
Q4	***Base-rich fens and calcareous spring-mires***			
Q41	Alkaline, calcareous, carbonate-rich small-sedge spring fen	Y	15-25	
Q42	Extremely rich moss-sedge fen	Y	15-25	
Q43	Tall-sedge base-rich fen	Y	15-25	
Q44	Calcareous quaking mire	Y	15-25	
Q45	Arctic-alpine rich fen	Y	15-25	

Q5	***Helophyte beds***			
Q51	Tall-helophyte bed	N	NA	N
Q52	Small-helophyte bed	N	NA	N
Q53	Tall-sedge bed	N	NA	N
R	**Grasslands and lands dominated by forbs mosses or lichens**			
R1	***Dry grasslands***			
R1A	Semi-dry perennial calcareous grassland (meadow steppe)	Y	10-20	
R1M	Lowland to montane, dry to mesic grassland usually dominated by *Nardus stricta*	Y	6-10	
R1P	Oceanic to subcontinental inland sand grassland on dry acid and neutral soils	Y	5-15	
R1S	Heavy-metal grassland in western and central Europe	N	NA	Y (R1A)
R2	***Mesic grasslands***			
R21	Mesic permanent pasture of lowlands and mountains	N	NA	N
R22	Low and medium altitude hay meadow	Y	10-20	
R3	***Seasonally wet and wet grasslands***			
R35	Moist or wet mesotrophic to eutrophic hay meadow	Y	15-25	
R36	Moist or wet mesotrophic to eutrophic pasture	N	NA	N
R37	Temperate and boreal moist or wet oligotrophic grassland	Y	10-20	
R4	***Alpine and subalpine grasslands***			
R41	Snow-bed vegetation	N	NA	N
R42	Boreal and arctic acidophilous alpine grassland	Y	5-10	
R43	Temperate acidophilous alpine grassland	Y	5-10	
R5	***Woodland fringes and clearings and tall forb stands***			
R52	Forest fringe of acidic nutrient-poor soils	N	NA	N
R54	*Pteridium aquilinum* vegetation	N	NA	N
R55	Lowland moist or wet tall-herb and fern fringe	N	NA	N
R56	Montane to subalpine moist or wet tall-herb and fern fringe	N	NA	N
S	**Heathland, scrub and tundra**			
S2	***Arctic, alpine and subalpine scrub***	**Y**	**5-10**	
S21	Subarctic and alpine dwarf *Salix* scrub	level 2	5-10	
S22	Alpine and subalpine ericoid heath	level 2	5-10	
S23	Alpine and subalpine *Juniperus* scrub	level 2	5-10	
S25	Subalpine and subarctic deciduous scrub	level 2	5-10	
S27	Krummholz with conifers other than *Pinus mugo*	level 2	5-10	
S3	***Temperate and mediterranean-montane scrub***			
S31	Lowland to montane temperate and submediterranean *Juniperus* scrub	Y	5-15	
S32	Temperate *Rubus* scrub	N	NA	N
S35	Temperate and submediterranean thorn scrub	N	NA	N
S37	*Corylus avellana* scrub	N	NA	Y (T1)
S38	Temperate forest clearing scrub	N	NA	N
S4	***Temperate heathland***			
S41	Wet heath	Y	5-15	
S42	Dry heath	Y	5-15	
S9	***Riverine and fen scrub***			

S92	*Salix* fen scrub	N	NA	
T	**Forest and other wooded land**			
T1	***Broadleaved deciduous forests***	***Y***	***10-15***	
T11	Temperate *Salix* and *Populus* riparian forest	level 2	10-15	
T12	*Alnus glutinosa-Alnus incana* forest on riparian and mineral soils	level 2	10-15	
T15	Broadleaved swamp forest on non-acid peat	level 2	10-15	
T16	Broadleaved mire forest on acid peat	level 2	10-15	
T18	*Fagus* forest on acid soils	Y	10-15	
T1B	Acidophilous *Quercus* forest	Y	10-15	
T1C	Temperate and boreal mountain *Betula* and *Populus tremula* forest on mineral soils	level 2	10-15	
T1E	*Carpinus* and *Quercus* mesic deciduous forest	Y	15-20	
T1H	Broadleaved deciduous plantations of non-site-native trees	level 2	10-15	
T3	***Coniferous forests***	***Y***	***3-15***	
T35	Temperate continental *Pinus sylvestris* forest	Y	5-15	
T3J	*Pinus* and *Larix* mire forest	level 2	3-15	
T3M	Coniferous plantation of non-site-native trees	level 2	3-15	
U	**Inland habitats with no or little soil and mostly with sparse vegetation**			
U2	***Screes***			
U22	Temperate high-mountain siliceous scree	N	NA	N
U26	Temperate high-mountain base-rich scree and moraine	N	NA	N
U3	***Inland cliffs, rock pavements and outcrops***			
U31	Boreal and arctic siliceous inland cliff	N	NA	N
U35	Boreal and arctic base-rich inland cliff	N	NA	N
U3D	Wet inland cliff	N	NA	N
U3E	Limestone pavement	N	NA	N
U5	***Miscellaneous inland habitats usually with very sparse or no vegetation***			
U51	Fjell field	N	NA	N

Table 2. *Ecosystem changes associated with critical load exceedance in Scottish terrestrial semi-natural habitats. Habitat types are indicated by level 2 or 3 EUNIS codes, see Table 1 for habitat names. Note that some habitats have not been studied at all and in others only a limited parameters have been examined. A 'Y' in the table indicates that this parameter has been observed to change in this habitat when the critical load has been exceeded. For some habitats, responses have only been investigated at the broader EUNIS level 2, indicated in bold type, or more than one level 3 habitat has been grouped together. Where this is the case, responses are shown in italic*.

	Soils											**Vegetation**									**Vascular plants**												**Bryophytes**							**Lichens**			
EUNIS 2022 habitat code	**1. Increased soil %N**	**2. Decreased soil C:N ratio**	**3. Increased N mineralisation**	**4. Increased N leaching**	**5. Increased peat water N**	**6. Increased N2O/NO emissions**	**7. Soil acidification**	**8. Decreased root biomass**	**9. Decreased mycorrhizal infection**	**10. Change in mycorrhiza community**	**11. Change in soil fauna**	**12. Increased biomass/productivity**	**13. Change in sp. composition**	**14. Decreased species richness/diversity**	**15. Decreased positive indicator sp.**	**16. Decreased oligotrophic sp.**	**17. Increased nitrophilous species**	**18. Increased Ellenberg N**	**19. Increased tissue N**	**20. Increased late successional sp.**	**21. Increased vascular cover/biomass**	**22. Increased graminoid cover/biomass**	**23. Increased tall vascular sp.**	**24. Increased mesophilic sp. cover**	**25. Increased grass:forb ratio**	**26. Decreased evergreen shrub cover**	**27. Increased Calluna growth increment**	**28. Decreased vascular plant C:N**	**29. Increased vascular plant tissue N**	**30. Decreased seed viability/seedbank**	**31. Increased abiotic stress sensitivity**	**32. Tree nutritional imbalance**	**33. Increased bryophyte biomass**	**34. Decreased bryophyte cover/biomass**	**35. Change in bryophyte sp. comp.**	**36. Decreased bryophyte sp. richness**	**37. Change in bryophyte growth rate**	**38. Increased bryophyte tissue N**	**39. Decreased bryophyte C:N**	**40. Decreased lichen cover/biomass**	**41. Decreased lichen sp. richness**	**42. Increased lichen tissue N**	**43. Decreased lichen C:N**
M																																											
MA1																																											
MA2																																											
MA22												Y			Y					Y		Y																					
N																																											
N1																																											
N11																																											
N13								Y				Y									Y	Y	Y																				
N15				Y			Y					Y	Y	Y		Y			Y	Y	Y	Y	Y	Y					Y				Y			Y		Y		Y	Y	Y	
N18				*Y*			*Y*					*Y*								*Y*	*Y*	*Y*					*Y*							*Y*						*Y*			*Y*
N19				*Y*			*Y*					*Y*								*Y*	*Y*	*Y*					*Y*							*Y*						*Y*			*Y*
N1A																																											
N1H												Y	Y	Y		Y		Y				Y	Y	Y																			
N2																																											
N21																																											
N23																																											

EUNIS	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**	**35**	**36**	**37**	**38**	**39**	**40**	**41**	**42**	**43**
N24																																											
N3																																											
N31																																											
N24																																											
Q																																											
Q1	**Y**		**Y**		**Y**	**Y**	**Y**						**Y**	**Y**	**Y**						**Y**	**Y**	**Y**											**Y**	**Y**	**Y**	**Y**	**Y**		**Y**	**Y**	**Y**	
Q11	*Y*		*Y*		*Y*	*Y*	*Y*						*Y*	*Y*	*Y*						*Y*	*Y*	*Y*											*Y*	*Y*	*Y*	*Y*	*Y*		*Y*	*Y*	*Y*	
Q12	*Y*		*Y*		*Y*	*Y*	*Y*						*Y*	*Y*	*Y*						*Y*	*Y*	*Y*											*Y*	*Y*	*Y*	*Y*	*Y*		*Y*	*Y*	*Y*	
Q2																					**Y**	**Y**												**Y**	**Y**		**Y**	**Y**					
Q21																					*Y*	*Y*												*Y*	*Y*		*Y*	*Y*					
Q22																					*Y*	*Y*												*Y*	*Y*		*Y*	*Y*					
Q24																					*Y*	*Y*												*Y*	*Y*		*Y*	*Y*					
Q25																					*Y*	*Y*												*Y*	*Y*		*Y*	*Y*					
Q4																																											
Q41													*Y*		*Y*				*Y*		*Y*	*Y*	*Y*											*Y*	*Y*		*Y*	*Y*					
Q42													*Y*		*Y*				*Y*		*Y*	*Y*	*Y*											*Y*	*Y*		*Y*	*Y*					
Q43													*Y*		*Y*				*Y*		*Y*	*Y*	*Y*											*Y*	*Y*		*Y*	*Y*					
Q44													*Y*		*Y*				*Y*		*Y*	*Y*	*Y*											*Y*	*Y*		*Y*	*Y*					
Q45																					Y													Y									
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R1A			Y	Y			Y			Y			Y	Y	Y			Y	Y			Y	Y		Y				Y						Y					Y	Y		
R1M			Y									Y	Y	Y	Y							Y			Y					Y				Y		Y		Y			Y		
R1P	Y		Y									Y		Y							Y	Y							Y					Y						Y	Y	Y	
R1S																																											
R2																																											
R21																																											

EUNIS	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43
R22														Y	Y							Y	Y																				
R3																																											
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S21		*Y*	*Y*	*Y*									*Y*	*Y*	*Y*	*Y*											*Y*		*Y*		*Y*			*Y*	*Y*					*Y*	*Y*	*Y*	
S22		*Y*	*Y*	*Y*									*Y*	*Y*	*Y*	*Y*											*Y*		*Y*		*Y*			*Y*	*Y*					*Y*	*Y*	*Y*	
S23		*Y*	*Y*	*Y*									*Y*	*Y*	*Y*	*Y*											*Y*		*Y*		*Y*			*Y*	*Y*					*Y*	*Y*	*Y*	
S25		*Y*	*Y*	*Y*									*Y*	*Y*	*Y*	*Y*											*Y*		*Y*		*Y*			*Y*	*Y*					*Y*	*Y*	*Y*	
S27		*Y*	*Y*	*Y*									*Y*	*Y*	*Y*	*Y*											*Y*		*Y*		*Y*			*Y*	*Y*					*Y*	*Y*	*Y*	
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S9																																											

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S92																																											
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T12		*Y*	*Y*	*Y*				*Y*		*Y*	*Y*	*Y*	*Y*			*Y*	*Y*												*Y*			*Y*								*Y*	*Y*		
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T16		*Y*	*Y*	*Y*				*Y*		*Y*	*Y*	*Y*	*Y*			*Y*	*Y*												*Y*			*Y*								*Y*	*Y*		
T18		Y	Y	Y				Y		Y	Y	Y	Y			Y	Y												Y		Y	Y								Y	Y		
T1B		Y	Y	Y				Y	Y	Y	Y	Y	Y			Y	Y												Y			Y		Y						Y	Y		
T1C		*Y*	*Y*	*Y*				*Y*		*Y*	*Y*	*Y*	*Y*			*Y*	*Y*												*Y*			*Y*								*Y*	*Y*		
T1E		Y	Y	Y				Y		Y	Y	Y	Y			Y	Y												Y			Y								Y	Y		
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T35	Y	Y	Y	Y		Y		Y		Y	Y		Y			Y	Y					Y				Y			Y		Y	Y		Y						Y	Y		
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T3M	*Y*	*Y*	*Y*	*Y*		*Y*		*Y*		*Y*			*Y*			*Y*	*Y*												*Y*		*Y*	*Y*								*Y*	*Y*		
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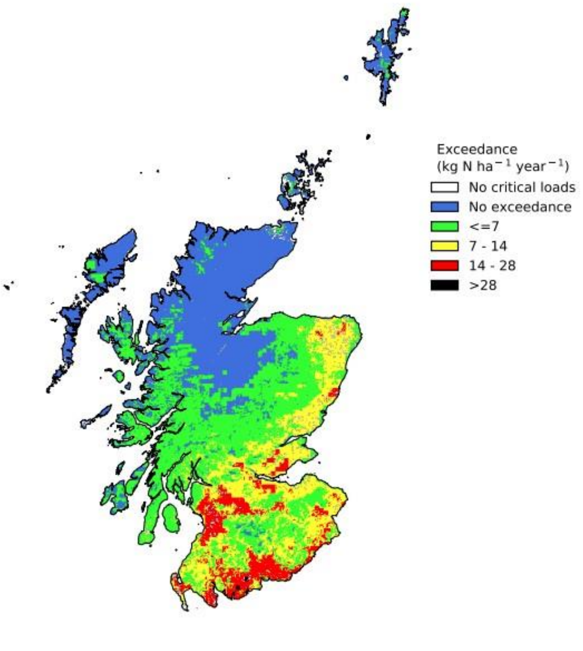
Section 2: What techniques could be used to mitigate nitrogen effects in Scottish terrestrial ecosystems and what is the evidence for their effectiveness under Scottish conditions?

Introduction

Semi-natural ecosystems in Scotland frequently experience nitrogen deposition in excess of critical loads. Although nitrogen deposition in Scotland is generally lower than in other parts of the UK, 45% of Scottish nitrogen-sensitive habitats received deposition in excess of their critical load in 2018-20 (Figure 1), with an average accumulated exceedance of 2.65 kg N ha^{-1} yr^{-1} above the critical load (Hina et al., 2022, Unpublished report to DEFRA). The recent review of empirical critical loads in Europe (Bobbink *et al.*, 2022) has resulted in a lowering of many critical load thresholds however, so the spatial extent of critical load exceedance and the amount of accumulated exceedance are set to rise. Such widespread exceedance of critical loads thresholds suggests that excess nitrogen is highly likely to be impacting Scottish semi-natural ecosystems. As outlined in Section 1, excess nitrogen deposition is known to impact on many aspects of ecosystem structure and function, from soil processes and water quality to productivity, phenology, species interactions, species diversity and community composition. Actions to control emissions of nitrogen from industry, transport, domestic sources, and agriculture have resulted in a slow decline in deposition of oxidized forms of nitrogen, but deposition of reduced nitrogen (such as ammonia) has a stable or slightly increasing trend (Rowe *et al.*, 2022). In Scotland, these declines in oxidized nitrogen deposition have resulted in a decrease in the area of habitat receiving deposition in excess of its critical load, of around 11% between 2009-11 and 2018-20. Since the magnitude of critical load exceedance across much of Scotland is not large (in a UK context) further action, particularly to control emissions of ammonia from agriculture, could result in significant declines in the area of semi-natural habitats which are exposed to excess nitrogen deposition.

Figure 1. *Average accumulated exceedance of nutrient nitrogen critical loads in Scotland 2018-2020. Figure supplied by N. Hina and E. Rowe, UKCEH.*



While there is much research into the impacts of elevated nitrogen deposition, the ability of ecosystems to recover from nitrogen impacts and our ability to improve or speed up the recovery process through habitat management activities have been much less researched. Evidence on the natural recovery of ecosystems from nitrogen impacts was reviewed by Stevens (2016) while evidence on the potential for nitrogen mitigation though habitat management was reviewed by Jones *et al.* (2017) for non-forest habitats and by Clark *et al.* (2019) for (primarily north American) forest habitats. In this section of the report, we review new information arising since these reviews which is relevant to Scottish semi-natural habitats and examine the potential for nitrogen impacts mitigation through habitat management in Scotland.

Methods

We conducted a literature review of the available evidence on ecosystem recovery from nitrogen deposition impacts and potential mitigation methods in Scottish natural and semi-natural terrestrial ecosystems, using the PICO methodology (Collaboration for Environmental Evidence, 2013) to refine search terms and to define the relevant ***P***opulation, ***I***ntervention, ***C***omparator and ***O***utcomes. Our primary question for the review was:

What is the timescale for recovery of biodiversity and ecosystem functioning in Scottish semi-natural ecosystems from the impacts of nitrogen deposition, and how effective is habitat management in terms of mitigating nitrogen impacts and improving recovery?

- ***Population*****:** Terrestrial natural and semi-natural habitats present in Scotland, or close analogues from cool temperate and boreal climate zones in Europe, North America, or Asia.
- ***Intervention***: Management methods to reduce adverse impacts associated with nitrogen deposition on biodiversity and/or ecosystem function or monitoring of ‘natural’ recovery without intervention.
- ***Comparator***: Control plots or reference sites without mitigation treatment, or before-after comparisons.
- ***Outcomes:*** Measures of biodiversity, or key ecosystem functions and properties relating to carbon or nutrient cycling and storage, soil pH or leachate chemistry.

Search strategy

Web of Science was searched for relevant documents in English with the date range “all years” using the search string below, and the results were assembled into an Endnote database. The search string was iteratively tested and developed in Web of Science to refine the search results and improve relevance to the primary question (see Appendix 2).

Population:
(tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*)

AND

Intervention:
(“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nutrient enrichment” OR “nitrogen fertili*”) AND (mitigation OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR “biomass removal” OR “turf stripping” OR “topsoil removal” OR “sod cutting” OR “turf cutting” OR “soil amendment” OR “nutrient removal” OR “carbo* addition” OR “soil disturbance” OR liming OR “canopy closure” OR thinning)

AND

Outcome:
(biodiversity OR diversity OR richness OR assemblage* OR "functional type" OR "functional group" OR "growth form" OR "species number" OR "species composition" OR "number of species" OR "floristic composition" OR "community composition" OR "habitat suitability" OR "ecosystem function" OR "decomposition" OR "carbon stock*" OR "carbon storage" OR "carbon cycl*" OR "nitrogen cycl*" OR "nutrient stock*" OR "nitrogen stock*" OR "nitrogen budget" OR "nitrogen pool*" OR acid* OR leach* OR producti*)

An additional search string was added to exclude studies from non-target ecosystems:
NOT (agricultur* OR urban OR river OR stream OR lake OR pond)

Complete search string:
TS=(((tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND ("nitrogen deposition" OR "nitrogen addition" OR "nitrogen pollution" OR "nitrogen enrichment" OR "nitrogen fertili*") AND (mitigate* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR "biomass removal" OR "turf stripping" OR "topsoil removal" OR "sod cutting" OR "turf cutting" OR "soil amendment" OR "nutrient removal" OR "carbo* addition" OR "soil disturbance" OR liming OR "canopy closure" OR thinning) AND (biodiversity OR diversity OR richness OR assemblage* OR "functional type" OR "functional group" OR "growth form" OR "species number" OR "species composition" OR "number of species" OR "floristic composition" OR "community composition" OR "habitat suitability" OR "ecosystem function" OR "decomposition" OR "carbon stock*" OR "carbon storage" OR "carbon cycl*" OR "nitrogen cycl*" OR "nutrient stock*" OR "nitrogen stock*" OR "nitrogen budget" OR "nitrogen pool*" OR acid* OR leach* OR producti*)) NOT (agricultur* OR urban OR river OR stream OR lake OR pond))

This search string returned 1593 documents on 9th September 2022 and provides a conservative (inclusive) starting point for the review.

Study selection criteria

First sift

Bibliographic details of each of the 1593 documents were downloaded into an Endnote database and a first sift was made based on the information contained in the titles and abstracts. Documents were retained if they referred to the following:

1. Natural or semi-natural terrestrial ecosystems present in Scotland or their close analogues from cool temperate or boreal zones.

AND

2. Recovery from, or mitigation of, nitrogen deposition or addition impacts.

AND

3. Some aspect of biodiversity or ecosystem properties/function.

Both field and laboratory experimental studies were retained, as well as survey-based studies, but we excluded studies based purely on modelling. Meta-analyses using spatial gradients to infer responses to management and nitrogen deposition were also retained. Reviews were retained for information but were kept separate from primary data. Studies had to be relevant to Scottish semi-natural ecosystems and clearly focused on assessing recovery from nitrogen deposition or addition. Studies in intensively managed agricultural and forestry habitats were excluded (e.g., intensive grassland, arable) as were studies from urban or non-terrestrial habitats. Only studies in English were included. If there was any doubt, papers were retained at this stage of the sift.

All of the documents were screened by a single reviewer to ensure consistency of inclusion/exclusion. To give a degree of quality assurance in terms of fit to the defined review criteria, a second reviewer then screened 10% (160) of the documents. The second screening indicated >99% agreement between reviewers. In total, 206 documents were retained after the first sift.

Second sift

Full texts were downloaded for the 206 documents retained from the first sift and these were then assessed in more detail to ensure that they met the review PICO criteria:

Population: Studies on terrestrial natural and semi-natural habitats present in Scotland, or their close analogues from cool temperate and boreal climate zones in Europe, North America, or Asia. We did not include laboratory studies which did not clearly relate to a defined habitat.

Intervention: Studies had to either (a) assess the suitability of one or more management methods to reduce adverse impacts associated with nitrogen deposition on biodiversity and/or ecosystem function or (b) assess the rate or extent of recovery from nitrogen deposition without intervention.

Comparator: Experimental studies had to include suitable control plots or reference sites without mitigation treatment. Studies of recovery without intervention could include before-after comparisons, time-series data or comparisons to a reference site.

Outcomes*:* Studies had to assess treatment effects on, or recovery of, some measure of biodiversity (any taxon group) such as species richness, community composition or functional group richness/composition, or key ecosystem functions relating to carbon or nutrient cycling and storage (e.g., nitrogen leaching, decomposition, nitrogen mineralisation) or soil/leachate acidity.

After the second sift, 107 documents were retained for inclusion in the review. These documents were sorted into categories according to the vegetation type to which they referred, using the European Nature Information System (EUNIS) classification scheme 2021/22 (https://eunis.eea.europa.eu/habitats.jsp). Review papers were retained but kept separately, to avoid inflating the apparent amount of primary research. The 107 papers were then read in detail and their reference lists were checked for any additional relevant papers. This step produced an additional 23 papers for review.

Summary of evidence

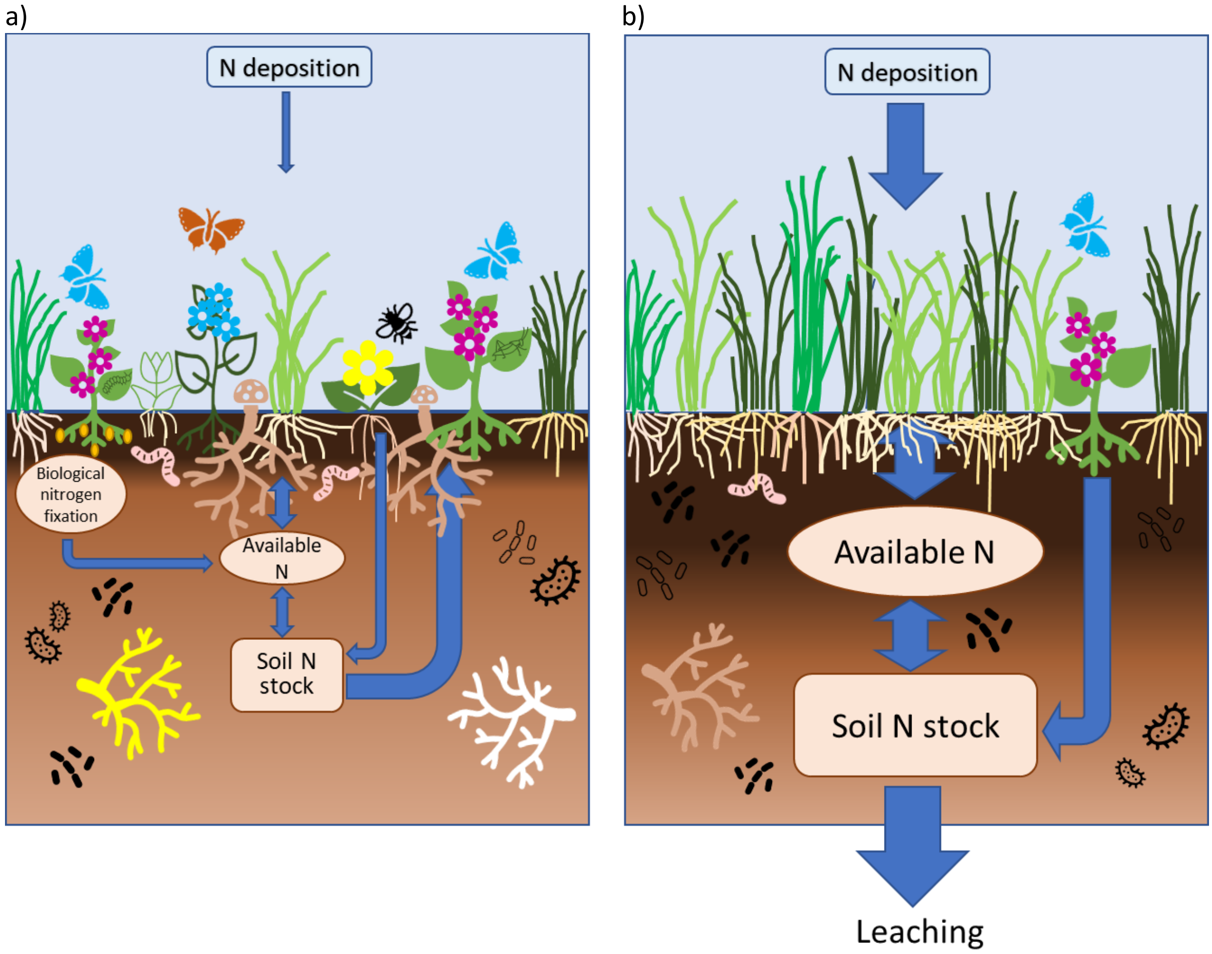
The 130 documents retained after the second sift and checking of reference lists were read in detail and the evidence on rates of natural recovery from nitrogen impacts and the influence of potential mitigation techniques was summarized by EUNIS habitat category, for those habitats where evidence was found. Following the literature review, we created a table of Scottish natural and semi-natural habitats by EUNIS category and identified which habitats are typically managed, what forms that management may take and therefore which habitats have scope for nitrogen mitigation through habitat management and which habitats are reliant on nitrogen emissions reduction and natural recovery.

Recovery and mitigation in nitrogen-impacted ecosystems

Nitrogen deposition impacts upon structure and function in a broadly similar way across a range of ecosystems (Figure 2). Under 'pristine' conditions nitrogen inputs from the atmosphere are small (1-3 kg N ha^{-1} yr^{-1}) and nitrogen is commonly a limiting nutrient for plant growth, particularly in the acidic and nutrient poor soils prevalent in Scotland. Under these conditions, biological nitrogen fixation is an advantageous trait and root-associated bacterial symbionts, nitrogen-fixing lichens and moss associations with cyanobacteria contribute significantly to ecosystem nitrogen inputs. Symbioses with mycorrhizal fungi are critical for plants in the uptake of nitrogen, phosphorus and other nutrients from the soil. Anthropogenic nitrogen emissions increase nitrogen inputs in rainfall and concentrations in the air and this additional nitrogen is readily taken up by plants and soil organisms which tends to increase growth. Plant canopies may become taller and/or denser, increasing competition for light and space between species and having negative impacts on diversity. Nitrogen deposition can also alter the ratios of carbon and nutrients such as nitrogen, phosphorus and potassium in plant tissues which has consequences both for plant physiology and consumer organisms which feed on plant material. Over time, nitrogen stocks accumulate in increased above ground biomass and within soil organic horizons as nitrogen-enriched dead plant material enters the soil. Increased nitrogen inputs to the soil may also result in increased activity of nitrogen cycling micro-organisms and further increased nitrogen availability for plants. Nitrogen fixing capabilities and root symbioses with fungi can become less advantageous, which impacts on both above and below ground biodiversity. Where the rate of nitrogen input exceeds the rate at which organisms can utilize it *in situ*, nitrogen leaching occurs, and nitrogen is exported to downstream ecosystems and ground and surface waters.

When anthropogenic nitrogen inputs decline, some aspects of nitrogen impacted ecosystems may recover naturally. Leaching of nitrogen should decline when the input of nitrogen no longer exceeds biological uptake, and organisms such as bryophytes and lichens, which principally depend on nutrient inputs in rainfall and are isolated from the soil may show recovery. However, accumulated nitrogen pools in the soil will remain and reduction of these accumulated pools, through loss of nitrogen back to the atmosphere, by transport of solid materials or by leaching may be very slow. Continued high nitrogen availability in the soil may maintain plant communities in a highly productive but low diversity state. Some habitat management activities such as grazing, burning, and turf removal may have potential to mitigate the impacts of nitrogen deposition by increasing export of nitrogen from the ecosystem in biomass or soil, or by altering plant community sward structure to reduce competition. Addition of nutrients, lime or carbon may also be used to mitigate nitrogen effects on soil acidity, nitrogen availability and nutrient imbalances. In the following section we review the evidence on the extent and rate of natural ecosystem recovery from nitrogen deposition and the effectiveness of management techniques to mitigate nitrogen impacts in Scottish semi-natural ecosystems.

Figure 2. *Typical changes in biodiversity and nitrogen stocks and flows associated with nitrogen deposition. In a pristine ecosystem (a) nitrogen inputs and nitrogen stocks are small and nitrogen availability is low – biological nitrogen fixation and mycorrhizal associations enable plants to access nitrogen for growth, while above ground biomass is low and biodiversity is high. In a nitrogen impacted ecosystem (b) nitrogen inputs and availability are high, the soil nitrogen stock is increased, and excess nitrogen is lost though leaching. Aboveground biomass is increased as fast-growing nitrogen-demanding species predominate and biodiversity is reduced.*



Evidence for ecosystem recovery potential and the mitigating effect of management in Scottish habitats

Coastal dunes (EUNIS N1)

Natural recovery

Critical loads are set at 10-20 kg N ha^{-1} yr^{-1} for shifting dunes and dune slack pools, 5-15 kg N ha^{-1} yr^{-1} for grey dunes and dune slacks, and 10-15 kg N ha^{-1}- yr^{-1} for dune heaths (Bobbink *et al.*, 2022). Nitrogen deposition has led to a reduction in diversity in many dune systems and a shift to more eutrophic vegetation. As yet, there is no evidence for natural recovery from nitrogen deposition in Scottish dune systems or similar systems in Western Europe. This is a consequence of the lack of long-term change data; where long-term data does exist (Pakeman *et al.*, 2016) there have not been sufficient sampling dates since nitrogen deposition peaked in the 1990s to allow assessment of recovery.

Mitigation

Grazing. There are studies from Dutch dune systems that demonstrate that continued grazing has enabled the survival of high levels of plant diversity in dune grasslands under high nitrogen deposition (tenHarkel & vanderMeulen, 1996; Kooijman *et al.*, 2017). However, where grazing has formed part of an investigation into mitigation, the results have been mixed. In a three-year experiment, Brunbjerg *et al.* (2014) showed grazing had no beneficial effects on diversity, as did Ford *et al.* (2016) in a six-year experiment. Other experiments showed a partial success at reducing the impacts of nitrogen deposition (Plassmann *et al.*, 2009; Plassmann *et al.*, 2010). Grazing acted by removing above-ground biomass, allowing light deeper into the canopy and reducing the dominance of tall grasses. If used for nitrogen mitigation, grazing systems must take into account the potential for nitrogen inputs through supplementary feeding to avoid overall eutrophication (van Dobben *et al.*, 2014). There is little information on the effects of nitrogen deposition on dune nutrient pools, but what there is suggests that grazing either has a limited capability of reducing total nitrogen pools (Jones *et al.*, 2017) or can only do this if animals are folded (removed) at night (Van den Berg *et al.*, 2014).

Disturbance. Only one experimental study included disturbances beyond grazing (Brunbjerg *et al.*, 2014). In that study disturbance (cutting and blowout - disturbance of vegetation to remobilise sand) increased species richness and decreased biomass, with cutting increasing forb and bryophyte biomass at the expense of grasses. Trampling reduced the cover of lichens. Large scale vegetation removal at Kenfig (South Wales) aimed at improving habitat quality for fen orchid has successfully expanded its population (Clark, 2019).

Summary – Coastal dunes

Although there are relatively few nitrogen mitigation studies from coastal dune systems, there is some evidence that grazing can partially mitigate the impacts of nitrogen deposition. However, experiments have been short-term and often with high nitrogen inputs making it harder for beneficial grazing impacts to be seen. There is very little published evidence about other disturbance options such as creating blowouts and no evidence on the potential of fire as a mitigation option. In the absence of management, it is unclear what the likely timescale would be for natural recovery of dune ecosystems from nitrogen deposition, or if changes towards less diverse later successional communities would be permanent.

Wetlands (EUNIS Q)

Natural recovery

Critical loads are set at 5-10 kg N ha^{-1} yr^{-1} for oligotrophic raised and blanket bogs (Q1), 5-15 kg N ha^{-1} yr^{-1} for valley mire, poor fen and transition mires (Q2), and 15-25 kg N ha^{-1} yr^{-1} for rich fens (Q4). However, no critical load has been set for helophyte bed habitats (swamps, Q5). Nitrogen deposition to wetland habitats including mires, bogs and fens results in increased growth of vascular plants and negative impacts on bryophytes with changes in growth and species composition (Bobbink *et al.*, 2022). Nitrogen concentrations in peat and peat water may also increase. Evidence for recovery from nitrogen deposition is only available for bogs and fens in Europe. A study of *Sphagnum magellanicum* dominated bogs in central Europe (Novak *et al.*, 2018) suggested that a previously nitrogen polluted bog (deposition of 40 kg N ha^{-1} yr^{-1}) was able to recover from nitrogen saturation over a period of 20-30 years during which time nitrogen deposition declined to 12-15 kg N ha^{-1} yr^{-1}. Although nitrogen deposition remained above the critical load, the *Sphagnum* layer regained its ability to retain and 'filter' incoming nitrogen, preventing it from reaching other ecosystem compartments. In an experimental study of a Dutch poor fen and eutrophic fen dominated by *S. magellanicum* and *S. fallax* respectively (Limpens & Heijmans, 2008), nitrogen content of *Sphagnum* shoots returned to control levels within 15 months of cessation of nitrogen additions (40 kg N ha^{-1} yr^{-1}). Background rates at these study sites were significantly above critical loads however, and additional nitrogen might have been more tightly retained by the *Sphagnum* if background levels were lower (Limpens & Heijmans, 2008). Rapid recovery of bog habitats is also only possible when the *Sphagnum* layer remains intact, once the plant community has changed, recovery may be much slower due to competition between mosses and vascular plants. In an Italian alpine transition mire where 10 and 30 kg N ha^{-1} yr^{-1} had been added for eight years and cover of *Calluna vulgaris*, *Molinia caerulea* and *Polytrichum strictum* was enhanced by nitrogen while *Sphagnum fuscum* was reduced, there was little sign of recovery after 3 years without nitrogen additions (Gerdol & Brancaleoni, 2015).

Mitigation

Only two studies have investigated the influence of management activities on the response of wetlands to nitrogen deposition and evidence is primarily confined to effects on vegetation biodiversity. A long term resurvey study of oligotrophic *Carex* dominated fens and wet meadows dominated by *Caltha*, *Molinia* and *Filipendula* in the Jura mountains found that over a 38 year period, vegetation became less species rich and more dominated by nutrient demanding species (Rion *et al.*, 2018). The changes were greatest at sites where grazing had been removed 25 years earlier, while those sites where grazing or mowing management had been continued appeared to have maintained their community composition in the face of ongoing eutrophication. In the Netherlands, the effectiveness of mowing, burning and liming was tested for restoration of rich fen habitat degraded by eutrophication, acidification and succession (van Diggelen *et al.*, 2015). Summer mowing shifted the degraded rich fen vegetation towards an acidic poor fen community dominated by *Sphagnum*, while burning after winter mowing caused an undesirable shift towards species associated with nutrient-rich conditions, common species and tall herbs. Liming had negative effects on *Sphagnum*, but limited effects on vascular plants, and rich fen species did not re-establish, possibly due to dispersal constraints.

Summary – Wetlands

Evidence for natural recovery and the effectiveness of management for nitrogen mitigation in wetlands is limited and confined to a small number of wetland types. When nitrogen deposition declines, natural recovery of *Sphagnum* dominated mires and bogs appears possible over annual to decadal timescales, provided that the moss layer remains intact. Where significant changes in vegetation species composition have occurred, it is unclear to what extent this can be reversed without intervention. However, management options for bogs and mires are limited and there is no evidence on their effectiveness at reducing nitrogen impacts. Management interventions have been tested in fen communities, but while there was some evidence that maintaining grazing or cutting could prevent adverse effects of ongoing nitrogen deposition, attempts at restoration of nitrogen impacted fen vegetation by various management techniques were generally not effective. There was no evidence on the timescale for natural recovery of fen vegetation. Nitrogen deposition reduction should be the clear priority for protection of wetland habitats, but more studies are needed in a wider range of wetland types typical of Scottish habitats, to explore how those communities already impacted could be restored.

Grasslands and lands dominated by forbs, mosses or lichens (EUNIS R)

Lowland to subalpine dry, mesic and wet grasslands (EUNIS R1, R2, R3)

Natural recovery

In grasslands, nitrogen deposition has led to increases in productivity and usually to reductions in plant species richness and diversity and critical loads range from 5-15 kg N ha^{-1} yr^{-1} for dry grasslands on sand (R1P) to 15-25 kg N ha^{-1} yr^{-1} for mesotrophic hay meadows (R35) (Bobbink *et al.*, 2022). Evidence of recovery from long-term survey data is limited as there are insufficient survey dates surveys since the peak of nitrogen deposition in the 1990s to detect change (Mitchell *et al.*, 2018). The long-term Park Grass grassland management experiment running at Rothamsted since the 1850s does however, provide evidence for recovery (increases) in plant diversity and legume cover fifteen years after peak nitrogen deposition (Storkey *et al.*, 2015). Recovery of plant diversity has also been seen after 20 years in a Dutch hay meadow (Berendse *et al.*, 2021) and in Swiss meadows receiving low levels of deposition (below 10 kg N $ha^{-1}yr^{-1}$, Kammer *et al.* (2022)). There is also evidence of recovery in experimental systems where inputs have ceased. Where measured, it appears that soil nitrogen availability for plants rapidly reduces after the cessation of treatment in acid grasslands (O'Sullivan *et al.*, 2011), neutral grasslands (Stevens *et al.*, 2012a), calcareous grasslands (O'Sullivan *et al.*, 2011) and prairie (Clark *et al.*, 2009; Nieland *et al.*, 2021). However, total soil nitrogen did not reduce in neutral grasslands (Stevens *et al.*, 2012a) or temperate steppe (Hu *et al.*, 2020b). Reduction to control levels was also seen in plant tissue nitrogen of mosses in acid grassland (Arroniz-Crespo *et al.*, 2008) and of vascular plants in neutral grassland (Stevens *et al.*, 2012a; Storkey *et al.*, 2015). Recovery in diversity was usually only partial, with plant diversity still reduced compared to controls after 20 years of recovery in prairie (Isbell *et al.*, 2013) and little recovery in neutral grassland (Stevens *et al.*, 2012a). Seed bank diversity did not show any signs of recovery four years after treatment cessation on an acidic grassland (Basto *et al.*, 2015).

Mitigation

Grazing. Surprisingly, there is an absence of studies looking at the interactions between grazing and nitrogen deposition to investigate the potential for mitigation. There are however, a wide range of studies that have shown that the removal of grazing can exacerbate the impacts of nitrogen deposition, primarily through the build-up of biomass and a shift towards species of more eutrophic habitats (Pakeman *et al.*, 2017; Hu *et al.*, 2020a; Chen *et al.*, 2021; Li, YB *et al.*, 2021; Lu *et al.*, 2021) with one study suggesting that grazing reductions to restore habitats such as heathlands from grasslands should be implemented slowly to allow soil processes to keep pace with above ground vegetation change (McGovern *et al.*, 2014).

Mowing (including litter removal). Mowing experiments could be divided roughly into two groups: those in combination with ambient nitrogen deposition and those alongside high experimental inputs (usually 100 kg N ha^{-1} yr^{-1}). Under the former situation, mowing and removal was enough to counter the effects of nitrogen deposition in Swiss hay meadows (Kammer *et al.*, 2022) and tall grass prairie (Collins *et al.*, 1998) as was twice yearly mowing in the Park Grass experiment's hay meadows (Storkey *et al.*, 2015). It appears that much of the positive effect of mowing on diversity is a result of the enhanced light penetration into the sward (Ilmarinen *et al.*, 2009) and that mowing at least twice a year with removal is critical to counter the effects of continuing deposition (Wamelink *et al.*, 2009). However, mowing and removal was not enough to counter the effects of artificially high levels of nitrogen deposition on groups such as plants, nematodes and arbuscular mycorrhizal fungi in steppe grasslands and meadows (Hou *et al.*, 2017; Yang *et al.*, 2019; Hu *et al.*, 2020b; Miao *et al.*, 2020; Li, YB *et al.*, 2021; Li, ZM *et al.*, 2021; Lu *et al.*, 2021; Qin *et al.*, 2022; Zhang *et al.*, 2022).

Liming. In a Dutch hay meadow liming increased the rate of recovery from nitrogen fertilisation (Berendse *et al.*, 2021) as it also did in the Park Grass experiment (Storkey *et al.*, 2015). Where soil has been acidified, liming can be an effective mitigation option

Burning. In an experiment on prairie vegetation, burning increased biomass and decreased species richness (Collins *et al.*, 1998) but a second prairie experiment (Nieland *et al.*, 2021) showed that annual burning could slowly remove nitrogen from the system through losses in smoke and ash. This mitigation method is untried in Scotland and may not have any impact except on grassland types where large amounts of litter build up, i.e., those dominated by grasses such as *Molinia caerulea*, and there may be resistance to its use.

Alpine moss-heaths (Boreal and arctic acidophilous alpine grassland EUNIS R42)

Natural recovery

Scottish studies show that nitrogen deposition to *Racomitrium* dominated alpine moss-sedge heath results in thinning and reduction in cover of moss carpets and transition to graminoid dominance, accompanied by soil acidification and leaching of nitrogen into soil water, and the critical load is set at 5-10 kg N ha^{-1} y^{-1} (Bobbink *et al.*, 2022). Field and laboratory transplant studies have been used to investigate timescales for recovery, but there is no survey-based evidence for recovery from nitrogen deposition. Transplantation of moss turfs from high to low nitrogen deposition sites within the UK (Armitage *et al.*, 2011) showed that moss depth and cover started to increase within two years of transplantation, while graminoid cover declined, but that moss tissue nitrogen remained elevated over this timescale. A laboratory based study, using soil/vegetation monoliths from a sites with a range of deposition histories, also showed that nitrogen leaching rapidly decreased and soil water pH increased within a few weeks of excess nitrogen additions being reduced (Britton *et al.*, 2019). Carbon and nitrogen stocks in vegetation and soil were unaffected by nitrogen input reductions over these short timescales, and vegetation and soils which had previously been exposed to high nitrogen additions rapidly leached large amounts of nitrogen if nitrogen inputs were increased, suggesting that they remained nitrogen saturated.

Mitigation

Since moss-dominated alpine habitats are not usually actively managed, except by low intensity grazing of domestic or wild herbivores, options for mitigation of nitrogen impacts are limited. One study has assessed the potential of phosphorus addition and grazing exclusion as techniques to promote recovery of degraded *Racomitrium* heath, in a short-term, two-year trial at 10 sites across the UK (Armitage *et al.*, 2012). Growth and depth of *Racomitrium* was increased by both grazing exclusion and phosphorus addition, but grazing exclusion also increased grass cover which could cause negative shading impacts on the mosses in the longer term. Phosphorus addition appeared to have the most positive effects, but it is unknown how long these might last or whether there could be issues around leaching of applied phosphorus resulting in negative impacts on surface waters downstream.

Alpine grasslands (Temperate acidophilous alpine grassland EUNIS R43)

Natural recovery

The critical load for alpine grasslands (EUNIS R43 & R44) is set at 5-10 kg N ha^{-1} y^{-1}. Studies in the European Alps have shown that addition of nitrogen to both acidic and calcareous alpine grassland results in increased plant growth rates and above ground biomass (Bobbink *et al.*, 2022). Shifts in species composition also occur, with *Carex* species becoming more dominant. There are no studies from the UK or Europe which have specifically addressed recovery from excess nitrogen deposition or nitrogen addition in alpine grasslands. However, a study in an alpine dry meadow at Niwot ridge in the USA (Bowman *et al.*, 2018) where nitrogen additions were ceased after 12 years suggests that recovery may be slow. Nine years after nitrogen additions ceased, cover of *Carex* showed evidence of a return towards control levels, but plant productivity, nitrogen cycling rates and available nitrogen pools in the soil remained elevated. Abundance of bacteria and fungi in nitrogen treated soils remained low, with no evidence of recovery. Similar to other grassland types (see above), timescales for recovery of alpine grasslands from elevated nitrogen inputs seem likely to be of the order of decades.

Mitigation

While there have been no studies of methods for nitrogen deposition mitigation in alpine grasslands in the UK or Europe, three recent studies have examined the interactive effects of grazing or clipping and nitrogen addition in dry alpine meadows on the Tibetan plateau. Moderately large additions of nitrogen (40-75 kg N ha^{-1} y^{-1}) increased above ground plant biomass and soil nitrogen and decreased plant species richness, but these changes were counteracted by grazing or clipping of the vegetation (Song *et al.*, 2020; Zong & Shi, 2020; Li, L *et al.*, 2021). However, while grazing was able to maintain species richness under elevated nitrogen deposition, the identity of species in the grazed and fertilised communities was not the same as under control conditions, and the species present did not have the same functional characteristics, with for example, leguminous species being absent (Song *et al.*, 2020).

Summary – Grasslands

In lowland dry, mesic and wet grasslands, natural recovery of some aspects of soil conditions appears possible over short timescales (months to years), while partial recovery of plant diversity may take decades. Similarly in alpine grasslands, the limited evidence available suggests recovery of plant diversity may take decades, with nitrogen tightly retained within the soil. In alpine moss heaths, some aspects of plant and soil chemistry and moss growth are able to recover rapidly when nitrogen deposition declines, but accumulated nitrogen remains within the ecosystem and may constrain complete recovery in the longer term. A variety of management methods which open up the sward to allow light to penetrate, remove biomass (and therefore nitrogen) or directly counteract impacts such as soil acidification have been tested in grasslands. Grazing and mowing may have some potential to mitigate nitrogen impacts in grasslands, but reduction of nitrogen stocks within the ecosystem will only be achieved if cut material is removed or grazing animals are folded (taken off the land at night). It will be important to consider both species richness and species identity when monitoring recovery from nitrogen impacts as alpine grassland studies show that even when an open sward structure and plant species diversity is maintained by grazing or cutting under high nitrogen inputs, some important plant functional groups may be absent from the sward. In some grassland types, burning may also be effective at reducing nitrogen stocks and opening up the sward, but there is limited evidence for this, and it would only be appropriate is a small number of grassland types. Liming and phosphorus additions also have potential to mitigate some of the chemical impacts of nitrogen deposition. More studies of management effects on biodiversity and ecosystem processes in grasslands under realistic levels of nitrogen deposition are needed, to understand the potential for mitigation of nitrogen impacts under real world conditions.

Heathland, scrub and tundra habitats (EUNIS S)

Arctic, alpine and subalpine scrub (EUNIS S2)

Natural recovery

The critical load for alpine dwarf-shrub heaths is set at 5-10 kg N ha^{-1} y^{-1}.Studies from Scotland and Europe show that excess nitrogen deposition or nitrogen addition to alpine dwarf-shrub heaths results in reduced species richness, particularly of bryophytes and lichens (Bobbink *et al.*, 2022). In addition, growth of some vascular plant species is increased and there is acidification of soils and leaching of nitrogen into soil water. There is very limited information on rates of recovery for these habitats. In Svalbard, tundra heaths dominated by *Dryas octopetala* or *Cassiope tetragona*, vascular plants and lichens responded negatively to nitrogen addition, but 13-18 years after cessation of nitrogen additions there were some signs of recovery (Street *et al.*, 2015). The ecosystem was no longer nitrogen saturated, and some aspects of species composition had recovered, but tissue nitrogen in mosses remained elevated, and added nitrogen appeared to be efficiently recycled and strongly retained within the ecosystem.

Mitigation

Two Scottish studies have assessed the interactions between burning and clipping and nitrogen addition in alpine *Calluna-Cladonia* heath (Britton & Fisher, 2007; Britton & Fisher, 2008). Nitrogen addition reduced species richness, primarily through loss of lichens, and increased growth of *Calluna*. Clipping did not mitigate the effect of nitrogen addition on species richness but did counteract the positive effect of nitrogen on *Calluna* growth. Burning greatly reduced species richness and also reduced *Calluna* growth immediately after the fire, but there was no evidence that it mitigated the effects of nitrogen addition over the five years of the study. Longer-term studies are needed to assess the full effects of these management practices.

Temperate heathland (EUNIS S4)

Natural recovery

The critical loads for wet and dry heathlands are set at 5-15 kg N ha^{-1} y^{-1}. Nitrogen deposition to lowland and upland ericaceous dwarf-shrub heathlands results in increased *Calluna* growth in the short-medium term, but also increased sensitivity to biotic and abiotic stresses and a longer-term shift from dwarf-shrub to graminoid dominance (Bobbink *et al.*, 2022). Cover of lichens and bryophytes declines, and nitrogen leaching to soil water may occur. There is limited evidence of the rate of recovery of dwarf-shrub heathland from nitrogen deposition, with no survey-based studies addressing recovery at the landscape scale, but three studies from the UK which have assessed recovery following cessation of experimental nitrogen additions. In a dry lowland heathland which had received low additions of ammonium sulphate (up to 15.4 kg N ha^{-1} yr^{-1}) for 7 years, Power *et al.* (2006) found that while soil pH recovered quickly (within 1-2 years), nitrogen effects on *Calluna* growth, canopy morphology and phenology and vegetation composition persisted for at least 8 years after treatments ceased. Soil nitrogen concentrations and activity of the soil microbial community also remained elevated, despite all plots having been subject to burning or cutting management during recovery. In comparison, in a Welsh upland *Calluna* heath Edmondson *et al.* (2013) found that there was a general trend of reduced *Calluna* shoot growth after 2 years of recovery, but this was only significant when very high rates of nitrogen (120 kg N ha^{-1} yr^{-1}) had been applied previously. There was no significant change in soil or litter chemistry after 2 years, but after 7 years some declines in litter nitrogen content were seen. No recovery of plant community composition was seen within 7 years. Similarly, in an upland *Calluna*-grass heath in Scotland, only limited recovery of plant community composition was seen 10 years after cessation of nitrogen addition and there was evidence that added nitrogen was retained in the soil (van Paassen *et al.*, 2020). These studies suggest that while some soil parameters (such as pH) may be responsive to declining nitrogen deposition, recovery of other ecosystem properties is likely to be slow.

Mitigation

Heathlands are one of the best studied ecosystems in terms of potential methods for mitigation of nitrogen deposition impacts. A variety of different management methods are traditionally used on heathlands in different geographical areas and there are many studies investigating the potential of these to mitigate nitrogen impacts. These studies have focused on a small number of sites in the Netherlands, northern Germany, southern England, and north Wales, with the majority addressing management of lowland dry heathlands rather than the upland wet heathlands which are prevalent in Scotland.

Burning. Burning can take the form of low intensity prescribed burns carried out during winter and affecting mainly the dwarf-shrub canopy, or high intensity accidental burns or wildfires which usually occur during the summer period and can affect the litter layer and upper soil horizons in addition to the vegetation. In lowland dry heathland in northern Germany, management burns were found to remove 90% of the vegetation nitrogen stock in a 10-year-old stand and 53% in a 15-year-old stand, equivalent to around 5 years of nitrogen inputs from deposition (Niemeyer *et al.*, 2005). In a southern English heath, nitrogen removal by management burning was estimated to be around 15% of total ecosystem (vegetation and soil) nitrogen stocks (Barker *et al.*, 2004). High intensity summer wildfires removed a much greater proportion of nitrogen stocks – around 82% - but the effects of previous nitrogen additions were still apparent after burning (Barker *et al.*, 2004; Green *et al.*, 2013). In addition to nitrogen losses by combustion of biomass, increased leaching of nitrogen has been observed in the years following a fire, possibly due to decreased nitrogen uptake by vegetation and warming of exposed dark soil surfaces causing an increase in mineralisation rate (Niemeyer *et al.*, 2005; Hardtle *et al.*, 2007a; Green *et al.*, 2013). Calculation of detailed nitrogen budgets suggests that burning on a typical 10 year cycle would only partly mitigate the impacts of nitrogen deposition on ecosystem nitrogen stocks and would not prevent accumulation of nitrogen in plants and soils (Barker *et al.*, 2004; Niemeyer *et al.*, 2005; Hardtle *et al.*, 2006). Burning of biomass will also cause emissions of nitrogen which will be deposited on other ecosystems.

In addition to its effects on nitrogen pools, burning also removes phosphorus. Most phosphorus is returned to the ecosystem in the form of ash, and so burning has less effect on nutrient balance than some other managements (Hardtle *et al.*, 2009), but measurements at some sites suggest that increased phosphorus limitation can occur after burning (Green *et al.*, 2013). Similar results have also been seen in lowland wet heath (Hardtle *et al.*, 2007a).

Aside from its effects on nutrient pools, burning in lowland heathland also mitigated the positive effects of nitrogen addition on *Calluna* growth resulting in reduced canopy height and cover (Barker *et al.*, 2004). Burning also promoted regeneration from seed however, and provided an opportunity for colonisation of grasses, potentially having negative effects on vegetation composition in the longer term (Barker *et al.*, 2004).

Only one study has addressed the effectiveness of burning as a nitrogen mitigation technique in upland heathlands. In a Welsh *Calluna* dominated heath, a management burn on plots previously exposed to nitrogen additions for 11 years removed the equivalent of 8 years of nitrogen deposition inputs (Pilkington *et al.*, 2007). Burning also resulted in an increase in nitrogen leaching for 2-3 years following the fire, and increased nitrogen saturation of mineral soil layers. The authors suggest that burning on a 10–15-year cycle could be effective at removing accumulated nitrogen, but increased nitrogen leaching would pose a risk to surface water quality, particularly where heathlands have previously been exposed to high levels of nitrogen deposition.

Mowing. Mowing has been tested as a mitigation technique for nitrogen deposition in lowland dry heathland only. On an English heath 16% and 23% of total nitrogen stocks were removed by low and high intensity mowing respectively, while in a German study mowing to 10 cm height removed the equivalent of 5 years' nitrogen deposition inputs (Barker *et al.*, 2004; Hardtle *et al.*, 2006). Some additional nitrogen was also lost due to enhanced leaching following management (Hardtle *et al.*, 2007b), but site nitrogen budgets suggested that mowing could not fully compensate for elevated nitrogen inputs (Hardtle *et al.*, 2006). Mowing also strongly impacted total ecosystem phosphorus stocks, suggesting that repeated mowing could cause a shift towards phosphorus limitation, exacerbating some of the effects of nitrogen deposition and favouring species such as *Molinia caerulea* (Hardtle *et al.*, 2006; Hardtle *et al.*, 2009). In terms of effects on the plant community, mowing results in rapid regeneration of dwarf-shrubs from rootstocks and thus may be effective at reducing competition from grasses, promoting shrub dominance in the longer term (Barker *et al.*, 2004).

Grazing. Grazing has been tested as a nitrogen deposition mitigation technique for lowland heathlands in Germany. Nutrient budgets for a lowland dry heathland grazed by a shepherded flock taken off the heath (folded) overnight and provided with no additional fodder, suggest that grazing reduced the nitrogen stock in above ground biomass and could compensate for inputs of nitrogen from atmospheric deposition over the long term (Fottner *et al.*, 2007; Hardtle *et al.*, 2009). Grazing also caused a high net loss of phosphorus however, and this would result in an undesirable shift to phosphorus limitation in the longer-term. Achieving a net export of nutrients from the grazed heath was reliant on sheep being folded away from the heath at night when most defecation takes place, which is not typical of grazing management in Scotland. The impact of grazing by free-ranging red deer has also been tested in Germany. With a density of 25 individuals per km^{-2} ranging across a mosaic of heathland and woodland habitats, a net export of 14 $kg\ N\ ha^{-1}\ y^{-1}$ was found for heathland, which could compensate for atmospheric inputs of nitrogen at this site (Riesch *et al.*, 2022). Nutrient exports in culled deer carcasses were small however, and most nutrient export from the heathland was likely to occur because of nutrient transport between feeding and resting habitats, suggesting that other habitats within the mosaic may become nutrient enriched.

Only one study has investigated interactive effects of nitrogen deposition and grazing on upland moorland habitats. Smith *et al.* (2015) found that for UK moorlands receiving greater than 7 kg N ha^{-}

^{1} y^{-1}, removal of grazing resulted in an increase in plant and soil C stocks, suggesting that grazing may be at least partly mitigating the nitrogen-induced build-up of organic matter.

Disturbance. The effectiveness of disturbance techniques which remove vegetation and soil for mitigation of nitrogen impacts and restoration of heathland vegetation has been tested in European lowland heathlands. Turf or sod cutting removes all above ground vegetation plus the litter layer, organic horizon and A horizon, leaving a bare mineral surface. Choppering is a less intensive technique which removes above ground vegetation, litter and part of the O horizon thus producing less waste. Sod cutting removes a significant proportion of the ecosystem nitrogen stocks, which are concentrated in the organic soil horizon, and has potential to maintain low nutrient conditions over the long term (De Graaf *et al.*, 1998; Hardtle *et al.*, 2009). However, it also removes carbon, phosphorus and other nutrients. Calculation of nutrient budgets for a German lowland heath suggested that sod cutting removed nitrogen equivalent to 89 years of deposition but also removed phosphorus equivalent to 144 years input (Niemeyer *et al.*, 2007). The high output: input ratio for phosphorus with this technique would likely lead to phosphorus limitation in the longer term (Hardtle *et al.*, 2006). In comparison, choppering at the same German site removed the equivalent of 61 years of nitrogen input and 83 years of phosphorus input, but removed a greater volume of nitrogen per unit waste due to the concentration of nutrients in the upper soil layers (Niemeyer *et al.*, 2007). Since both sod-cutting and choppering remove above ground vegetation and thus reduce biological uptake of nitrogen, a spike of nitrogen availability in the soil and leaching of nitrogen to soil water follows management (De Graaf *et al.*, 1998; Hardtle *et al.*, 2007b).

Aside from effects on ecosystem nutrient pools, sod cutting and choppering also appear to be effective at regenerating both wet and dry lowland heathland vegetation, particularly where this is dominated by *Calluna* (Jansen *et al.*, 1996; De Graaf *et al.*, 1998). However not all species respond positively to this type of management and species which tend to spread vegetatively and those with small seedbanks, such as *Empetrum* sp., *Parnassia palustris*, *Succisa* sp. and *Dactylorhiza* sp. may fail to re-establish (Jansen *et al.*, 1996; De Graaf *et al.*, 1998). Disturbance techniques also have significant impacts (both positive and negative) on the wider heathland community of vertebrates and invertebrates, through impacts on habitat structure, food availability and food quality (Vogels *et al.*, 2017; Vogels *et al.*, 2021).

Suitability of disturbance techniques such as sod cutting and choppering for typical Scottish upland heaths with deep organic horizons has not yet been tested. There may be significant technical constraints to applying this technique on wet organic soils, and exposure of bare soil surfaces in the uplands is likely to be undesirable due to risks of erosion.

Liming and nutrient additions. While disturbance treatments can be effective at reducing ecosystem nitrogen stocks, they are not effective at counteracting acidification resulting from nitrogen deposition and can also result in increasing phosphorus limitation due to unbalanced removal of nutrients. Addition of lime and phosphorus to heathlands managed by sod-cutting has been tested as a means to alleviate these issues. Addition of lime to sod cut plots was found to rapidly increase soil pH and base cation concentrations and to reduce the aluminium: calcium ratio and soil ammonium concentration, thus mitigating some of the negative effects of acidification (De Graaf *et al.*, 1998; Vogels *et al.*, 2020). However, while liming enabled more acid-sensitive plant species to establish, some of these species were characteristic of grasslands rather than heathlands and key heathland species were sometimes negatively impacted (De Graaf *et al.*, 1998; Vogels *et al.*, 2020). Effects of phosphorus addition appear to be similar, except that acid-insensitive species also increased after phosphorus addition (Vogels *et al.*, 2020).

Soil nutrient imbalances resulting from disturbance management can also impact heathland fauna through changes in plant stoichiometry and food quality (Vogels *et al.*, 2017). Addition of phosphorus and calcium after sod-cutting can alleviate phosphorus limitation and has been found to result in significant increases of soil micro-arthropod fauna, particularly medium-sized herbivores and predators, but also to have negative impacts on fungivorous species (Siepel *et al.*, 2018). Above ground plant-feeding invertebrates are also negatively impacted by high plant N:P ratios. Improvement of food quality by phosphorus addition improved the nutritional status and increased the reproductive rate of female crickets, which produced the same number of eggs but over a shorter time period than crickets with lower quality diets (Vogels *et al.*, 2021). Addition of lime however, skewed Mn:Mg and Fe:Mg ratios in ways which were unfavourable to terrestrial invertebrates (Vogels *et al.*, 2021).

Summary – Heathlands

Natural recovery of heathland communities from nitrogen deposition appears likely to be slow. While some aspects of soil chemistry such as pH appear to recover fairly quickly, nitrogen is strongly retained within soils and vegetation and continues to be cycled within the ecosystem. Recovery of plant community composition appears to be slow, with little change after a decade of recovery. Heathlands are one of the best-studied ecosystems in terms of nitrogen mitigation through habitat management, but most evidence comes from lowland dry heathlands which are not particularly prevalent in Scotland, and potential management options need to be trialled in more typical Scottish upland heaths. Burning, mowing, grazing and disturbance such as turf-cutting can all be effective at reducing accumulated nitrogen stocks in vegetation and soil and restoring stand structure, but need to be correctly implemented. Grazing animals need to be folded (removed at night) to effectively export nutrients from heathland areas, and consideration needs to be given to the practicalities of waste disposal from techniques such as turf cutting. Burning can have adverse impacts on air quality and may cause export of nutrients to other areas. All of these techniques could also have implications for carbon stocks in heathland soils, which can be considerable in typical Scottish wet upland heath with deep organic soil horizons. Effects of management on the relative availability of nutrients such as phosphorus and potassium also need to be considered, as excessive depletion of these nutrients may actually exacerbate some of the effects of nitrogen deposition and can have negative effects on organisms further up the food chain. Experiments are needed to test the effects of potential management options in Scottish upland heaths before making recommendations.

Forests and woodlands (EUNIS T)

Natural recovery

In general, studies suggest that with nitrogen deposition rates below 10 kg N ha^{-1} y^{-1} all deposited nitrogen is retained within forest ecosystems, while above this level increasing amounts are lost as leachate and above 25 kg N ha^{-1} y^{-1} losses are high as the ecosystem becomes nitrogen saturated (Wright *et al.*, 1995). Nitrogen losses *via* leachate are accompanied by losses of base cations and decreasing pH, indicating acidification (Högberg *et al.*, 2006). Trees take up additional nitrogen resulting in higher foliage nitrogen concentrations and an initially positive growth response, but this can decline or become negative over time due to increasing nutritional imbalances (Högberg *et al.*, 2006; Blasko *et al.*, 2013). Soil fungal communities show altered composition, with declines of ectomycorrhizal (ECM) species as trees become less reliant on fungi able to obtain nitrogen from organic sources and reduce the amount of carbon supplied to fungal partners (Arnolds, 1991; Högberg *et al.*, 2011; Högberg *et al.*, 2014b; Choma *et al.*, 2017). Changes in forest understory vegetation composition can also occur, with declines in some shrub and moss species and an increase in grasses and species associated with eutrophic conditions (Nordin *et al.*, 2005). Epiphytic moss and lichen communities are extremely sensitive to nitrogen deposition (Mitchell *et al.*, 2004; Mitchell *et al.*, 2005; Geiser *et al.*, 2010; Giordani *et al.*, 2014), with European data showing declining lichen diversity and biomass when deposition exceeds 2.4 kg N ha^{-1} y^{-1} (Giordani *et al.*, 2014).

Most studies of natural recovery from nitrogen additions have been in coniferous forests, particularly in areas of Scandinavia with generally low background rates of nitrogen deposition. Following reduction or cessation of nitrogen additions, nitrogen leaching is reduced with rapid responses (within 1 year) often observed (Wright *et al.*, 1995; Högberg *et al.*, 2011; Dorr *et al.*, 2012). Nitrogen mineralisation and availability of ammonium and nitrate in the soil have also been seen to decline back to control levels within 10-20 years (Högberg *et al.*, 2006; Blasko *et al.*, 2013). Soil nitrogen pools may remain elevated however (Schmitz *et al.*, 2019), with one study finding no evidence of change after 13 years of recovery (Nohrstedt *et al.*, 2000). These experimental studies concur with survey based observations across Europe which show a decrease in nitrogen leaching, increase of pH and reduction of base cation and aluminium leaching in response to declining nitrogen deposition (Schmitz *et al.*, 2019).

Evidence also suggests that plant chemistry responds to decreased nitrogen input over a 10-20 year timescale with decreased conifer needle nitrogen content and increased base status observed (Högberg *et al.*, 2006; Dorr *et al.*, 2012; Blasko *et al.*, 2013). In some cases, changes in tissue needle chemistry within 2-4 years have been reported (Wright *et al.*, 1995). However other studies report that needle nitrogen content may still be elevated after 18 years (Högberg *et al.*, 2014a) and observations of long term change in needle chemistry across Europe suggest that pine needle nitrogen content is stable despite decreasing deposition (Schmitz *et al.*, 2019). The limited observations in broadleaf forests show declines in beech and oak leaf nitrogen content, but also a continuing decline in phosphorus content, resulting in increasing N:P (Schmitz *et al.*, 2019).

Several studies have investigated the responses of ECM fungi in coniferous forests to reductions of nitrogen inputs. Increased fruit body production has been seen after 9 years of recovery (Strengbom *et al.*, 2001), while stable isotope studies suggest that nitrogen uptake from organic sources by ECM fungi begins to increase after 6 years (Högberg *et al.*, 2011). After 14-18 years of recovery there is evidence that ECM community composition and abundance can recover and that trees increasingly supply carbon belowground in order to increase uptake of organic nitrogen *via* fungi (Högberg *et al.*, 2014a; Högberg *et al.*, 2014b). However, nitrogen effects on ECM community composition can persist for at least 23 years (Choma *et al.*, 2017) due to the high nitrogen retention capacity of these ecosystems, with one study reporting that nitrogen-sensitive ECM species were still reduced after 47 years (Strengbom *et al.*, 2001). In a deciduous forest in the USA with a mixture of ECM and arbuscular mycorrhizal (AM) mycorrhizal tree species, one recent study found that nitrogen fertilisation altered the nutrient uptake strategies of trees and their mycorrhiza, with ECM trees switching from enzymatic mining of organic nitrogen to root foraging, while AM trees switched from mycorrhizal foraging to root foraging (Carrara *et al.*, 2022). When nutrient additions ceased, tree root architecture and enzymatic mining of nitrogen recovered within one year in ECM trees, while AM trees showed no recovery. Mycorrhizal colonisation remained depressed in both tree types, however. Surveillance data from the Netherlands covering both conifers and birch woodlands concur with experimental observations, with ECM populations showing a switch from a decreasing to an increasing trend when nitrogen deposition declines, and recovery first being detected after 4-6 years of reduced nitrogen deposition (van Strien *et al.*, 2018). These surveillance data also show that recovery is strongest where nitrogen deposition has been lowest.

Only a small number of studies have investigated the recovery of forest ground flora from nitrogen addition. Mosses in particular may show long term effects of nitrogen addition, with moss cover still suppressed after 9 years of recovery (Nordin *et al.*, 2005). Another study found abundance of nitrophilic bryophytes still enhanced and nitrogen sensitive bryophytes suppressed after 47 years of recovery (Strengbom *et al.*, 2001). Modelling studies suggest that recovery of forest ground flora in response to reductions in background nitrogen deposition across Europe are likely to be small, as deposition reductions are unlikely to reduce eutrophication, and concurrent climate change and reductions in sulphur deposition may further disadvantage oligotrophic species, many of which are also acidophilic and cold tolerant (Dirnbock *et al.*, 2018). In the case of epiphytic bryophyte communities, studies in UK Atlantic oakwoods showed that when bryophytes were transplanted from high to low deposition sites, tissue nitrogen content declined and growth increased within 1 year, although responses to decreased nitrogen were smaller and slower than those to increased nitrogen (Mitchell *et al.*, 2004).

Mitigation

The nature of forest ecosystems means that they are generally less intensively managed, particularly in the case of semi-natural forests not used for production forestry. A range of mitigation techniques have been tested, which focus on remediating different nitrogen impacts such as enhanced nitrogen availability in the soil, accumulated nitrogen pools in biomass and soil, acidification or reduction in habitat suitability for key species. Effects of burning, thinning, liming and carbon addition (to immobilise nitrogen) were the subject of a recent review and meta-analysis in the context of American forests by Clark *et al.* (2019). Here we summarise their findings and include additional European studies and techniques including sod-cutting, litter removal and nutrient addition.

Burning. Prescribed burning is not generally used as a forest management technique in the UK, but Clark *et al.* (2019) reviewed 17 studies testing the nitrogen mitigation potential of this technique in the USA, where burning is more commonly used to control fuel loads. They found that burning was not effective at mitigating nitrogen impacts on forests as it tended to increase nitrogen availability in the short term, had no effect on soil pH and could be detrimental to plant diversity. However, most of the studies were short term (1-4 years) and a limited amount of longer-term experimental evidence showed initial pulses of nitrogen availability were not sustained after 11 years (Ganzlin *et al.*, 2016). Reductions in soil and vegetation nitrogen pools following fire also appeared to transitory, returning to control levels within five years. There was also some evidence that burning led to forest ground flora being increasingly dominated by a small number of species, but long-term data were lacking (Clark *et al.*, 2019).

Disturbance. Although not typically used as a forest management, sod cutting has been tested as a mitigation technique for nitrogen deposition in Dutch Scots pine forests (Smit *et al.*, 2003; Boxman & Roelofs, 2006). After three years, sod cutting reduced nitrogen leaching in forests exposed to high nitrogen deposition (42 kg N ha^{-1} y^{-1}) but was most effective in forests receiving a lower deposition rate of 15-20 kg N ha^{-1} y^{-1} (Boxman & Roelofs, 2006). Effects of sod cutting on ECM fungal communities were tested by Smit *et al.* (2003) who found increased fungal fruitbody production after three years and a greater diversity of ECM fungi detected as fruitbodies and in the soil after five years, suggesting that sod cutting is effective as a restoration measure.

Tree thinning is carried out routinely in production forestry, but was also explored as a nitrogen mitigation option by Clark *et al.* (2019) who reviewed 13 studies and found it had similar effects to burning, with slightly increased nitrogen availability in the short term and no, or negative longer term effects on nitrogen availability, soil acidity and understory vegetation. Thinning removes carbon and nitrogen in the form of stem wood, but the overall effects depend on whether slash from the thinning is removed or left on site. Where carbon rich material is left onsite it may additionally enhance immobilisation of nitrogen by soil microbes (Wolk & Rocca, 2009). Thinning increased ground flora plant richness and decreased Shannon diversity, but effects varied according to whether slash was retained or removed. Thinning was also reported to have less effect on low diversity forest understory vegetation or at higher latitudes (Clark *et al.*, 2019). In a recent study in a European boreal forest, Jorgensen *et al.* (2022) found that while nitrogen fertilisation increased fungal biomass in the soil, thinning appeared to reduce fertilisation effects.

Liming. Applications of lime have been tested as a means of mitigating soil acidification in both coniferous and broadleaf forests exposed to high levels of nitrogen and sulphur deposition and are widely used in countries such as Germany (Thomas *et al.*, 2019). Liming is a short term tool to mitigate acidification but its effects tend to be focused on the upper part of the soil profile, which may lead to roots accumulating at the soil surface; it may also stimulate nitrification resulting in increased nitrogen leaching as nitrate (Matzner & Dise, 1996). In pine and spruce forest in the Netherlands and Germany, liming reduced soil acidity but was not effective at reducing nitrogen leaching (Durka *et al.*, 1994; Boxman & Roelofs, 2006). While in a German beech forest, liming increased nitrogen mineralisation but not microbial biomass, and increased nitrogen leaching (Corre *et al.*, 2003). Similarly in their meta-analysis of American studies Clark *et al.* (2019) found that while liming was effective at decreasing soil acidity, it also increased soil nitrogen availability, increasing the risk of nitrogen leaching to surface waters.

Effects of liming on forest ground flora were investigated in a number of studies in the late 1990s and early 2000s (Thomas *et al.*, 2019) with either no effect or increased occurrence of nitrogen indicator species reported from short to medium term studies (2-10 years). In a longer 20 year study however, Thomas *et al.* (2019) found that while 10 years after liming, Ellenberg indicator values for nitrogen were elevated across a range of forest types and in spruce forests there was evidence of decreased acidity, the effects of liming on ground flora reduced or disappeared after 20 years, and they concluded liming could safely be used to mitigate nitrogen impacts.

Nutrient addition. Since nitrogen deposition commonly results in altered nutrient ratios in vegetation and soils and can result in tree health declines due to nutrient imbalance, compensating fertiliser additions of phosphorus, potassium (K) and magnesium (Mg) have been explored as potential mitigation options. Boxman and Roelofs (2006) added K and Mg to Dutch pine forest receiving low or high nitrogen deposition but found that while adding fertiliser under low deposition conditions led to some improvements in nutrient balance in the soil, under high nitrogen deposition it led to very high leaching of nutrients into soil water. In a more recent study of phosphorus addition to mixed broadleaf-pine forest in China, Xia *et al.* (2020) found that phosphorus mitigated the negative impacts of nitrogen addition on soil bacterial community diversity and functional composition. Aside from addition of chemical fertilisers, nutrient inputs to forest ecosystems can also be altered by manipulating the chemical composition of leaf litter entering the soil. Desie *et al.* (2020) tested the effects of variation in leaf litter chemistry on soil chemistry and soil biota in forests exposed to high nitrogen deposition. They found that on poorly buffered soils, tree species with litter rich in base cations induced faster nutrient cycling, improved topsoil base saturation and higher earthworm biomass, suggesting that including such ‘rich litter’ species in tree planting schemes may help to improve forest vitality and mitigate nitrogen impacts.

Carbon additions. Theory suggests that addition of carbon rich materials to soil should induce the soil microbial community to immobilise nitrogen, reducing its availability to plants (Török *et al.*, 2000). This technique has mainly been tested in grasslands, but a small number of studies have tested its effects in forests (Clark *et al.*, 2019). Some studies report reductions in soil nitrogen availability with carbon addition (Cassidy *et al.*, 2004), while others report no effect (Hunt *et al.*, 1988; Koorem *et al.*, 2012), or significant effects only during certain seasons (Chapman *et al.*, 2016). Likewise, there is limited evidence on the effects of carbon additions on soil acidity and ground flora composition, and more studies are needed to determine long term effectiveness (Clark *et al.*, 2019).

Summary – Forests

Forests are one of the best studied ecosystems in terms of natural recovery from nitrogen deposition with a variety of aspects of ecosystem composition and function having been investigated. Leaching of nitrogen from forest ecosystems appears to respond rapidly to reductions in nitrogen inputs with declines reported within one year. Within 1-2 decades of nitrogen reductions, nitrogen mineralization rates in the soil decline and there is evidence of recovery of ECM fungal community activity. Soil nitrogen pools remain elevated over longer timescales however, and species composition of ground flora and fungal communities likewise appear to remain impacted over several decades. In general, forest ecosystem recovery appears to be better in areas where nitrogen deposition exceedance of critical loads has been lower, but interactions with climate change and other drivers could change ecosystem trajectories and prevent recovery to prior states in the longer term. A variety of management techniques for nitrogen mitigation have been tested in forest ecosystems but results have generally been inconclusive. Burning had mainly negative effects, while thinning and carbon addition had limited impacts. Theory suggests that while removal of thinned wood would export nitrogen from the ecosystem, retention or addition of carbon-rich materials on the forest floor should enhance immobilization of nitrogen, but experimental results have been inconclusive. Liming appears to be effective at reducing the effects of acidification but could have unwanted side effects resulting in increased nitrogen availability and increased nitrogen leaching. Likewise, nutrient additions can also be used to alleviate nutritional imbalances in trees subject to high nitrogen deposition but can also result in increased nitrogen leaching. Given that none of these options appear likely to have highly beneficial results, reduction in nitrogen deposition must be the main focus for promoting recovery of forest ecosystems.

Potential for nitrogen mitigation through management in Scottish semi-natural ecosystems

Evidence for natural recovery from nitrogen deposition was focused on bog, grassland, alpine moss heath, dwarf-shrub heath and forest ecosystems. Generalising across these ecosystems, most studies suggested that reductions in nitrogen leaching, and recovery of soil or soil water pH could be expected relatively rapidly, within months to years of reductions in nitrogen deposition. Reduction of nitrogen leaching is particularly important as it benefits for downslope terrestrial habitats and freshwaters by reducing lateral transfer of nitrogen between ecosystems. There was also evidence that bryophytes, which play important functional roles in many Scottish habitats, showed relatively rapid recovery following deposition reductions, with tissue chemistry, growth and nitrogen uptake capacity recovering over timescales of months to years. However, where bryophyte cover had been severely reduced or species had been lost, the evidence suggested that recovery may take much longer (decades). In the medium term (up to 20 years) evidence from grasslands, dwarf-shrub heaths and forests suggested that nitrogen mineralisation and nitrogen availability to rooted plants may start to decline and functioning of mycorrhizal symbioses may start to recover. However, across all ecosystems for which there was evidence, nitrogen stocks in the soil remained elevated over the long term and recovery of plant diversity was slow, both showing little change over several decades. This suggests that natural recovery of ecosystems from nitrogen deposition may be only partial, particularly in soil-based ecosystems. Although nitrogen deposition reductions could bring some relatively rapid improvements, for example to surface water quality and bryophyte communities, full recovery may take many decades, if it occurs at all. Several studies did however report that ecosystem recovery was best in low deposition areas where critical loads had not been greatly exceeded, and this would currently be the case for large areas of Scotland.

The effectiveness of management to mitigate nitrogen impacts has been tested in dunes, fens, grasslands, alpine moss heaths, dwarf shrub heaths and forests. For many systems, the evidence was limited and covered only a small number of potential management options. Studies often included only selected aspects of ecosystem response, for example focussing on vascular plant diversity or nitrogen stocks. There was evidence that some management activities could help to mitigate nitrogen impacts. In grasslands and forests, liming reduced acidification and in grasslands grazing or mowing was able to maintain species richness (but not necessarily species composition). Management to reduce vegetation and soil nitrogen stocks was best explored in heathlands, where burning, mowing, grazing and turf-cutting all had potential to reduce nitrogen accumulation. However, there was also evidence that some management interventions could have negative effects, for example increased nitrogen leaching following vegetation/soil disturbance, increased export of nitrogen to other ecosystems, or changes in plant stoichiometry or habitat structure which adversely impacted fauna. All managements had differing benefits and trade-offs. Walmsley *et al.* (2021) explored this in detail for lowland dry heath and concluded that different combinations of management in time and space provided different benefits in terms of ecosystem services and functions. There was a risk that trying to solve one problem could create another, and novel or adapted management practices might be needed to remove nitrogen from ecosystems while maintaining other functions. Trade-offs between carbon storage and nitrogen removal would be one important example. In conclusion, it will be important to have a holistic view of habitat management impacts under Scottish conditions and to understand the potential trade-offs before attempting to use these techniques for nitrogen mitigation in Scotland.

In Table 3, we summarise the potential for nitrogen impact mitigation through management for the full range of Scottish semi-natural terrestrial habitats. Of the 76 EUNIS level 3 terrestrial semi-natural habitats occurring in Scotland, 48 are not usually subject to any form of management, 11 are typically managed and a further 17 may be managed in some locations. Those habitats which are not usually managed comprise near natural habitats in the alpine zone and on cliffs and scree, wetlands such as springs, fens and swamps, coastal and inland scrub communities, and some types of forests. For 32 of these habitats there was not judged to be any appropriate form of management that might mitigate nitrogen impacts. For these habitats especially, reduction of nitrogen deposition or other nitrogen inputs should be the primary focus. Some of the most widespread Scottish semi-natural habitats are however managed to some degree. These include grasslands, which almost always require some form of management to maintain them, and temperate heathlands, coastal grasslands and heaths, some wetlands, bogs and forests which may currently be managed in some but not all locations. For these habitats there could be potential to mitigate nitrogen impacts through management practices which maintain or restore vegetation structure or contribute to depletion of accumulated nitrogen stocks. However, the strength of evidence for effectiveness of these interventions is variable and even where evidence exists, we recommend that trials should be undertaken to explore their efficacy under Scottish conditions (climate and soils) and to identify any unintended negative consequences.

Table 3. *Summary of typical management methods in Scottish semi-natural terrestrial ecosystems and scope for mitigation of nitrogen impacts through management actions.*

EUNIS 2022 code	EUNIS 2022 habitat name	Habitat is typically managed? Y/N/S (S= in some places)	Types of management currently used	Scope for N mitigation through management? Y/N/P (Y = Scope + evidence, P = Scope but no evidence, N = no scope)	Potential management options for N mitigation
M	**Marine benthic habitats**				
MA1	***Littoral rock***	N	NA	N	
MA2	***Littoral biogenic habitat***				
MA22	Atlantic littoral biogenic habitat (saltmarsh)	S	Grazing	P	Grazing
N	**Coastal habitats**				
N1	***Coastal dunes and sandy shores***				
N11	Atlantic, Baltic and Arctic sand beach	N	NA	N	
N13	Atlantic and Baltic shifting coastal dune	S	Grazing	N	
N15	Atlantic and Baltic coastal dune grassland (grey dune)	S	Grazing, disturbance	Y	Grazing, mechanical disturbance
N18	Atlantic and Baltic coastal *Empetrum* heath	S	Grazing	P	Grazing, cutting
N19	Atlantic coastal *Calluna* and *Ulex* heath	S	Grazing	P	Grazing, cutting
N1A	Atlantic and Baltic coastal dune scrub	N	NA	P	Grazing, cutting
N1H	Atlantic and Baltic moist and wet dune slack	S	Grazing, disturbance	Y	Grazing, mechanical disturbance
N2	***Coastal shingle***				
N21	Atlantic, Baltic and Arctic coastal shingle beach	N	NA	N	
N23	Shingle and gravel beach with scrub	N	NA	P	Grazing, cutting
N24	Shingle and gravel beach forest	N	NA	P	Grazing, thinning
N3	***Rock cliffs, ledges and shores, including the supralittoral***				
N31	Atlantic and Baltic rocky sea cliff and shore	N	NA	N	
N24	Atlantic and Baltic soft sea cliff	N	NA	N	

Q	**Wetlands**				
Q1	***Raised and blanket mires***				
Q11	Raised bog	S	Grazing	P	Grazing
Q12	Blanket bog	S	Grazing	P	Grazing
Q2	***Valley mires, poor fens and transition mires***				
Q21	Oceanic valley mire	N	NA	N	
Q22	Poor fen	N	NA	N	
Q24	Intermediate fen and soft-water spring mire	N	NA	N	
Q25	Non-calcareous quaking mire	N	NA	N	
Q4	***Base-rich fens and calcareous spring-mires***				
Q41	Alkaline, calcareous, carbonate-rich small-sedge spring fen	N	NA	N	
Q42	Extremely rich moss-sedge fen	N	NA	N	
Q43	Tall-sedge base-rich fen	N	NA	N	
Q44	Calcareous quaking mire	N	NA	N	
Q45	Arctic-alpine rich fen	N	NA	N	
Q5	***Helophyte beds***				
Q51	Tall-helophyte bed	N	N	P	Cutting
Q52	Small-helophyte bed	N	N	P	Cutting
Q53	Tall-sedge bed	N	N	P	Cutting
R	**Grasslands and lands dominated by forbs mosses or lichens**				
R1	***Dry grasslands***				
R1A	Semi-dry perennial calcareous grassland (meadow steppe)	Y	Grazing	Y	Grazing, cutting, liming
R1M	Lowland to montane, dry to mesic grassland usually dominated by *Nardus stricta*	Y	Grazing	Y	Grazing, cutting, liming
R1P	Oceanic to subcontinental inland sand grassland on dry acid and neutral soils	Y	Grazing	Y	Grazing, cutting, liming
R1S	Heavy-metal grassland in western and central Europe	Y	Grazing	Y	Grazing, cutting, liming
R2	***Mesic grasslands***				
R21	Mesic permanent pasture of lowlands and mountains	Y	Grazing, cutting	Y	Grazing, cutting, liming

R22	Low and medium altitude hay meadow	Y	Grazing, cutting	Y	Grazing, cutting, liming
R3	***Seasonally wet and wet grasslands***				
R35	Moist or wet mesotrophic to eutrophic hay meadow	Y	Grazing, cutting	Y	Grazing, cutting, liming
R36	Moist or wet mesotrophic to eutrophic pasture	Y	Grazing, cutting	Y	Grazing, cutting, liming
R37	Temperate and boreal moist or wet oligotrophic grassland	Y	Grazing, burning	Y	Grazing, cutting, liming, burning
R4	***Alpine and subalpine grasslands***				
R41	Snow-bed vegetation	N	NA	N	
R42	Boreal and arctic acidophilous alpine grassland	S	Grazing	Y	Reduce grazing, P addition
R43	Temperate acidophilous alpine grassland	S	Grazing	Y	Grazing
R5	***Woodland fringes and clearings and tall forb stands***				
R52	Forest fringe of acidic nutrient-poor soils	N	NA	P	Cutting
R54	*Pteridium aquilinum* vegetation	S	Cutting, herbicide	N	
R55	Lowland moist or wet tall-herb and fern fringe	N	NA	N	
R56	Montane to subalpine moist or wet tall-herb and fern fringe	N	NA	N	
S	**Heathland scrub and tundra**				
S2	***Arctic, alpine and subalpine scrub***				
S21	Subarctic and alpine dwarf *Salix* scrub	N	NA	N	
S22	Alpine and subalpine ericoid heath	N	NA	P	Grazing
S23	Alpine and subalpine *Juniperus* scrub	N	NA	N	
S25	Subalpine and subarctic deciduous scrub	N	NA	N	
S27	Krummholz with conifers other than *Pinus mugo*	N	NA	N	
S3	***Temperate and mediterranean-montane scrub***				
S31	Lowland to montane temperate and submediterranean *Juniperus* scrub	S	Grazing	N	
S32	Temperate *Rubus* scrub	N	NA	N	
S35	Temperate and submediterranean thorn scrub	N	NA	P	Cutting, grazing
S37	*Corylus avellana* scrub	S	Coppicing	P	Coppicing, grazing
S38	Temperate forest clearing scrub	N	NA	P	Burning, cutting, grazing
S4	***Temperate heathland***				

S41	Wet heath	S	Grazing, burning	Y	Grazing, burning
S42	Dry heath	S	Grazing, burning, cutting	Y	Grazing, burning, cutting, disturbance (e.g. choppering), liming, P addition
S9	***Riverine and fen scrub***				
S92	*Salix* fen scrub	S	Grazing	P	Grazing
T	**Forest and other wooded land**				
T1	***Broadleaved deciduous forests***				
T11	Temperate *Salix* and *Populus* riparian forest	N	NA	N	
T12	*Alnus glutinosa-Alnus incana* forest on riparian and mineral soils	N	NA	P	Grazing
T15	Broadleaved swamp forest on non-acid peat	N	NA	N	
T16	Broadleaved mire forest on acid peat	N	NA	N	
T18	*Fagus* forest on acid soils	N	NA	P	Liming, nutrient addition
T1B	Acidophilous *Quercus* forest	S	Coppicing	P	Liming, nutrient addition, grazing
T1C	Temperate and boreal mountain *Betula* and *Populus tremula* forest on mineral soils	N	NA	P	Grazing, liming, nutrient addition, thinning
T1E	*Carpinus* and *Quercus* mesic deciduous forest	N	NA	P	Grazing, liming, nutrient addition,
T1H	Broadleaved deciduous plantations of non-site-native trees	Y	Felling	Y	Grazing, liming, nutrient addition, thinning
T3	***Coniferous forests***				
T35	Temperate continental *Pinus sylvestris* forest	N	NA	Y	Grazing, liming, nutrient addition, thinning
T3J	*Pinus* and *Larix* mire forest	N	NA	P	Grazing, liming, nutrient addition, thinning
T3M	Coniferous plantation of non-site-native trees	Y	Felling	Y	Liming, nutrient addition, thinning
U	**Inland habitats with no or little soil and mostly with sparse vegetation**				
U2	***Screes***				
U22	Temperate high-mountain siliceous scree	N	NA	N	
U26	Temperate high-mountain base-rich scree and moraine	N	NA	N	
U3	***Inland cliffs, rock pavements and outcrops***				

U31	Boreal and arctic siliceous inland cliff	N	NA	N	
U35	Boreal and arctic base-rich inland cliff	N	NA	N	
U3D	Wet inland cliff	N	NA	N	
U3E	Limestone pavement	N	NA	N	
U5	***Miscellaneous inland habitats usually with very sparse or no vegetation***				
U51	Fjell field	N	NA	N	

Section 3: Metrics for monitoring the impacts of, and recovery from, nitrogen deposition.

Introduction

Although the negative impacts of nitrogen deposition on semi-natural ecosystem functioning and biodiversity are well known and have been demonstrated through numerous experiments and surveys, there are currently no effective metrics of nitrogen deposition impact included within regular statutory habitat condition monitoring in Scotland (Jones *et al.*, 2016; Britton & Ross, 2018). In order to be able to detect the impacts of nitrogen deposition on habitat condition in both protected areas and the wider countryside, and to monitor the progress of recovery in response to efforts to reduce air pollution, suitable indicator metrics are urgently needed. In this section of the report, we briefly review work carried out to date to develop indicator metrics for nitrogen impacts suitable for use in Scottish semi-natural habitats.

Selection of metrics

Many types of measurements of ecosystem functions and properties have been used to demonstrate the response of ecosystems to anthropogenic nitrogen inputs (see Section 1 of this report) but not all of these are suitable for use as metrics of nitrogen impacts across the wider landscape. An ideal metric of nitrogen impact should be responsive to changes in nitrogen impacts over reasonably short timescales, be uniquely influenced by nitrogen inputs and be easy to implement in country-wide monitoring, having a low skill requirement and low cost for data collection and interpretation. In reality no single metric is likely to meet all of these requirements and all metrics will have pros and cons. Metrics to indicate nitrogen deposition impacts can be divided into *pressure metrics* which are measures of nitrogen deposition inputs, and response metrics which could be *midpoint metrics* (measures of change in primarily chemical ecosystem properties which indicate a change in functioning), or *endpoint metrics* which are measures of biodiversity change or other 'final outcomes' relevant to people such as water quality (Rowe *et al.*, 2017).

Pressure metrics

Modelled maps of nitrogen deposition provide information on likely exposure of ecosystems to nitrogen deposition, but do not reflect the variation among ecosystems in sensitivity to these nitrogen inputs. Empirical critical loads of nitrogen quantify ecosystem sensitivity to nitrogen inputs and, when combined with deposition data, can be used to produce metrics describing risk to ecosystems from nitrogen deposition. Information on deposition is combined with maps of semi-natural habitat distribution and accompanying habitat specific critical loads, to quantify the amount of nitrogen deposition in excess of the critical load for the habitats present at a particular location. Known as the Average Accumulated Exceedance (Rowe *et al.*, 2022) this metric is calculated at the 1 km scale for the whole UK and is reported on by DEFRA as one of the UK biodiversity indicators, indicator B5a - Air Pollution (DEFRA, 2020). Pressure metrics do not however give any information on the extent to which ecosystem condition changes as a result of nitrogen impacts and this must be measured directly in the habitats themselves using a range of response metrics.

Response metrics

A wide range of ecosystem responses to nitrogen deposition have been recorded from nitrogen addition experiments and habitat surveys (Table 2). Midpoint responses include measurements of soil, plant and water chemical status such as soil %N, soil C:N ratio, peat water nitrogen content and vascular plant, bryophyte or lichen tissue nitrogen content. Non-chemical midpoint responses include aspects such as root mycorrhizal infection and vegetation height or biomass. Endpoint responses, particularly those relating to above ground biodiversity, are the most commonly measured and reported responses to nitrogen deposition (Table 2). These include a range of measures of vegetation community composition, species richness and diversity and that of the separate bryophyte, lichen and vascular plant communities. Some studies have also reported responses of mycorrhizal fungal and soil fauna community composition.

Some midpoint and endpoint metrics have been tested as potential indicators of nitrogen impacts in survey studies across the UK. In heathlands, Edmondson *et al.* (2010) and Caporn *et al.* (2014) tested a range of potential nitrogen indicators and found that vascular plant and bryophyte richness were both strongly related to nitrogen deposition while among biogeochemical indicators litter nitrogen content and *Calluna* foliar nitrogen content had the strongest relationships. In acid grasslands, (Stevens *et al.*, 2009) tested a range of potential nitrogen deposition metrics and found that graminoid:forb ratio was the best indicator of nitrogen impacts, with species richness and forb richness also being well correlated. A larger study of acid grasslands across Europe (Stevens *et al.*, 2011) tested tissue nitrogen of *Agrostis capillaris*, *Galium saxatile* and *Rhytidiadelphus squarrosus* as indicators of deposition, but none performed well and grass richness as a proportion of the total richness, and forb richness were again better indicators. The differences between these studies and between the ecosystems involved suggest that indicators will likely have to be habitat specific, however, some metrics have been suggested for use as broad scale cross-habitat measures. These include moss nitrogen content (Pitcairn *et al.*, 2006) further modified into a 'moss enrichment index' by Rowe *et al.* (2017). Lichen diversity has also been proposed as a sensitive bioindicator of nitrogen (and other types of pollution) in a number of settings including terricolous (ground dwelling) and epiphytic communities in woodland canopies (Rogers *et al.*, 2009; Stevens *et al.*, 2012b). Impacts of nitrogen on fauna are much less studied, but one example of a nitrogen indicator developed for faunal communities is the Community Nitrogen Index (CNI) for butterflies used in the Netherlands (WallisDeVries & van Swaay, 2017). This uses data from the Dutch national butterfly monitoring scheme and combines it with information on the Ellenberg N value of the plant community in which each butterfly species has highest occupancy to give a metric of average nitrogen preference of the butterfly community at a site.

While a range of potential nitrogen impact metrics have been explored in the UK, none has yet been fully tested and implemented. One of the biggest difficulties with using midpoint and endpoint metrics as indicators of nitrogen deposition is that they can also be affected by confounding factors such as climate, soil type and habitat management. While surveys of potential indicator metrics across multiple sites can show trends associated with nitrogen deposition, there is often a lot of variability in the relationships. The SEPA Botanical Benchmarks project (Jones *et al.*, 2018) tested a series of plant community metrics as indicators of nitrogen impacts across four widespread habitat types and found that not all metrics were related to nitrogen in all habitats and for those which were related to nitrogen deposition, the amount of scatter meant that identifying nitrogen impacts at a single site could be problematic. The authors suggested that lack of precision could be compensated for by having a 'basket' of nitrogen metrics and adopting a weight of evidence approach whereby indication of nitrogen exceedance by multiple metrics would give stronger evidence than failure of a single metric.

New types of metrics

New techniques in the ecosystem monitoring toolkit have the potential to produce new metrics and methods for monitoring the impact of changes in nitrogen deposition loads. Spectroscopy is one method which has been explored in relation to determining nitrogen impacts since spectroscopic information can be derived from cameras mounted on drones or remote sensing data. Spectral information can be used to predict foliar chemical composition and studies have explored whether this could be used to detect impacts of nitrogen on foliar chemistry. Gidman *et al.* (2006) found that FTIR spectra of *Galium saxatile* could be related to nitrogen deposition, but the predictive power of the relationship was fairly weak and they concluded that the method needed further development. Kalaitzidis *et al.* (2008) reached similar conclusions for *Calluna vulgaris*. More recent studies have shown that spectra are capable of detecting differences between species, but that they are also strongly affected by plant canopy morphology and other factors (Girard *et al.*, 2020; Moeneclaey *et al.*, 2022). While within-species differences in nitrogen status can be detected within monospecific stands of uniform structure (such as crops) the methods will require further development before nitrogen status could be remotely sensed in complex semi-natural communities. Girard *et al.* (2020) suggest that in the context of detecting nitrogen impacts the technique is currently best suited to remote sensing biodiversity changes rather than detecting within-species changes.

Other vegetation metrics also have potential for remote sensing - biomass, sward height, gross primary productivity and vegetation greenness can be determined from satellite data (Gimenez *et al.*, 2019; Jiang *et al.*, 2021). These metrics are influenced by nitrogen and this approach could allow remote monitoring over large scales, but these metrics will also be strongly influenced by climate, soil and management factors and so the approach might be better suited to detecting large scale nitrogen impacts across many sites, rather than determining single site condition.

Away from remote sensing, DNA technologies also have potential to play a role in future nitrogen impacts monitoring. Studies have shown that aspects of soil biodiversity such as mycorrhizal fungal communities and soil fauna are sensitive to nitrogen, but use of these as nitrogen metrics is limited by the taxonomic requirements of identification. Advances in soil DNA technology could allow rapid and cost-effective assessment of soil biodiversity in future and this has the potential to allow development of soil biodiversity metrics for nitrogen deposition impacts.

Knowledge gaps

Developing effective indicators for nitrogen deposition impacts across the full range of semi-natural ecosystems in Scotland will require knowledge gaps to be filled. At present many habitats are still unexplored in relation to their thresholds and responses to nitrogen deposition, options for nitrogen deposition metrics are therefore limited. Some general metrics such as moss nitrogen content may be suitable in certain habitats, but in general, this is a big gap in knowledge and work needs to be undertaken to identify impacts, thresholds and response metrics for these habitats.

Recommendations

Critical load exceedance (AAE) is well established as a pressure metric and although there are some uncertainties associated with pollution modelling data, it provides a way to visualise the potential impacts of nitrogen pollution and to measure progress with reducing pollution pressures.

Demonstrating impact of or recovery from nitrogen deposition in semi-natural ecosystems will require vegetation or soil-based metrics which can be readily sampled in a non-destructive way. Remote sensing technologies are not yet sufficiently developed to be deployed to detect the impacts of nitrogen and so monitoring will need to focus on more traditional soil and biodiversity sampling. The results of studies to date suggest that the selection of metrics will need to be habitat-specific and that several metrics should be selected for each habitat to overcome issues around the imprecise relationships between biodiversity or soil metrics and nitrogen deposition. Nitrogen impacts at single sites should then be assessed on the results of a basket of metrics rather than a single pass or fail. The basket of metrics could also aim to include a selection of faster and slower responding parameters; the relatively limited number of nitrogen recovery studies reviewed in Section 2 suggest that plant tissue nitrogen and measures of nitrogen turnover in the soil may be fairly quick to respond, while biodiversity indicators will be more demonstrative of the longer-term impacts of nitrogen. Many of the biodiversity indicators which have been proposed in the literature will also require more detailed botanical monitoring than is currently the case, e.g., full species lists to assess species richness or bryophyte species recording.

In order to allow interpretation of results from single sites, benchmarking data will be needed for all metrics in each habitat to inform on the range of potential values and the nature of the relationship between nitrogen deposition and the metric. For many of the more widespread habitats, suitable survey data to test and benchmark metrics may already exist. Although relatively few metrics have yet been fully tested as nitrogen indicators, this is a relatively straight forward task given sufficient resources.

Conclusions

Thresholds for nitrogen impacts and ecosystem responses to nitrogen deposition over and above these thresholds are currently known for approximately 60% of Scottish semi-natural habitats. The remaining 40% of habitats include many which are significant for biodiversity in Scotland, including communities of rocky substrates, alpine habitats, scrub communities and many wetlands. Filling this knowledge gap on the impacts of nitrogen in these habitats should be a priority. For those habitats which do have critical loads, much of the current knowledge on nitrogen impacts is focussed on above ground community responses and some aspects of plant and soil chemistry. Future studies should focus more attention on nitrogen impacts on belowground biodiversity which is critical to ecosystem functioning and also on wider above ground biodiversity beyond vascular plants.

Recovery from nitrogen impacts and potential management actions to improve or speed up recovery are much less studied than the effects of nitrogen deposition, however evidence was found for a number of important Scottish habitats. Evidence for natural recovery in bog, grassland, alpine moss heath, dwarf-shrub heath and forests suggested that some soil chemical parameters such as pH, nitrogen availability and leaching of excess nitrogen may recover fairly quickly without intervention. Bryophyte and lichen communities may also recover quickly, where recolonisation is not subject to dispersal limitations. The evidence suggests however that nitrogen stocks in soils and vegetation may remain elevated in the long term and biodiversity may be slow to recover, if it recovers at all. For the 60% of Scottish habitats which are not actively managed, natural recovery is generally the best or only option for recovery from nitrogen pollution. This emphasises the need for action to reduce nitrogen deposition to semi-natural ecosystems.

In habitats which are managed, there may be options to mitigate nitrogen impacts, with grasslands and heathlands (including coastal types) being most amenable to intervention. Several of the typical management techniques currently used on these communities in Scotland could be modified to maximise export of nitrogen and support the maintenance of biodiversity. However, studies elsewhere in the UK and Europe have shown that management to reduce nitrogen impacts can have undesirable (and sometime unforeseen) side effects, including transfer of nitrogen to other communities, loss of carbon stocks or negative impacts on some aspects of biodiversity. Potential mitigation management techniques need to be tested under Scottish conditions and a full analysis of benefits and trade-offs made before they are implemented more widely.

Monitoring of progress towards recovery from the impacts of nitrogen deposition requires suitable metrics. Exceedance of critical loads (Average Accumulated Exceedance) is a well-established metric for monitoring changes in nitrogen deposition pressures but does not reveal nitrogen impacts. Many different ecosystem responses to nitrogen deposition have been reported and a number of soil and vegetation metrics could be suitable for indicating impacts and recovery. Few of these metrics have been fully tested as indicators, however. Attempts to develop indicators so far suggest that sets of indicators will be habitat specific, and that benchmarking of relationships between nitrogen and the indicator in each habitat will be required. Detection of nitrogen impacts at a single site can be difficult due to variability in confounding factors such as climate and soils and stochastic variability. Detection of impacts could be improved by using a suite of nitrogen metrics to indicate the weight of evidence for nitrogen impacts. These metrics should include both rapidly responding parameters such as soil chemistry and biodiversity metrics indicative of longer-term impacts.

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Appendix 1: Habitat type correspondence table.

Table S1: *Summary of semi-natural terrestrial habitats present in Scotland classified to level 3 of the 2022 version of the European Nature Information System (EUNIS). For each level 3 EUNIS habitat equivalent communities in the UK National Vegetation Classification (NVC) and Natura 2000 classification are also given. Habitat listing and correspondences are based on Strachan (2017) with classification updated according to Chytry et al. (2020) and the 2021/22 revised EUNIS classification* (https://eunis.eea.europa.eu/habitats.jsp).

EUNIS level	EUNIS 2022 code	EUNIS 2022 habitat name	NVC communities	Natura2000
1	**M**	**Marine benthic habitats**		
2	***MA1***	***Littoral rock***	non-NVC lichen communities	
2	***MA2***	***Littoral biogenic habitat***		
3	MA22	Atlantic littoral biogenic habitat (saltmarsh)	SM5-6, SM8-10, SM12-20, SM23, SM27-28	H1330
1	**N**	**Coastal habitats**		
2	***N1***	***Coastal dunes and sandy shores***		
3	N11	Atlantic, Baltic and Arctic sand beach	SD2, SD5, non-NVC	
3	N13	Atlantic and Baltic shifting coastal dune	SD2, SD4-6	H2110, H2120
3	N15	Atlantic and Baltic coastal dune grassland (grey dune)	SD7-12, SD17, SD19, CG10, non-NVC	H2130
3	N18	Atlantic and Baltic coastal *Empetrum* heath	H11	H2140
3	N19	Atlantic coastal *Calluna* and *Ulex* heath	H11, H10	H2150
3	N1A	Atlantic and Baltic coastal dune scrub	SD12, SD16, SD18, W23	H2170, H2250
3	N1H	Atlantic and Baltic moist and wet dune slack	A10, A11, A13, A16, A22, SD13-17, S4, S19, W1, W2, W4, W6, non-NVC	H2190
2	***N2***	***Coastal shingle***		
3	N21	Atlantic, Baltic and Arctic coastal shingle beach	SD1-3, MC5-6, MC8-9, MG1, CG10, U1, U4, U20, non-NVC	H1210
3	N23	Shingle and gravel beach with scrub	H10, W22-24	H1220
3	N24	Shingle and gravel beach forest	W1, W9, W11, non-NVC	H1220

2	***N3***	***Rock cliffs, ledges and shores, including the supralittoral***		
3	N31	Atlantic and Baltic rocky sea cliff and shore	MC1-3, MC5, MC6, MC7-10, MC12, H7, H8, H10, CG10, MG1, U20, W21-25	H1230
3	N24	Atlantic and Baltic soft sea cliff	non-NVC	H1230
1	**Q**	**Wetlands**		
2	***Q1***	***Raised and blanket mires***		
3	Q11	Raised bog	M1-3, M15-20, M25	H7110, H7120
3	Q12	Blanket bog	M1-3, M15, M17-20, M25	H7130
2	***Q2***	***Valley mires, poor fens and transition mires***		
3	Q21	Oceanic valley mire	M21	
3	Q22	Poor fen	M6-7	
3	Q24	Intermediate fen and soft-water spring mire	M31-33, M35-36	
3	Q25	Non-calcareous quaking mire	M1-2, M4-5, M8-9, S27, M29	H7140, H7150
2	***Q4***	***Base-rich fens and calcareous spring-mires***		
3	Q41	Alkaline, calcareous, carbonate-rich small-sedge spring fen	M9-12, S25, M37-38	H7220, H7230, H7240
3	Q42	Extremely rich moss-sedge fen	NVC correspondence not clear	
3	Q43	Tall-sedge base-rich fen	NVC correspondence not clear	
3	Q44	Calcareous quaking mire	NVC correspondence not clear	
3	Q45	Arctic-alpine rich fen	M10-12, M34	H7240
2	***Q5***	***Helophyte beds***		
3	Q51	Tall-helophyte bed	S4-5, S10, S12-14, S19, S25, S26, S28	H3110, H3130, H3140, H3150, H3160, H3260
3	Q52	Small-helophyte bed	S22-23, A22-24, M29-30	H3110, H3130, H3140, H3150, H3160, H3260
3	Q53	Tall-sedge bed	S2-3, S6-9, S11, S18, S20-21	H3130, H3140, H3150, H3160, H3260
1	**R**	**Grasslands and lands dominated by forbs, mosses or lichens**		
2	***R1***	***Dry grasslands***		

3	R1A	Semi-dry perennial calcareous grassland (meadow steppe)	CG2, CG7, CG10	H6210
3	R1M	Lowland to montane, dry to mesic grassland usually dominated by *Nardus stricta*	U5, CG10, CG11, U2, U4	H6230
3	R1P	Oceanic to subcontinental inland sand grassland on dry acid and neutral soils	U1	
3	R1S	Heavy-metal grassland in western and central Europe	CG10, CG13, OV37, non-NVC	H6130
2	***R2***	***Mesic grasslands***		
3	R21	Mesic permanent pasture of lowlands and mountains	MG5-6, non-NVC	
3	R22	Low and medium altitude hay meadow	MG1-3	H6520
2	***R3***	***Seasonally wet and wet grasslands***		
3	R35	Moist or wet mesotrophic to eutrophic hay meadow	M22-23, MG8-13, OV28	
3	R36	Moist or wet mesotrophic to eutrophic pasture	M22-23, MG8-13, OV28	
3	R37	Temperate and boreal moist or wet oligotrophic grassland	M25-26, U5b-6	H6410
2	***R4***	***Alpine and subalpine grasslands***		
3	R41	Snow-bed vegetation	U11, U13, U14, CG12-14, U18	H6150, H6170, H8110
3	R42	Boreal and arctic acidophilous alpine grassland	U9-10, non-NVC moss and lichen communities	H6150
3	R43	Temperate acidophilous alpine grassland	U7-12, U14	H6150
2	***R5***	***Woodland fringes and clearings and tall forb stands***		
3	R52	Forest fringe of acidic nutrient-poor soils	non-NVC	
3	R54	*Pteridium aquilinum* vegetation	U20, W25	
3	R55	Lowland moist or wet tall-herb and fern fringe	M27-28, non-NVC	
3	R56	Montane to subalpine moist or wet tall-herb and fern fringe	U16, U17, U19	H6430
1	**S**	**Heathland scrub and tundra**		
2	***S2***	***Arctic, alpine and subalpine scrub***		
3	S21	Subarctic and alpine dwarf *Salix* scrub	U12, W20	H6150, H4080
3	S22	Alpine and subalpine ericoid heath	H10, H12, H13-22, CG13, CG14	H4060, H6170, H8240
3	S23	Alpine and subalpine *Juniperus* scrub	H15, W19	
3	S25	Subalpine and subarctic deciduous scrub	non-NVC	

3	S27	Krummholz with conifers other than *Pinus mugo*	W19	
2	***S3***	***Temperate and mediterranean-montane scrub***		
3	S31	Lowland to montane temperate and submediterranean *Juniperus* scrub	W19	H5130, H91C0
3	S32	Temperate *Rubus* scrub	W24-25	
3	S35	Temperate and submediterranean thorn scrub	W21-22	
3	S37	*Corylus avellana* scrub	W9, W11	H9180, H8240
3	S38	Temperate forest clearing scrub	W23	
2	***S4***	***Temperate heathland***		
3	S41	Wet heath	M15, M16, M25	H4010
3	S42	Dry heath	H7-10, 12, H16, H18, H21-22	H4030
2	***S9***	***Riverine and fen scrub***		
3	S92	*Salix* fen scrub	W1-5	
1	**T**	**Forest and other wooded land**		
2	***T1***	***Broadleaved deciduous forests***		
3	T11	Temperate *Salix* and *Populus* riparian forest	W6	H91E0
3	T12	*Alnus glutinosa-Alnus incana* forest on riparian and mineral soils	W2, W5, W6-7	H91E0
3	T15	Broadleaved swamp forest on non-acid peat	W2-3, W5-7	
3	T16	Broadleaved mire forest on acid peat	W2, W4, M17, M18	H91D0
3	T18	*Fagus* forest on acid soils	W15	
3	T1B	Acidophilous *Quercus* forest	W11, W16-17	H91A0
3	T1C	Temperate and boreal mountain *Betula* and *Populus tremula* forest on mineral soils	W10-11, W16-17	H91A0, H91C0
3	T1E	*Carpinus* and *Quercus* mesic deciduous forest	W7-10	H9180, H8240
3	T1H	Broadleaved deciduous plantations of non-site-native trees	none	
2	***T3***	***Coniferous forests***		
3	T35	Temperate continental *Pinus sylvestris* forest	W11, W17-19	H91C0
3	T3J	*Pinus* and *Larix* mire forest	W4, W18, M17-19	H91D0

3	T3M	Coniferous plantation of non-site-native trees	W18, non-NVC exotic conifer plantations	
1	**U**	**Inland habitats with no or little soil and mostly with sparse vegetation**		
2	***U2***	***Screes***		
3	U22	Temperate high-mountain siliceous scree	U18, U21, non-NVC	H8110
3	U26	Temperate high-mountain base-rich scree and moraine	OV38, OV40, non-NVC	H8120
2	***U3***	***Inland cliffs, rock pavements and outcrops***		
3	U31	Boreal and arctic siliceous inland cliff	U18, U21, non-NVC	H8220
3	U35	Boreal and arctic base-rich inland cliff	OV39-40, non-NVC	H8210
3	U3D	Wet inland cliff	U15, non-NVC	
3	U3E	Limestone pavement	OV38-40, CG10, CG13, W9, non-NVC	H8240
2	***U5***	***Miscellaneous inland habitats usually with very sparse or no vegetation***		
3	U51	Fjell field	non-NVC	H8110, H8120

Appendix 2: Development of Web of Science search term

Search term	No of hits in WoS
TS=((tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR eutrophication OR “nutrient enrichment”))	14169
Remove eutrophication term – too broad TS=((tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nutrient enrichment”))	7903
Add “nitrogen fertili” TS=((tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nutrient enrichment” OR “nitrogen fertili*”))	12073
Add NOT agricultur to exclude agricultural grasslands and change to ‘nitrogen enrichment’ to be more precise TS=((tundra OR fell-field* OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nitrogen enrichment” OR “nitrogen fertili*”) NOT agricultur*)	9368
Add mitigation/recovery term TS=((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nitrogen enrichment” OR “nitrogen fertili*”) NOT agricultur* AND (mitigation OR manage* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR “biomass removal” OR “turf stripping” OR “topsoil removal” OR “sod cutting” OR “turf cutting” OR “soil amendment” OR “ nutrient removal” OR “carbo* addition” OR “soil disturbance” OR liming OR “canopy closure” OR thinning))	3392
Add outcomes term TS=((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen	2562

addition” OR “nitrogen pollution” OR “nitrogen enrichment” OR “nitrogen fertili*”) NOT agricultur* AND (mitigation OR manage* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR “biomass removal” OR “turf stripping” OR “topsoil removal” OR “sod cutting” OR “turf cutting” OR “soil amendment” OR “ nutrient removal” OR “carbo* addition” OR “soil disturbance” OR liming OR “canopy closure” OR thinning) AND (biodiversity OR diversity OR richness OR assemblage* OR “functional type” OR “functional group” OR “growth form” OR “species number” OR “species composition” OR “number of species” OR “floristic composition” OR “community composition” OR “habitat suitability” OR “ecosystem function” OR “decomposition” OR “carbon stock*” OR “carbon storage” OR “carbon cycl*” OR “nitrogen cycl*” OR “nutrient stock*” OR “nitrogen stock*” OR “nitrogen budget” OR “nitrogen pool*” OR acid* OR leach* OR product*))	
Refine to remove freshwater and urban habitats TS=((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nitrogen enrichment” OR “nitrogen fertili*”) NOT (agricultur* OR urban OR river or lake OR pond) AND (mitigation OR manage* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR “biomass removal” OR “turf stripping” OR “topsoil removal” OR “sod cutting” OR “turf cutting” OR “soil amendment” OR “ nutrient removal” OR “carbo* addition” OR “soil disturbance” OR liming OR “canopy closure” OR thinning) AND (biodiversity OR diversity OR richness OR assemblage* OR “functional type” OR “functional group” OR “growth form” OR “species number” OR “species composition” OR “number of species” OR “floristic composition” OR “community composition” OR “habitat suitability” OR “ecosystem function” OR “decomposition” OR “carbon stock*” OR “carbon storage” OR “carbon cycl*” OR “nitrogen cycl*” OR “nutrient stock*” OR “nitrogen stock*” OR “nitrogen budget” OR “nitrogen pool*” OR acid* OR leach* OR product*))	2407
Search only in abstract AB=(((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND (“nitrogen deposition” OR “nitrogen addition” OR “nitrogen pollution” OR “nitrogen enrichment” OR “nitrogen fertili*”) AND (mitigation OR manage* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR “biomass removal” OR “turf stripping” OR “topsoil removal” OR “sod cutting” OR “turf cutting” OR “soil amendment” OR “ nutrient removal” OR “carbo* addition” OR “soil disturbance” OR liming OR “canopy closure” OR thinning) AND (biodiversity OR diversity OR richness OR assemblage* OR “functional type” OR “functional group” OR “growth form” OR “species number” OR “species composition” OR “number of species” OR “floristic composition” OR “community composition” OR “habitat suitability” OR “ecosystem function” OR “decomposition” OR “carbon stock*” OR “carbon storage” OR “carbon cycl*” OR “nitrogen cycl*” OR “nutrient stock*” OR “nitrogen stock*” OR “nitrogen budget” OR “nitrogen pool*” OR acid* OR leach* OR product*)) NOT (agricultur* OR urban OR river or lake OR pond))	

Adjust intervention term to ensure that mitigation, management, restoration or recovery is mentioned TS=(((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND ("nitrogen deposition" OR "nitrogen addition" OR "nitrogen pollution" OR "nitrogen enrichment" OR "nitrogen fertili*") AND (mitigation OR manage* OR recover* OR restor*) AND (biodiversity OR diversity OR richness OR assemblage* OR "functional type" OR "functional group" OR "growth form" OR "species number" OR "species composition" OR "number of species" OR "floristic composition" OR "community composition" OR "habitat suitability" OR "ecosystem function" OR "decomposition" OR "carbon stock*" OR "carbon storage" OR "carbon cycl*" OR "nitrogen cycl*" OR "nutrient stock*" OR "nitrogen stock*" OR "nitrogen budget" OR "nitrogen pool*" OR acid* OR leach* OR product*)) NOT (agricultur* OR urban OR river or lake OR pond))	1934
Search only in abstract AB=(((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND ("nitrogen deposition" OR "nitrogen addition" OR "nitrogen pollution" OR "nitrogen enrichment" OR "nitrogen fertili*") AND (mitigation OR manage* OR recover* OR restor*) AND (biodiversity OR diversity OR richness OR assemblage* OR "functional type" OR "functional group" OR "growth form" OR "species number" OR "species composition" OR "number of species" OR "floristic composition" OR "community composition" OR "habitat suitability" OR "ecosystem function" OR "decomposition" OR "carbon stock*" OR "carbon storage" OR "carbon cycl*" OR "nitrogen cycl*" OR "nutrient stock*" OR "nitrogen stock*" OR "nitrogen budget" OR "nitrogen pool*" OR acid* OR leach* OR product*)) NOT (agricultur* OR urban OR river OR lake OR pond))	507
Remove management as too broad but re-add individual management actions TS=(((tundra OR fell-field OR snowbed OR heath* OR moorland* OR peatland* OR bog* OR mire* OR fen* OR spring* OR flush* OR wetland* OR swamp* OR reedbed* OR saltmarsh* OR dune* OR machair OR grassland* OR meadow OR scrub OR woodland* OR forest*) AND ("nitrogen deposition" OR "nitrogen addition" OR "nitrogen pollution" OR "nitrogen enrichment" OR "nitrogen fertili*") AND (mitigate* OR recover* OR restor* OR cutting OR mowing OR burning OR grazing OR "biomass removal" OR "turf stripping" OR "topsoil removal" OR "sod cutting" OR "turf cutting" OR "soil amendment" OR "nutrient removal" OR "carbo* addition" OR "soil disturbance" OR liming OR "canopy closure" OR thinning) AND (biodiversity OR diversity OR richness OR assemblage* OR "functional type" OR "functional group" OR "growth form" OR "species number" OR "species composition" OR "number of species" OR "floristic composition" OR "community composition" OR "habitat suitability" OR "ecosystem function" OR "decomposition" OR "carbon stock*" OR "carbon storage" OR "carbon cycl*" OR "nitrogen cycl*" OR "nutrient stock*" OR "nitrogen stock*" OR "nitrogen budget" OR "nitrogen pool*" OR acid* OR leach* OR producti*)) NOT (agricultur* OR urban OR river OR stream OR lake OR pond))	1593

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