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Can on-site management mitigate nitrogen deposition impacts in non-wooded habitats?

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Abstract

Nitrogen (N) deposition is a major cause of plant biodiversity loss, with serious implications for appropriate management of protected sites. Reducing N emissions is the only long-term solution. However, on-site management has the potential to mitigate some of the adverse effects of N deposition. In this paper we review how management activities such as grazing, cutting, burning, hydrological management and soil disturbance measures can mitigate the negative impacts of N across a range of temperate habitats (acid, calcareous and neutral grasslands, sand dunes and other coastal habitats, heathlands, bogs and fens). The review focuses mainly on European habitats, which have a long history of N deposition, and it excludes forested systems. For each management type we distinguish between actions that improve habitat suitability for plant species of conservation importance, and actions that immobilize N or remove it from the system. For grasslands and heathlands we collate data on the quantity of N removal by each management type. Our findings show that while most activities improve habitat suitability, the majority do little to slow or to reduce the amount of N accumulating in soil pools at current deposition rates. Only heavy cutting/mowing with removal in grasslands, high intensity burns in heathlands and sod cutting remove more N than comes in from deposition under typical management cycles. We conclude by discussing some of the unintended consequences of managing specifically for N impacts, which can include damage to non-target species, alteration of soil processes, loss of the seedbank and loss of soil carbon.

1 Introduction

The deposition of reactive nitrogen (N) has more than doubled over the last one hundred years as a result of agricultural intensification and increased burning of fossil fuels by traffic and industry (Galloway et al. 2008). Globally, deposition of nitrogen is set to increase in the future while in Europe only small declines in N deposition are predicted in the next ten years (Dentener et al. 2006). Therefore it remains a pressing problem. Atmospheric N deposition affects semi-natural habitats through three main mechanisms: eutrophication, acidification and direct toxicity (Bobbink et al. 1998; Jones et al. 2014).

Many studies have reported negative consequences of N deposition on species diversity and ecosystem function in different habitats (e.g. Aber et al., 1989; Bowman et al., 2008; Jones et al.

2004; Stevens et al., 2004) and severe impacts have been observed in some regions of the world. Amongst the most widely recognised examples has been the reduction of heather (*Calluna vulgaris*) cover in Dutch heathlands (Heil and Diemont, 1983). Reductions in plant species richness at high levels of N deposition have been observed across a broad suite of habitats (e.g. Clark and Tilman, 2008; Field et al., 2014; Maskell et al., 2010) together with changes in the composition of plant communities (Bai et al., 2010; Phoenix et al., 2012; Stevens et al., 2011). Nitrogen accumulates in the soil, augmenting soil N pools and altering soil processes. Experimental and gradient studies across the world have shown changes in the concentrations and processing of nitrogen in the soil (e.g. Aber, 2003; Gundersen et al., 1998), nutrient stoichiometry (e.g. Rowe et al., 2008), and leaching to surface waters (e.g. Boxman et al., 1998). Changes in above- and below-ground production and carbon cycling have also been widely observed (e.g. Lee et al., 2010; Reay et al., 2008). These changes affect the provision of a range of ecosystem services such as water quality regulation and greenhouse gas emissions with an associated economic cost (Compton et al., 2011; Jones et al., 2014; Sobota et al., 2015; van Grinsven et al., 2013).

As a result of such widespread impacts on biodiversity, soil processes and ecosystem services there is increasing recognition of the need to manage habitats, and particularly those of a high conservation value, in order to mitigate the effects of N deposition. In the extensive literature on N deposition impacts, there is relatively little focus on how on-site management activities might mitigate N impacts, and few experimental manipulations examining the interaction between N deposition and management. The primary studies that have been conducted (e.g. Britton et al., 2000; Pilkington et al., 2007; Plassmann et al., 2009; Power et al., 2001) have tended to focus on a single habitat and one or a few management options.

The aim of this paper is to draw together much of this disparate evidence to synthesise how on-site habitat management can reduce some of the direct impacts of N on biodiversity, and indirect effects mediated by altered soil N processes and pools of accumulated soil N. In doing so, we review evidence across a wide range of habitats (grasslands, heathlands, coastal habitats, fens and bogs) and management techniques (cutting, grazing, burning, disturbance and other measures). We exclude forested systems, where there is an extensive literature and where management, and therefore removal of N, is more complex. We separately explore impacts on habitat suitability for plant species of conservation interest, and on N removal and cycling. We define habitat suitability as the conditions affecting light, competition and regeneration, while N cycling and removal are separately considered as the conditions affecting biogeochemical cycling of N. We also collate N budget data to quantify N removal by management and we discuss the optimum management measures in the context of managing N deposition impacts. The focus is primarily on semi-natural habitats in the temperate zone, as this is where N deposition has historically been greatest, and where the greatest need for management responses currently lies. However, the findings have implications for other areas around the world where N deposition is increasingly a problem (e.g. Bobbink et al., 2010; Fenn et al., 2010).

2 Methods

The review searched literature databases using web of knowledge and Google Scholar. Keywords for searches were based on habitat and the management techniques using synonyms for both habitats and management techniques (e.g. moorland, heathland; sod cutting, turf cutting, turf stripping). We also searched for grey literature using web searches and databases available on websites of relevant charities and conservation organisations. From the studies identified through literature searches we selected those where nitrogen had been applied in combination with management and which reported management trials that had measured impact on nitrogen pools and/or on habitat suitability.

3 Impacts of pollutant N on ecosystem processes and biodiversity

Understanding the cycle of reactive N within ecosystems (Figure 1) provides insights into appropriate management for reducing effects on biodiversity. Although N is an essential macronutrient needed for plants and animals to grow, natural systems typically have a low rate of N input, from N fixation and the effects of lightning, of the order of 3-5 kg N ha^{-1} yr^{-1} (DeLuca et al., 2008).

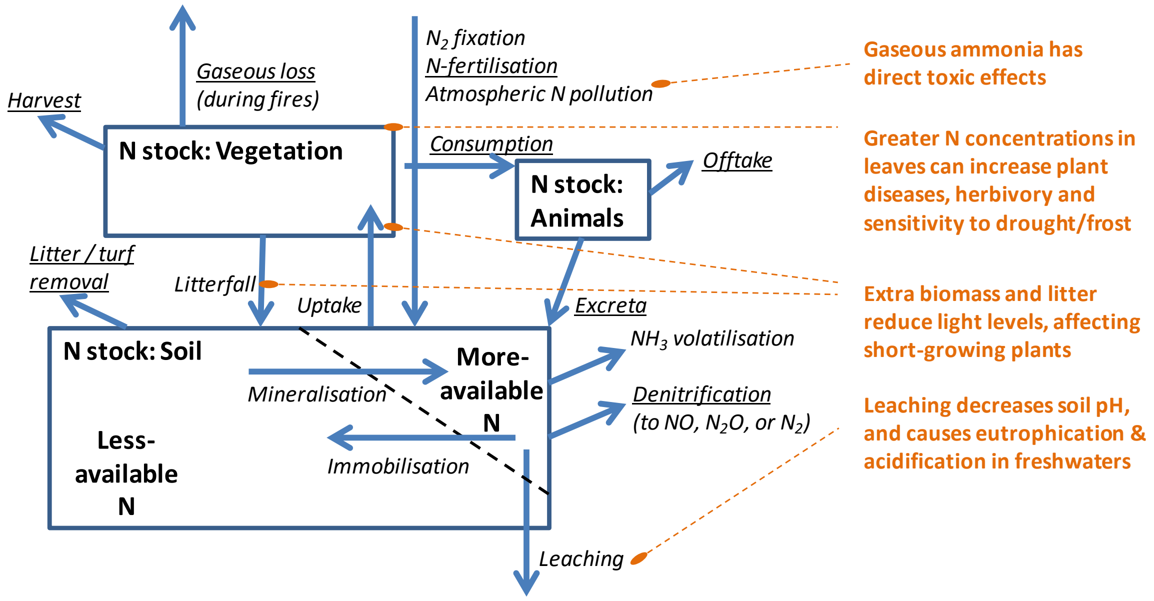

Figure 1. Simplified summary of the nitrogen cycle in terrestrial ecosystems, indicating major effects on species and biodiversity. The arrows represent nitrogen flows. Underlined flows are those that are most readily influenced by on-site management.

Nitrogen pollution can have rapid effects (Hendriks et al., 2014), and in particular gaseous ammonia is toxic for many lichens even at low concentrations of 1 µg m^{-3} (Cape et al., 2009). However, most impacts result from increasing N fluxes into soil and vegetation. In systems that are not N-saturated, the N leaching and other losses are lower than N input rates (Phoenix et al., 2012), so the majority of pollutant N deposited on semi-natural habitats since the onset of industrialization and agricultural intensification has accumulated in the soil. This causes a sustained increase in N mineralization, and the increased availability of N is likely to increase plant growth. Faster growth and greater litterfall rates make conditions difficult for short-growing plants, and these plant species tend to be the more threatened (Hodgson et al., 2014). The same processes also threaten animal species that require warm microclimates (WallisDeVries and Van Swaay, 2006). Increased N availability also tends to increase N concentrations in plant tissue, which can increase susceptibility to insect pests (Lee and Caporn, 1998), and change the structure of foliar invertebrate communities (Rowe et al., 2006).

In systems receiving large and/or prolonged N inputs, eventually the capacity for plant uptake and immobilization is saturated, and N leaching increases. This removes cations such as calcium from the soil, leading to acidification and species loss in poorly buffered habitats, although it should be noted that the dominant cause of soil acidification historically has been sulphur (S) rather than N deposition (Curtis et al., 2005).

4 Overview of management options and how they can mitigate N impacts

A range of management activities are routinely used to manage semi-natural habitats. These may be continuation of traditional management practices such as grazing, or may be implemented as a response to perceived problems affecting conservation status such as scrub encroachment, or loss of forb diversity (e.g. Backshall et al., 2001). Some of these conservation problems may in fact have been caused by, or exacerbated by, N deposition (Dise et al., 2011). Therefore, while many site managers have relatively little awareness of N deposition impacts, existing management may already be responding to the threat and providing some degree of mitigation.

Management has two main effects in the context of N deposition: improving habitat suitability, and removing N from the system. Improving habitat suitability is usually the primary aim of site managers, and is typically achieved by the removal of above-ground vegetation biomass and litter which opens up the canopy and increases light at ground level (Borer et al., 2014). This favours the less-dominant species of plants, and sun-loving animals such as reptiles and many invertebrates. Disturbance associated with some forms of management also provides germination sites for seedling establishment (Isselstein et al., 2002). However, management activities also influence N cycling and N losses from the system. Opening up the canopy can increase daytime soil temperatures, stimulating N mineralisation and therefore the amount of N in the soil available for plant uptake. Mowing and other disturbance events can also transfer living material into litter pools, further increasing N mineralisation rates, and stimulating plant growth in the short-term. Greater N availability may increase N losses due to denitrification and leaching, while export of biomass or soil off-site will also remove N from the system (Verhoeven et al., 1996). These processes have the potential to decrease soil N pools, a key long-term goal of management to reduce N impacts. Information on N losses for the main habitat and management types are summarised in Tables 1 and 2. They are referred to in the text, but are discussed in detail in section 7.

Two issues which affect the potential success of management to reduce N impacts are propagule availability and hysteresis. If propagules of the target species are not in the seed bank anymore because of long term N accumulation (Basto et al., 2015; Plassmann et al., 2008) recovery will not happen unless seed or vegetative material is also reintroduced. Hysteresis is a major issue, where trajectories of biotic community and soil chemistry change in response to management can be drastically different from the development leading to the altered state (Clark and Tilman, 2010; Suding and Hobbs, 2009). 'Natural' recovery in unmanaged systems can take many decades to show signs of recovery from N deposition, if at all (Isbell et al., 2013; Rowe et al., this volume; Stevens, this volume). In managed systems this can be quicker (Storkey et al., 2016), but there are as yet too few studies to adequately characterize the timescales, extent and trajectories of recovery. This review helps to identify the mechanisms and management activities which might best aid recovery.

5 Review of the effectiveness of management options to mitigate N

Conservation management activities applied in temperate systems are grazing, biomass removal including scrub clearance, burning, hydrological management and disturbance (turf stripping, deep ploughing, dune remobilisation). In this section of the paper, for each management option we explore the impacts on habitat suitability and on N cycling and removal, and quantify where possible the amount of N removed by these activities. Two methods which we do not consider, since neither of these are widely practiced to reduce N effects, are the addition of carbon to lock up nitrogen and liming.

5.1 Grazing

Grazing is the dominant management practice in most grasslands with the exception of hay-meadows, but occurs to a lesser extent and at lower intensity in heathlands, bogs and fens. Grazing is a common management tool as it is relatively cheap to implement, is practical over large areas and uneven terrain, and may provide some economic benefit (Rook et al., 2004). Grazing tends to favour graminoids over shrubs since their meristematic tissue is near the soil surface and rarely damaged by grazing, although not in the case of dominance by grazing-sensitive grasses such as *Molinia* or *Deschampsia* where cattle grazing has been shown to favour *Calluna* heath instead (Bokdam and Gleichman, 2000). While there is not space in this paper to provide further detail, the type of grazer plays a large role in altering both habitat suitability and N cycling due to differential grazing/browsing methods, diet selectivity, digestion systems, hoof pressures and dunging behaviour.

5.1.1 Impacts on habitat suitability

The main effect of grazing is to mitigate some of the direct adverse effects of N, by reducing the dominance of fast growing competitive species and opening up the canopy, creating germination niches and aiding local seed dispersal, thus increasing botanical diversity (Hewett, 1985; Plassmann et al., 2010). In acid grasslands, calcareous grasslands and heathlands, experiments combining grazing and N addition have all shown the potential for grazing to mitigate the effects of nitrogen to some extent (Bokdam and Gleichman, 2000; Jacquemyn et al., 2003; Phoenix et al., 2012). The impacts of grazing are likely to depend on vegetation type, stock type, stocking density, timing and duration of grazing period. A long-term N addition experiment on acid grassland with realistic N additions (10 or 20 kg N ha^{-1} yr^{-1}) showed differential N responses under managed grazing at 'light' or 'heavy' grazing pressure (1.87 sheep ha^{-1} or 3.74 sheep ha^{-1}) (Emmett et al., 2004). The dwarf-shrub *Vaccinium myrtillus* declined under the lower grazing intensity treatment but not in the higher grazing intensity treatment, and grazing intensity changed the competitive balance among the lower plants, with light-demanding species benefiting at high grazing intensity (Emmett et al., 2004). These results suggests that heavy grazing may mask the effects of N addition (Phoenix et al., 2012). In montane *Racomitrium* heath there is evidence to suggest that heavy grazing exacerbates the impact of N deposition, where grazing-mediated feedbacks such as increased trampling and faecal enrichment reinforce the negative impacts of nitrogen stimulation of grass and sedge growth (Van der Wal et al., 2003). However, low levels of grazing may benefit *Racomitrium* by opening up the canopy, increasing light levels and the ability of *Racomitrium* to process ammonium, thus reducing ammonium toxicity (Emmett et al., 2004; Jones, 2005). This suggests that the intensity of grazing is important, and that increasing grazing pressure to counteract effects of N deposition may have non-linear effects which are undesirable.

5.1.2 Impacts on N removal and N cycling

The influence of grazing on ecosystem N budgets is generally small (see Table 1). The amount of N removed by grazers depends on their live-weight gain, whether they are taken off site, and whether supplementary feed is provided on-site. Typical net quantities removed in live-weight gain by grazers on unproductive systems range from -2 (i.e. import of N) to +3 kg N $ha^{-1}yr^{-1}$ (Frissel, 1978; Perkins, 1978). Since live-weight gain is low on the poor quality grazing of most semi-natural habitats, and supplementary feed is often provided, grazing livestock can be a net source of N rather than a net sink (van Dobben et al., 2014). A potential exception is the practice of sheep folding, bringing stock off unimproved habitats and putting them on arable land overnight. Since sheep produce dung mainly at night this has been suggested as a technique for removing nutrients from the soil (Gibson, 1995).

Grazing also affects N cycling and leaching, but with variable and contrasting effects. In upland acid grassland, studies have shown higher nitrate leaching in ungrazed compared with grazed areas (Emmett et al., 2004; McGovern et al., 2014), but varying the grazing intensity did not change N cycling or losses (Emmett et al., 2004). In long-term grazing exclosures, McGovern et al. (2014) observed fundamental shifts in ecosystem properties such as C/N ratios and fungal/bacterial ratios, suggesting that complete grazing removal had led to a transition from grasslands with high N turnover which were N-limited to heathlands with slower N turnover which were N-saturated. By contrast, in dune grassland N leaching was 50% lower in ungrazed than in grazed plots (Hall et al., 2011), despite higher mineralisation rates (Ford et al., 2012). Higher mineralization in ungrazed areas were also observed in floodplain meadows (Bakker, 2003). The variability of these findings suggest that the efficacy of grazing as a mitigation measure for N deposition depends on the type of ecosystem, the level of N saturation in relation to other factors limiting plant growth, and the nature and intensity of grazing. It is likely that in systems approaching N saturation, small changes in uptake by plants such as a shift from fast growing grasses to slower growing shrubs in the acid grassland, or differences in the quantity and localized intensity of nutrient return by grazers in dune grassland mean that N inputs exceed biological demand resulting in leaching. What is interesting is that grazing appears to be able to shift the balance in either direction depending on the processes affected. However, despite these variable responses, the net N removal by grazing remains very low (Table 1).

5.2 Cutting with biomass removal

Cutting or mowing is a management tool in hay meadow (neutral) grasslands, heathlands, and fens. It is used at small scale as a conservation tool in dune grassland, calcareous grassland and saltmarsh. Scrub clearance is used to restore under-managed habitats, or where scrub and trees are considered undesirable. Good practice includes removal of biomass from the habitat, since cutting and leaving biomass does not deplete N pools. Leaving litter *in situ* also significantly reduces light levels to the lower canopy, with comparable adverse effects to those of tall vegetation in grasslands and heathlands (Diemont and Linthorst Homan, 1989). In hay meadows and productive grasslands, cutting frequency normally varies from one to three times per year (Pavlů et al., 2011). In less productive systems, or for conservation purposes, cutting may only occur annually.

5.2.1 Impacts on habitat suitability

Cutting removes standing biomass, opening up the canopy and increasing light levels by up to 70 % in grasslands (Borer et al., 2014; Jacquemyn et al., 2003). Cutting also alters the microclimate at ground level, increasing diurnal temperature ranges. In contrast to grazing where the canopy is usually kept short, between cutting events the canopy can regrow to a substantial height and, as a result, impacts on habitat suitability are only temporary. Nonetheless, cutting acts to decrease the abundance of dominant species, giving low-growing or slower growing species the chance to persist. In contrast to grazing there is minimal disturbance to soil layers, and so relatively few new germination niches are created under cutting. The timing and frequency of cutting affect species composition, as they determine which of the component species can set seed under a particular cutting regime (Brys et al., 2004).

The majority of research comes from grasslands. In an experiment investigating timing of cutting combined with N addition at 56 kg N ha^{-1} yr^{-1} in an *Arrhenatherion elatioris* grassland community (Čámská and Skálová, 2012), cutting two weeks earlier than usual countered N impacts by reducing the abundance of tall herbs, but did not increase the abundance of smaller herbs and grasses. In a *Lolio-Cynosuretum* meadow cut twice or four times per year combined with N application levels of 0, 60, 120, 240 kg N ha^{-1} yr^{-1} over 20 years (Pavlů et al., 2011), cutting four times per year reduced the sward height, allowing less nitrophilous grasses to replace previously dominant species. Cutting frequency had a stronger effect on species composition than N. In a P-limited calcareous grassland in Belgium, mowing was as effective as grazing in maintaining light levels and species richness at high N (60-90 kg N ha^{-1} yr^{-1} addition), but less effective at low N (<=30 kg N ha^{-1} yr^{-1} addition) (Jacquemyn et al., 2003). A seven year study using calcareous grassland and acid grassland mesocosms (Jones, 2005) showed that both heavy (6 cm) and light (11 cm) clipping with two cuts per year increased species richness and Simpson’s evenness index in both grasslands compared with the uncut control. However, cutting may only decrease the dominance of nitrophilic species rather than removing them. Bakker et al., (2002) found that species characteristic of eutrophic soil were still present after 25 years of annual cutting for restoration of a *Nardo-Galion* saxatilis grassland. In the 160-year Park Grass experiment, twice annual mowing allowed species richness to recover partially following a decline in background N deposition (Storkey et al., 2016). In heathlands, cutting can rejuvenate the vigour of *Calluna* plants, effectively re-setting the *Calluna* cycle, and considerably opening up the canopy. However, the impact on species composition appears limited. Diemont and Linthorst Homan (1989) found that mowing increased dwarf shrub abundance in *Molinia*-, but not in *Deschampsia*-dominated degraded heathland. In N-recovery experiments, N impacts on plant decomposition were lessened by mowing, but N effects on *Calluna* shoot length, canopy density or height remained (Power et al., 2001). In a degraded, N-impacted poor fen valley mire, Hogg et al. (1995) recorded an increase in *Sphagnum* cover following summer cutting of *Molinia*. In coastal habitats, mowing reduces dominance of tall herbs and perennial grasses, and increases cover of small herbs in dune grassland and saltmarsh (Boorman, 2003; Hewett, 1985). Mowing can also encourage the activity of wild grazers such as rabbits, by reducing the height of the canopy to the level they prefer (Anderson and Romeril, 1992).

5.2.2 Impacts on N removal and N cycling

Cutting has considerable potential to remove N, and other nutrients, from the system, but may not be sufficient to keep pace with N inputs at high deposition rates, or at low rates of biomass removal (see Table 1, Table 2). In a long-running mesocosm experiment, cutting twice per year removed between 20 and 60 kg N ha^{-1} yr^{-1} in calcareous grassland mesocosms while in acid grassland mesocosms cutting twice per year removed 7 - 34 kg N ha^{-1} yr^{-1} depending on the cutting height (Jones, 2005). In heathland, Barker et al. (2004) found that high intensity mowing with litter removal removed 23% of total system N and low intensity mowing 16%, considerably less than the 82% removed by deep burning. In fens, Verhoeven et al. (1996) note that regular mowing has been able to maintain the N balance of managed Dutch fens despite very high N deposition levels. In a managed fen meadow in the Somerset Levels, UK (Lloyd, 2006) the estimated annual hay crop removal of C was around 200 g C m^{-2} yr^{-1}. Assuming a mean C/N ratio of 20 g g^{-1} for biomass this would give an annual N removal of 100 kg N ha^{-1} yr^{-1}, well in excess of deposition. Relatively little is yet known about impacts on mineralisation rates and soil N availability, but Olff et al. (1994) found that long-term hay removal was associated with reduced N mineralisation rates in fen meadows.

Regular mowing tends to remove comparatively large amounts of P and potassium (K). These removals may be in excess of long-term inputs, with the consequence that habitats subject to long-term management tend to be P-limited and therefore less sensitive to N-stimulated plant growth (Verhoeven et al., 1996). However, P removal by mowing alone may take many decades, even where P-mining is stimulated by short term application of nutrients such as N and K to stimulate P uptake and removal (Perring et al., 2009; Schelfhout et al., 2015).

5.3 Burning

Managed burning of vegetation is common in many northwest European habitats including some grasslands, fens, bogs, and particularly heathlands. Upland heaths in the UK are widely burned to encourage new *Calluna vulgaris* growth and create a suitable patchwork of habitats for red grouse *Lagopus scoticus* (Grant et al., 2012). Accidental burning may affect almost any ecosystem. There is evidence of interactions between N and burning. In Californian desert shrublands, elevated N deposition is suggested to increase fire frequency (Rao et al., 2014).

5.3.1 Impacts on habitat suitability

The impacts of burning on N-polluted ecosystems have been most thoroughly investigated in heathlands. Burning allows light to penetrate and increases species diversity in heathlands in the short-term before *Calluna* expands and progressively shades out other species; short rotation burning is therefore recommended to maintain plant diversity (Harris et al., 2011). In a montane heath Britton and Fisher (2007) showed that N had little impact on plant communities of burned plots but considerable impacts on unburned plots, reducing species richness and changing community composition. In a lowland heath, burning did not remove the effect of N on *Calluna* growth and enhanced post-burning establishment of both *Calluna* and *Deschampsia* seedlings (Power et al., 2001; Barker et al., 2004). In a German dry heathland, burning produced conditions more favourable to *Calluna vulgaris* than *Deschampsia flexuosa* (Mohamed et al., 2007).

There has been comparatively little research on burning as an N management approach in other habitats. Burning is generally considered undesirable in bogs as it is likely to lead to loss of C through combustion and promotion of erosion, and to the loss of sensitive species of plants and animals and their replacement by burning-tolerant taxa (Worrall et al., 2010). Burning of blanket bog can lead to elevated nitrate concentrations in both porewaters and drainage waters (Cresser et al., 2004; Helliwell et al., 2007). In sand dunes there is some research showing that fire may improve the structure and composition of vegetation (Rhind and Sandison, 1999) but there is also evidence that invasive moss species such as *Ceratodon purpureus* and *Campylopus introflexus* can become dominant after burning (Ketner Oostra et al., 2006).

5.3.2 Impacts on N removal and N cycling

Plant material and litter comprise an important ecosystem-level pool of N and therefore significant N stocks can be removed by burning (see Table 2). In an upland heath in north Wales, UK, Pilkington et al. (2007) found that burning removed a large proportion of accumulated N from the vegetation and litter but did not lead to any short term reduction in N stores in the soil. In Germany, burning removed N pools in the vegetation, but returned 5.2 kg ha^{-1} as ash (Härdtle et al., 2009). Beyond the instantaneous removal of N to the atmosphere burning may also enhance N leaching for several years subsequently (Mohamed et al., 2007; Pilkington et al., 2007). These combined losses may be sufficient to mitigate the impacts of N deposition in moderately polluted heathlands but in heavily polluted sites the required burning frequency may be unrealistically high (Pilkington et al., 2007). It is important to recognise the importance of the age and growth phase of the *Calluna* canopy when burnt; *Calluna* is most productive in the mature phase (~15 years, Gimingham, 1972), therefore very short, or very long management cycles are likely to be less effective in their relative removal of accumulated N. An important disadvantage to burning as a mitigation option is the displacement of the N lost from the burn-site to nearby habitats through re-deposition of N oxides, and consequently elevated N fluxes in water-courses (Cresser et al., 2004). As with other interventions, the effectiveness of burning as an N mitigation approach will depend on the details of implementation. For instance in fens, burning generally takes place in winter when water levels are high (Middleton et al., 2006) but at these times, most plant N is likely to be stored below-ground so burning may remove less N than summer mowing.

5.4 Hydrological management

Hydrological management is relevant to wetland ecosystems such as bogs (acidic) and fens (alkaline) which have been affected by historical drainage. Wetland drainage was often accompanied by conversion to intensive agriculture or forestry, and some of these areas are now being re-wetted for conservation purposes. Nutrient enrichment issues in these sites are likely to be associated with previous agricultural fertilisation with multiple nutrients, rather than atmospheric N deposition, although continued N deposition may inhibit their restoration to semi-natural, low-nutrient conditions. On areas that were drained but not fertilised, including extensive areas of upland blanket bog, N deposition may have a more significant impact.

Fens and other wetlands which receive much of their water supply from sources such as groundwater and surface water, may become enriched with N and other nutrients from agriculture or other human activities (Rhymes et al., 2014; Rhymes et al., 2015). In these ecosystems, N inputs in surface flow and groundwater must be considered alongside atmospheric N pollution, and management activities designed to mitigate against this overall load.

5.4.1 Impacts on habitat suitability

Ombrotrophic bogs receive all of their water and nutrient inputs directly from the atmosphere, and are thus adapted to very low nutrient conditions. Drainage increases N mineralisation rates, with similar impacts to N deposition. Therefore re-wetting would be expected to favour the re-establishment of wet, low-nutrient adapted *Sphagnum* species and other rare bog species. This was demonstrated for example in Northern Scotland by Bellamy et al. (2012), who showed a progressive shift towards bog indicator species (and away from heathland species) in the years following re-wetting of a blanket bog. In more polluted areas such as The Netherlands and the English Pennines, however, many key peat-forming species were largely eradicated by high rates of S and N deposition (e.g. Lee and Studholme, 1992), and recovery may be slow and incomplete due to continuing high N levels in peat and pore waters (Caporn et al., 2006). Some charismatic bog plant species such as *Drosera* spp. and *Narthecium ossifragum* show negative relationships with N (Field et al., 2014). Therefore, in N polluted areas re-wetting can still improve habitat suitability for bog species, but the endpoint is likely to differ from that in more pristine regions.

In fens, water management is often more active (e.g. controlled via sluices or pumps) and more complex (e.g. increasing or decreasing hydrological connectivity with adjacent mineral soils, rivers or groundwater inputs). Drainage of fens can reduce lateral input of nutrient-enriched water, one result of which is that deposition may become proportionally more important for overall N supply. Lowered water tables can also increase N mineralisation rates, leading to ‘internal eutrophication’ (potentially representing a delayed impact of past N enrichment). However, the hydrological isolation of drained fens can also lead to potassium (K) limitation (Venterink et al., 2009), and management-related biomass removal can instigate either P or K limitation, allowing rare species to persist under elevated N inputs (Wassen et al., 2005). Rewetting and hydrological reconnection are likely to be prerequisites for the restoration of these degraded habitats, but are unlikely to lead to ecological recovery without concurrent removal of nutrient-enriched biomass and litter, or the control of lateral nutrient inputs from nearby agricultural land.

5.4.2 Impacts on N removal and N cycling

Re-wetting of bogs and fens will significantly alter N cycling, although effects on ecosystem N stocks are likely to be gradual. Raising water tables allows the re-establishment of anaerobic conditions close to the wetland surface, reducing decomposition rates and hence N mineralisation. Surface water observations indicate that peatlands in regions subject to a combination of both high N deposition and management disturbance can leach mineral N, whereas this is rarely observed in undrained (and unburnt) bogs even in areas of fairly high N deposition (Helliwell et al., 2007). If peat formation can be re-established via re-wetting, reactive N from the atmosphere can be effectively locked up in stable organic matter, reducing internal N availability and reducing or halting mineral N leaching. Therefore, provided that levels of N deposition to the bog surface are below the retention capacity of *Sphagnum* (e.g. Lamers et al., 2000), re-wetting can be expected to facilitate the restoration of N-impacted bogs. However, there are concerns that rewetting shows hysteresis effects and bogs do not return to their pre-impacted state (Schimelpfenig et al., 2014).

5.5 Sod cutting and other soil disturbance measures

Sod cutting, also called turf-stripping, and a range of soil disturbance measures are used in some grassland, heathland, fen and sand dune habitats. They are used most widely in heathlands where many lowland heaths owe their origin to turf removal (Webb, 1998), and more recently in sand dunes. A key determinant of efficacy is the depth of cutting with very deep cutting generally more costly and less effective in terms of habitat restoration, due for instance to modified soil moisture levels (Diemont and Linthorst Homan, 1989; Niemeyer et al., 2007). In ex-arable areas and sand dunes, a restoration technique called deep-ploughing, or topsoil inversion, has been applied (Glen et al., 2007; Jones et al., 2010), where the whole soil profile is inverted to a depth of 1 m. A similar but less intensive management intervention is soil disturbance by ploughing or shallow rotavation where the soil layers are mixed but not removed. Soil disturbance is traditional management in some semi-natural habitats (e.g. within arable cultivation areas in machair) and may minimise some of the disadvantages of sod cutting such as high cost and disposal of waste material while achieving many of the same effects (Britton et al., 2000). In sand dunes a range of disturbance techniques (topsoil inversion, turf stripping, dune reprofiling) have been trialled to reactivate natural dynamics of dune movement and restore early successional conditions (Arens and Geelen, 2006; Rhind et al., 2013).

5.5.1 Impacts on habitat suitability

Sod cutting removes vegetation and humic material which may include desirable as well as undesirable species and is also likely to remove much of the near-surface seed bank (Dorland et al., 2005). This may have positive effects, favouring species with a persistent seed-bank at depth which in heathlands may favour *Calluna* over invasive graminoids such as *Molinia caerulea* and *Deschampsia flexuosa*. However there may also be negative impacts, particularly the removal of seeds of species with high conservation value (e.g. van den Berg et al., 2003). In heathlands, the humus layer also regulates soil moisture and water infiltration, so removing it may have negative effects (Bijlsma et al., 2013). Soil disturbance by rotavation or ploughing has similar advantages and disadvantages but with generally lower efficacy. A Dutch experiment showed such treatments to be ineffective at returning degraded grass-dominated heath to dwarf-shrub domination although overall productivity was reduced (Diemont and Linthorst Homan, 1989; Diemont, 1994). As many species of high conservation value are likely to have few local sources of propagules and may have limited dispersal ability, direct planting combined with weed control may be required to avoid dominance by common ruderal species. Sod cutting results in large-scale disruption of the soil ecosystem. Vergeer et al. (2006) found that most arbuscular mycorrhiza spores were removed by cutting and numbers were still reduced after 2.5 years. Such disruption of the mycorrhizosphere may limit the ability of many plant species to re-colonise. In dune habitats, the re-creation of bare sand and the increase in soil pH by sand mixing or removing decalcified surface sand by large-scale disturbance favour early successional colonizers of high conservation value (Houston and Dargie, 2010). Shallow sod cutting in older dune grasslands led to rapid rejuvenation of characteristic dune plant and invertebrate species (van Til and Kooijman, 2007).

5.5.2 Impacts on N removal and N cycling

As a large proportion of deposited N is retained in vegetation and in the upper layers of organo-mineral soils, it is clear that sod cutting is a very effective means to remove N from the system and may reduce N mineralisation rates and plant productivity for considerable periods of time (Berendse, 1990; Diemont, 1994; Heil and Bobbink, 1993) (see Table 1, Table 2). In one lowland heath the nutrient removal by sod cutting was equivalent to roughly 176 years of atmospheric deposition (Härdtle et al., 2006) which is likely to be much greater than removal by less-intensive management options (Härdtle et al., 2006; 2007). With removal of the oxidised surface layers, the form of available N shifts to ammonium (Troelstra et al., 1997). However, short-term increases in ammonium concentrations were suggested to inhibit recolonization of target species in Dutch heathlands, but this could be alleviated by liming (Dorland et al., 2004). In contrast to sod cutting, soil disturbance without removal generally has little or no impact on soil N stock, although in habitats with a thin organic layer such as sand dunes the nutrient-enriched upper soil may be sufficiently buried as to become inaccessible to plants (Jones et al., 2010). However, where large quantities of soil nutrients remain on-site, even buried, there is potential for them to be re-exposed or accessible to deep-rooting species, negating any short-term benefit of management (Rhind et al., 2013). Soil disturbance is likely to increase N and C mineralisation through aeration (Balesdent et al., 2000) but may lead to soil erosion (Alonso et al., 2012). Soil disturbance or removal may also modify the hydrological regime through soil compaction, reduced infiltration and surface lowering (Andrew and Lange, 1986; Jansen and Roelofs, 1996; Kahlon et al., 2013). In a *Cirsio-Molinietum* grassland sod cutting led to inundation of the soils creating anaerobic conditions with no nitrification, and higher nutrient availability in the rooting zone (Jansen and Roelofs, 1996). Sod cutting may lead to a reduction in buffering capacity in some habitats which may require active intervention through liming to allow the re-establishment of sensitive plants (Dorland et al., 2005; van den Berg, 2003). By contrast, in dune slack habitats and in fens, surface lowering can be beneficial by ‘chasing the watertable’ thereby increasing buffering capacity from more frequent contact with base-rich groundwater, necessary for basiphilous species of high conservation value (Emsens et al., 2015; Grootjans et al., 2004).

6 Data synthesis of management impacts on N budgets in grasslands and heathlands

For grasslands, Table 1 summarises the quantities of N removed by different management practices, and calculates net N accumulation under N loads from atmospheric deposition in those studies, which span the typical range of N deposition in Europe. Of the uncontrolled N removals (losses from gaseous emissions and leaching), gaseous emissions remove relatively little N, with a maximum of 2 kg N ha^{-1} yr^{-1} reported using cutting (Jones, 2005). Leaching can remove larger quantities of N, depending on the habitat characteristics. In sandy dune grassland soils with relatively low binding capacity, leaching fluxes can be up to 4 kg N ha^{-1} yr^{-1}, while in N-saturated acid grassland, leaching fluxes were as high as 10 kg N ha^{-1} yr^{-1} (Emmett et al., 2004), nearly 50% of atmospheric inputs. However, in systems which are not N-saturated, leaching losses are likely to be much lower. Of the controlled N removals, grazing removes relatively little N, typically under 1 kg N ha^{-1} yr^{-1}, unless folding takes place in which case the estimate may be as high as 8-18 kg N ha^{-1} yr^{-1}. Conversely, winter stock feeding can result in considerable N input to the site, as high as 6.3 – 9 kg N ha^{-1} yr^{-1} in habitats where cattle spend most of their time (Van Dobben et al., 2014). Of the annual measures, cutting removes the most N, ranging from 11 - 70 kg N ha^{-1} yr^{-1} in unimproved grasslands, depending on the severity of the cut. For comparison, in improved leys cutting can remove 100 – 280 kg N ha^{-1} yr^{-1}, but note these are under higher N inputs of 50 – 230 kg N ha^{-1} yr^{-1} including biological N fixation. The greatest quantity of N removed is by sod cutting, which removes 1,300 – 4,700 kg N ha^{-1} yr^{-1}, equivalent to 64-235 years of N deposition. However, this is practiced infrequently in grasslands, primarily to revert dune grasslands to an earlier successional stage.

When the net N accumulation was calculated under each management option (Table 1), in the absence of active management all grassland studies showed a net accumulation of N ranging from 7 – 24 kg N ha^{-1} yr^{-1}. Under managed grazing there was still a net accumulation of N, with the exception of the higher estimate for N removal in sand dune grasslands where stock folding is practiced. Generally, cutting with biomass removal is the only routine management option leading to a net export of N from the system, ranging from an annual removal of 1 – 56 kg N ha^{-1} yr^{-1} under a heavy cutting regime (low cutting height, or multiple cuts in a year). Note that under a light cutting regime (higher cutting height, or only one cut per year) there is still the potential for net N accumulation. More extreme management options such as sod cutting and scrub clearance can remove 1288 to 4704 kg N ha^{-1}, equivalent to 64 to 235 years N deposition. Their efficacy for N removal therefore depends on the frequency with which they are undertaken.

For heathland systems Table 2 summarises the quantities of N removed and the implications for net N accumulation. Since heathland management activities are typically episodic rather than annual, repeated on multi-year cycles, the summaries are expressed as years of N deposition removed (Theoretical Effective Period *sensu* Härdtle et al. (2006)). Low intensity burns, typical of prescribed burning remove the equivalent of 2 – 9 years of atmospheric N deposition. This is broadly equivalent to that removed by cutting with biomass removal (2 – 13 years of N deposition). Härdtle et al. (2009) show that on a 10 year cycle, both burning and mowing still result in a net accumulation of ~10 kg N ha^{-1} yr^{-1}. By contrast, high intensity burns equivalent to wildfires remove much more N, equivalent to 47 years of N deposition. The more extreme management options of sod cutting remove up to 10 times as much N as low intensity burns or cutting, equivalent to 61 – 90 years of N deposition, which can remove significant quantities of N when applied on a 15 or 30-year cycle (Härdtle et al., 2006; 2009). Grazing is occasionally used to manage heathlands, and if applied as sheep folding, can remove up to 8.2 kg N ha^{-1} yr^{-1} (data calculated from Fottner et al., 2007), but this still leaves a net accumulation of 12.4 kg N ha^{-1} yr^{-1}. The choice of management practice to remove N may depend on the severity of the eutrophication problem and other constraints on what is acceptable or practical, together with consideration of the possible unintended consequences (see section 8).

Table 1. Nitrogen budget for **grasslands** under different management regimes, showing the main N loss pathways, and net annual N accumulation (all data as kg N ha^{-1} yr^{-1}, unless specified). Negative numbers represent N removal. Light cut represents a moderate cutting height (11-15 cm). Heavy cut represents a short cutting height (5-6 cm), or multiple cuts in a year. Burning excluded due to lack of N budget data.

Habitat (and management)	Literature source	N deposition	**Uncontrolled removals**		**Controlled removals**				**Net annual N accum (Deposition minus Controlled & Uncontrolled removals)**				
			Leaching	Gaseous emisssions (Ammonia + denitrification)	Grazing (in animal products)	Light cut	Heavy cut	Turf-stripping/ Scrub clearance	No active manage-ment	Grazing	Light cut	Heavy cut	Turf-stripping/ Scrub clearance[k]
Grassland (grazing)													
Acid grassland	Emmett et al., 2004	22	10	0.3	0.5-1				11.7	10.7			
Sand dune grassland	Hall et al., 2011; Plassmann et al., 2009; Jones et al., 2013	11.5	4	0	(1) [a]				7.5	6.5			
Sand dune grassland	Van den Berg et al., 2014	>20	(4) [b]		8 -18 [d]				16	8 to -2			
Sand dune grassland	van Dobben et al., 2014	16.5	(3.3) [b]		0				13.2	19.5 to 22 [e]			
Grassland (cutting)													
Acid grassland (mesocosms)	Jones, 2005	27	0.7	2		11	25.3		24.3		13.3	-1	
Acid grassland (mesocosms)	Phoenix et al., 2003	26	(0.7) [c]	(2) [c]			70		23.3			-46.7	
Calcareous grassland (mesocosms)	Jones, 2005	27	7.2	2		25	53.9		17.8		-7.2	-36.1	
Calcareous grassland (mesocosms)	Phoenix et al., 2003	26	(7.2) [c]	(2) [c]			25.7		16.8			-8.9	
Chalk grassland	Wells and Cox, 1993	?					26						
Sand dune grassland	Hall et al., 2011; Plassmann et al., 2009; Jones et al., 2013	11.5	2 [f]				65		9.5			-55.5	
Sand dune grassland	Van den Berg et al., 2014	>20	(3.6) [b]				32-35		16.4			-15.6 to -18.6	

Lolium clover ley, (Year1: 3x cut, Year2: 12x cut)	Vinther, 2006	20 + (214, 38 N fix) [g]	Not known	Not known	287, 103		234 to 58 [h]	-63, -65	
Improved Lolium grassland, (2x, 4x cut)	Pavlu et al., 2011	Not known	Not known	Not known	160, 200				
Grassland (turf-stripping)									
Sand dune grassland	Van den Berg et al., 2014	20 [i]	Not known	Not known		1288 to 4704 [j]			64 to 235 yrs [k]
Grassland (scrub clearance)									
Sand dune grassland	Van den Berg et al., 2014	20 [i]	Not known	Not known		154 to 210			8 to 11 yrs [k]

a Assumed max 1 kg N ha^{-1} yr^{-1} removed in animal products after Emmett et al. (2004);

b No data provided; estimated from ~20% loss as fraction of inputs, in Hall et al. (2011);

c Data provided but unrealistic due to rapid mineralisation in small mesocosms; data here estimated from Jones (2005);

d Extensive grazing or mob grazing, assumes 'folding' i.e. Removal of stock off-site at night.

e Includes N inputs of 6.3 – 9 kg N ha^{-1} yr^{-1} from supplementary feeding (van Dobben et al., 2014)

f Ungrazed grassland, estimated from single biomass cut

g Estimated background N deposition; showing extremes from treatments: Max N fixation in 1st year, reduced N fixation in 2nd year.

h Including N fixation. Leaching and gaseous uncontrolled losses not known

i Deposition not given, assumed at least 20 kg N ha^{-1} yr^{-1}

j Turf stripping/sod-cutting to 10 cm

k Due to episodic nature of this management type, N removal is expressed in years of N deposition equivalent removed.

Table 2. Nitrogen budget for **heathlands** under different management regimes, showing N removals (all as kg N ha^{-1} yr^{-1}) and Theoretical Effective Period (years of N deposition removal) for each management practice.

Habitat	Literature source	N deposition	Instantaneous removal (+loss in enhanced leaching per year [a]).					Theoretical Effective Period (Years of N deposition removal) [b]				
			Low intensity (managed) burn	High intensity burn (wildfire)	Mowing with removal	Restoration of degraded heath (biomass removal including litter) [c]	Sod cutting	Low intensity (managed) burn	High intensity burn (wildfire)	Mowing with removal	Restoration of degraded heath (biomass removal including litter) [c]	Sod cutting
Lowland heath												
Thursley Common, England	Power et al., 2001; Barker et al., 2004	8	68 [d]	373 [d]	105 [de] 73 [df]			9	47	13 9		
Budworth Common, England	Ray, 2007	20.9			72 (+1 [r])					2-4		
Lüneburg Heath, Germany	Hardtle et al., 2006; 2009	22.8	99 [h] (+1[g])		97 [hj] (+1 [g])		1683 [hi] (+2 [g])	5		5		90
	Mohamed et al., 2007	22.8	109 [l] (+3 [m])					5				
	Niemeyer et al., 2005	22.8	91 (+3 [g]) to 99 (+2 [g])					5				
	Niemeyer et al., 2007	22.8					988 [k] (+2 [g])					61
	Niemeyer et al., 2007	22.8					1679 [i] (+3 [g])					90
Diepholzer Moorniederung, Germany	Hardtle et al., 2007	33.1	45 to 65					2-4				
Upland heath												
Ruabon, Wales	Pilkington, 2003; Pilkington et al., 2007	16.4	131 (+4) [q]					5-8				
Degraded lowland heath												
Arne (Dorset)	Mitchell et al., 2000	13.5				562-2662 [n]					N/A	
Blackhill (Dorset)	Mitchell et al., 2000	13.5 [p]				1212 [n]					N/A	
Cranbourne (Dorset)	Mitchell et al., 2000	13.5 [p]				870 [n]					N/A	

All figures except N deposition rounded to whole numbers.

a Gaseous losses are considered minimal in heathlands (Power et al., 1998) and are therefore not included in calculations. Therefore, the uncontrolled losses are restricted to leaching losses.

b Estimate of Theoretical Effective Period (*sensu* Härdtle et al., 2006) based on figures from original papers or calculations from the available data.

c Treatments in this study are intensive restoration involving removal of large quantities of plant material from very degraded sites. These treatments are not comparable to 'routine management' in other studies. Results are presented for several degraded habitat types but only the range is presented here.

d Calculated using percentage losses from Barker et al. (2004, Table 3) and stocks from Power et al. (1998, Table 1). High intensity burn involved removal of plant material, litter and humus off-site followed by combustion and replacement of ashes. Low-intensity burn was conducted in-situ. Leaching losses considered negligible in this study (Power et al., 1998).

e High intensity mowing- ground level

f Low intensity mowing- 10cm height

g Biomass removal and leaching data annualised based on data presented over 5, 10 or 15 year interval.

h Non-leaching N outputs over 5 (burning/mowing), or 15 (sod cutting) year frequency calculated from above-ground biomass loss, organic and A-horizon loss while accounting for N returned in ash (Hardtle et al., 2006; Table 2).

i Sod cutting treatment based on removal of all organic layer and part of A-horizon.

j Mowing at 10cm height.

k Cutting based on removal of above-ground vegetation and part of O-horizon.

l Based on loss from biomass combined with ash input to O-horizon.

m Annual loss over first year post-burn.

n N in litter and aboveground biomass.

p Not given, assumed to be comparable to Arne heath in Dorset.

q Plots around 20 years old when managed. Leachate losses measured in gley horizon and extrapolated to annual figure from 6 months pre and post-burn.

r Plots around 9 years old when managed. Removal values shown represent the main vascular plant canopy. Leachate losses 12 months pre and post-burn.

7 Unintended consequences of management options

All of the management options reviewed have the potential to affect habitats in other, unintended ways. Potential consequences of management actions for non-target processes and organisms are summarised in Table 3. Increasing grazing intensity may cause the loss of plant species less tolerant of grazing and a corresponding increase in grazing tolerant species (Del-Val and Crawley, 2005). Overgrazing can therefore result in reduced species richness and diversity and may favour species that are less typical of the community. If supplemental feed is imported to maintain high levels of grazing during winter months this can be an additional source of nutrients (Van Dobben et al., 2014) and a seed source of undesirable plant species. Burning may similarly promote some species over others (Mallik and Gimmingham, 1983), typically favouring nitrophiles such as the moss *Campylopus introflexus* (Ketner Oostra et al., 2006). In North American tall-grass prairies, over-frequent burning can reduce species diversity (Collins et al., 1998). The species composition of hay meadows is particularly sensitive to the time of cutting (Brys et al., 2004). Management is also likely to affect taxa other than plants; grazing or cutting may have a negative impact on ground nesting birds (Vickery et al., 2001). Inevitably there are trade-offs in conservation management and the optimum management choice depends on balancing the requirements to remove N with potential consequences.

For those management measures targeted at the removal of N, attention should be paid to the fate of the removed N to ensure that it does not have detrimental ecological consequences. For example, burning removes N from vegetation and litter, but much of this N will be converted to nitrogen oxides and deposited elsewhere. Burning may also lead to an increase in nitrate leaching to surface waters for several years (Cresser et al., 2004), and trigger the loss of soil carbon both directly through combustion losses and indirectly through increases in erosion (Farage et al., 2008). Burning is particularly damaging in peatland systems after drainage or during dry periods, when the soil itself can ignite. Management measures may also have implications for soil structure, soil chemistry and hydrological regimes, for example compaction from grazing animals (Sharrow, 2007). Intense interventions such as sod cutting are likely to have abrupt consequences for many taxa, and should only be used when there is evidence that the resulting surface will support target species, or that they can recolonise easily. Jansen et al. (2004) advise caution applying sod cutting and soil-removal measures due to their impact on seed banks.

The frequency and intensity of management are key to balancing benefits against adverse impacts. Under conditions of high N inputs, more frequent and/or intense management is required to counter the effects of N, e.g. to keep pace with faster growth of nitrophiles or to increase N removal and prevent additional accumulation of N in soil or plant pools. Therefore, the adverse effects of increased frequency or intensity of existing management methods should also be taken into account.

Table 3. Potential for management methods to mitigate N impacts on habitat suitability or N removal, by habitat. Additional columns show strength of evidence and unintended consequences for each option. Options with high potential to improve habitat suitability and to remove N are highlighted in bold.

Broad habitat	Management method	Potential to mitigate N impacts on habitat suitability	Strength of evidence[1]	Potential to immobilise or remove N	Strength of evidence[1]	Unintended consequences[2]
Neutral grassland	Grazing	High	#	Low	(#)	NTS,FS,(So)
	Cutting	**High**	**#**	**High**	**##**	**NTS,SB**
	Hydrological management	Medium	(#)	Medium	(#)	NTS,So
	Sod cutting	Low	(#)	High	#	NTS,FS,So,C,SB
Calcareous grassland	Grazing	High	#	Low	(#)	NTS,FS,(So)
	Cutting	**High**	**#**	**High**	**#**	**NTS,SB**
	Sheep folding	High	(#)	Medium	(#)	NTS,FS,(So)
Acid grasslands	Grazing	High	##	Low	#	NTS,FS,(So)
	Burning	Low	(#)	High	(#)	NTS,FS,So,C,(SB)
	Cutting	Medium	#	High	#	NTS,SB
Dwarf shrub heath	**Sod cutting**	**High**	**#**	**High**	**#**	**NTS,FS,So,C,SB**
	Rotavating	Medium	#	Low	(#)	NTS,FS,So,C,SB
	Grazing	Medium	#	Low	(#)	NTS,FS,(So)
	Cutting	**High**	**##**	**High**	**##**	**NTS,SB**
	Burning	**High**	**##**	**High**	**#**	**NTS,FS,So,C,(SB)**
Fen, marsh and swamp	Grazing	Low	#	Low	(#)	NTS,FS,(So)
	Cutting	**High**	**##**	**High**	**##**	**NTS,SB**
	Burning	Medium	(#)	High	(#)	NTS,FS,So,C,(SB)
	Hydrological management	Medium	(#)	Medium	(#)	NTS,So
	Sod cutting	Medium	(#)	High	#	NTS,FS,So,C,SB
Bogs	Hydrological management	Medium	(#)	Medium	(#)	NTS,So
	Burning	Low	(#)	High	(#)	NTS,FS,So,C,(SB)
Coastal dune grassland and dune slacks	Grazing	High	##	Low	#	NTS,FS,(So)
	Cutting	Low	##	High	#	NTS,SB
	Burning	Medium	(#)	High	(#)	NTS,FS,So,C,(SB)
	Hydrological management	Medium	#	Medium	(#)	NTS,So
	Sod cutting	**High**	**#**	**High**	**#**	**NTS,FS,So,C,SB**
	Dune remobilisation	High	#	Low	(#)	NTS,FS,So,C,SB
Other coastal habitats (saltmarsh, shingle, cliffs)	Grazing	Medium	#	Low	(#)	NTS,FS,(So)
	Cutting	Medium	(#)	High	(#)	NTS,SB

[1] Strength of evidence score: ## -Numerous, consistent studies; # -A few, or inconsistent studies, (#) -Expert judgement.

[2] Codes for unintended consequences: NTS - impacts on non-target species; FS - damage to fragile species; So - major change in soil processes; C – loss of soil carbon; SB - loss of seedbank.

8 Conclusions

Site management is undertaken to maintain a desired habitat condition, usually defined by conservation objectives for the site. An example for UK acid grassland would be to maintain sward height between 10 and 50 cm (JNCC, 2004). Evidence collated here has shown that the management options reviewed are beneficial for habitat quality, but most do little to remove N, and all can have unintended consequences. The benefits and weaknesses of each practice in terms of improving habitat suitability, removing N and the unintended consequences are summarised in Table 3. Outcomes for the most commonly applied management techniques are: Grazing at the appropriate intensity for the habitat improves habitat suitability but removes very little N, and winter stock feeding can result in a net N import to the site (van Dobben et al., 2014). Overgrazing can have adverse impacts on habitat suitability, and may exacerbate N impacts (Van der Wal et al., 2003). Burning is only used in certain habitats; while it can remove considerable N, it can also have many negative impacts. Achieving the correct frequency of burning is critical to balance beneficial and adverse effects. Cutting is the most efficient technique for removing N, provided cut biomass is taken off-site, within a period of a week (Schaffers et al., 1998), but its impacts on habitat suitability are transient and cutting needs to be repeated to maintain habitat quality. For grasslands and some wetlands, in-combination management such as cutting followed by after-math grazing (Walker et al., 2004) may provide the most overall benefit for both habitat suitability and N removal. Disturbance measures such as sod cutting or dune remobilization can be very efficient at removing N but are fairly drastic and only suitable in certain habitats. Their use in some dune habitats is increasing, where the re-instatement of natural dynamics can achieve multiple aims, combating impacts of N deposition and increasing resilience to climate change.

There is a trade-off between the amount of disturbance and the amount of N removed. Removing large quantities of N requires more intensive management, usually with machinery, and more disturbance. Adaptive management can be used to scale the degree of intervention to the severity of the N problem, or the stage of recovery. In a heavily degraded system, intensive intervention such as sod cutting may be needed to initiate a trajectory towards recovery at first, followed up by lighter-touch options such as grazing to maintain habitat suitability once sufficient N has been removed from the system.

It will always be preferable if atmospheric N inputs can be reduced rather than attempting to mitigate their effects. Some reductions in N inputs can be achieved using tree belts where these can be situated to intercept ammonia, either around the point of emission or near the boundary of the conservation site (Dragosits et al., 2006). However, reducing atmospheric N inputs depends to a large extent on controlling emission sources at local and wider scales. Where such measures are beyond the power of site managers, or likely to be implemented only slowly, the site management measures discussed above can mitigate to some extent the adverse impacts of N pollution.

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