



Short communication

Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland

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ARTICLE INFO

Article history:

Received 15 July 2011

Received in revised form

14 December 2011

Accepted 19 December 2011

Available online 28 January 2012

Keywords:

Moorland

Grass-dominated ecosystems

Carbon

Nitrogen

Soil properties

Calluna vulgaris

Ecosystem services

Uplands

Grazing management

Yorkshire Dales

ABSTRACT

We evaluated the impact of 7 years of grazing exclusion on vegetation and belowground properties related to soil carbon (C) and nitrogen (N) cycling in grazed, upland grassland in northern England. For this, we compared a landscape-level, moorland restoration project (grazing exclusion) with adjacent continuously grazed acidic grasslands to test whether changes in vegetation composition after restoration impacted on soil properties including soil C storage. Grazing exclusion significantly increased the proportion of dwarf-shrubs at the expense of graminoids. Despite high seasonal variability, this change in vegetation was associated with increased plant litter mass, soil moisture content and the ratio of dissolved organic to inorganic N, and reductions in rates of ammonium mineralisation, soil microbial activity, and microbial biomass N. Our observations suggest that grazing-exclusion as a restoration tool for upland habitats results in a slowing down of rates of C and N cycling. However, as yet, this has had no detectable impact on total C and N stocks in surface soil. Whereas increases in soil C and N stocks might be expected in the longer term, our results suggest that a certain level of grazing is compatible with the provision of ecosystem services such as soil C storage under traditional upland farming practices.

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1. Introduction

Despite their limited agricultural value, British upland ecosystems show a long history of land use (Simmons, 2003), particularly grazing management (Anderson and Yalden, 1981; Done and Muir, 2001). Effects of grazing on upland ecosystems include the replacement of vegetation dominated by *Calluna vulgaris* and other dwarf-shrubs (moorlands) by acidic, low productivity grasslands of a lower conservation value (Bardgett et al., 1995; Thompson et al., 1995). Herbivore exclusion is a common practice to restore moorlands, but its effects on ecosystem services related to soil carbon (C) and nutrient cycling has been poorly studied in upland habitats. Such lack of knowledge occurs despite the recognition that upland habitats represent one of the UK's largest stores of soil C (Milne and Brown, 1997), and their status as sensitive habitats (Thompson et al., 1995).

Here, we assessed 7 years of grazing exclusion on vegetation biomass (aboveground and belowground), plant functional

group composition, and soil properties, including biomass and activity of soil microbes, rates of soil N mineralisation, concentration of soluble C and N forms, and total contents of C and N. Our sampling included seasonal variability in order to determine grazing-exclusion effects against seasonal variation. We tested the hypothesis that grazing-exclusion is associated with a slowing-down in soil N and C cycling, characterised by reductions in rates of N mineralisation and microbial activity, and an increase in soil C and N content.

2. Materials and methods

2.1. Site description and experimental design

In 2007, we selected 2 adjacent areas with similar topography and altitude, but with contrasting recent grazing management in the Ingleborough National Nature Reserve in the Yorkshire Dales northern England (54.18°N, 2.36°E, Fig. 1, climate is temperate maritime, mean annual precipitation is 1840 mm). One area (hereafter grazed area) is a 58 ha acidic grassland dominated by *Nardus stricta* L., *Festuca ovina* L., *Agrostis capillaris* L., with a minor presence of *Eriophorum vaginatum* L. The grazed area was grazed at

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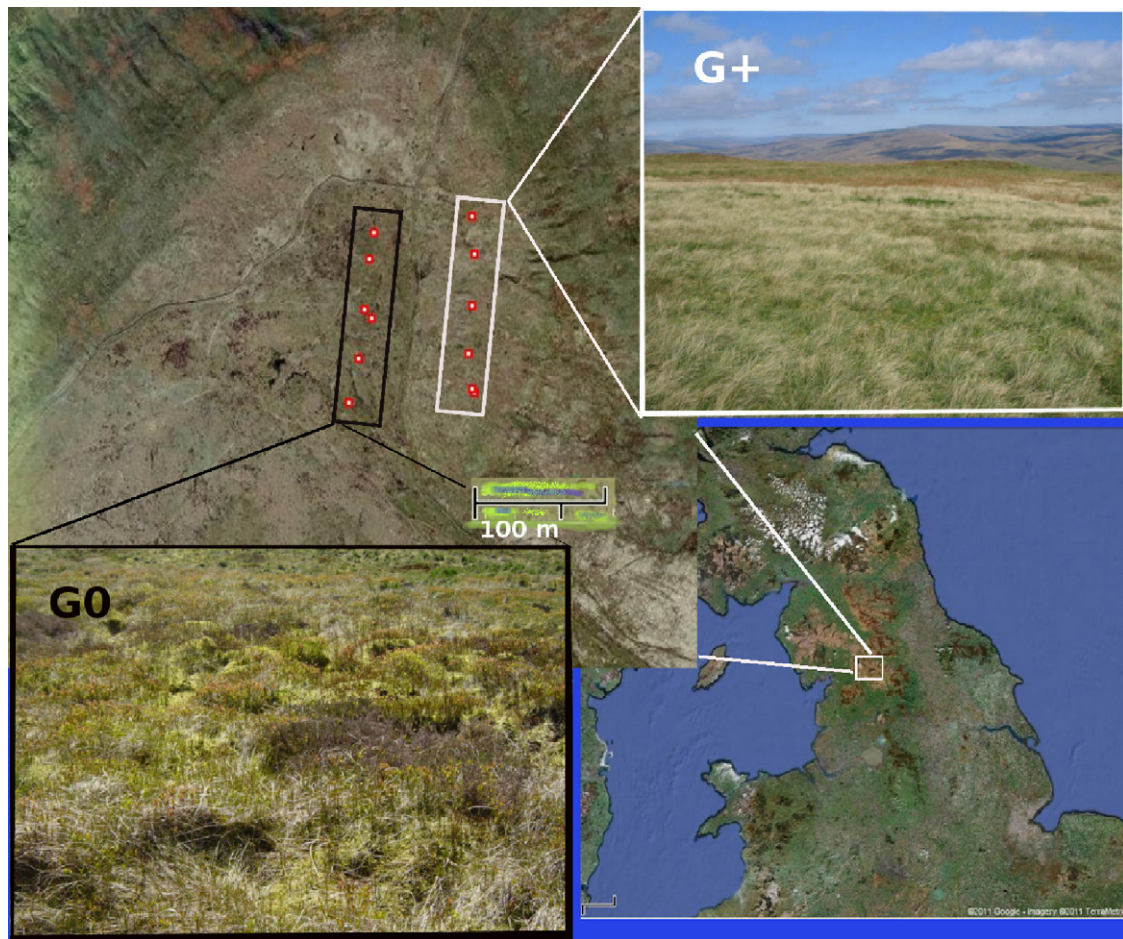


Fig. 1. The field site at Ingleborough National Nature Reserve, northern England, showing the approximate location of the experimental plots as determined by geographical positioning (red squares), and the main vegetation characteristics of an area where domestic herbivores have been excluded over 7 years (G0) and an adjacent grazed acidic grassland (G+). Images obtained from <http://maps.google.com/> (© Google 2011) and authors' own collection.

1–3 ewes ha⁻¹ before 1996, but at 4 ewes ha⁻¹ since then, with some cattle grazing for 10 weeks since mid-July onwards (Colin Newlands, Natural England, *pers. comm.*). Winter grazing stocking rate is 1.5 ewes ha⁻¹ from early November until mid-March. The other area (hereafter grazing exclusion) is an adjacent 170 ha field fenced off in 2000 to exclude livestock to re-establish moorland vegetation. Soils in both areas are derived from carboniferous sandstones in the Yoredale group (Waltham, 2008), pH of 4.5 (triplicate measure in 1:2.5 soil to water suspensions), and have an organic horizon of 20–30 cm. Neither area has received artificial fertilisers or manures.

Since the restoration project was done at the landscape level, pseudo-replication was unavoidable. We randomly set up, 6 plots (4 m × 4 m) along a NW–SE transect within each area. Plots selected had representative vegetation, similar soil depth (40 cm on average), and were located at similar topographical positions in both areas (Fig. 1). Each replicated plot was split into 16 1 m × 1 m sub-plots wherein all soil and vegetation samples were randomly taken.

2.2. Soil and vegetation sampling

On each of 5 dates from mid spring 2007 until early summer 2008, we excavated a turf including vegetation and the organic peat horizon (20 cm × 20 cm area, 20 cm depth) from each sub-plot with

a shovel. Samples were kept within coolers, transported, and stored at 4 °C, then analysed within 5 days after collection. Root biomass was measured from two soil cores (10 cm depth within the organic horizon since high moisture made recovery of intact soil cores from higher depths infeasible) adjacent to sub-plots. We inserted perforated PVC pipes (21 mm diameter × 1000 mm length) on each plot to record two-weekly water table depth (Oechel et al., 1998) from mid July until mid November 2007.

2.3. Aboveground and belowground biomass, and size of litter horizons

After being oven-dried at 70 °C for 48 h and weighed, dry mass pools of sampled turves were divided into 4 components following Ward et al. (2007): (1) aboveground plant biomass; (2) litter (L) horizon underlying vegetation; (3) a horizon (F+H not recorded on the second date) with unidentifiable plant residues and humified material; and (4) the organic horizon (O). Aboveground plant biomass was sorted into plant functional groups (Ward et al., 2009), namely graminoids, dwarf-shrubs, forbs and non-flowering plants including mosses and clubmosses (*Lycopodium* sp.). Roots were recovered by washing and sieving (minimum mesh size 0.5 mm) soil cores, then oven-dried and weighed to determine root biomass.

2.4. Soil analysis

Before O horizon perturbation, we took a core (3.5 cm diameter, 10 cm depth) from sampled turves to measure bulk density and soil moisture content by standard procedures. Remaining soil was sieved (2 mm mesh) prior to analysis. Total C and N content for two sampling dates were done in a Carlo Erba EA-1108 elemental analyser. Since no differences in total C between the two dates were observed (Wilcoxon test $W=71$, $n=12$, $P=0.97$), we calculated total C and N assuming no change over the sampling period.

Soil ammonium (NH_4^+) and nitrate (NO_3^-) contents, and rates of net NH_4^+ and NO_3^- mineralisation, were estimated by continuous-flow colorimetry in a Bran and Luebbe AutoAnalyzer 3 as described in Medina-Roldán and Bardgett (2011). Dissolved organic C (DOC) and N (DON) were extracted in water and measured in a Shimadzu 5000A TOC analyser (Shimadzu Inc., Japan) and by continuous-flow colorimetry respectively as described in Gordon et al. (2008). Fumigation-extraction was used to determine microbial biomass C and N as described in Medina-Roldán and Bardgett (2011). Soil basal respiration was determined in an ADC 225 MK3 IRGA (ADC Bioscientific Ltd., Hoddesdon, UK) following Bardgett et al. (1999).

2.5. Statistical analysis

We tested grazing exclusion effects on all dependent variables, except plant functional composition, by using repeated measures ANOVA in R (Pinheiro et al., 2008). Grazing management (grazing exclusion vs grazing) and date of sampling (with a linear correlation structure) were fixed factors. Replicated plots (subjects) within sampling dates were the random factor. Total soil C differences between land management regimes were tested with a t -test. Most variables were analysed on an area basis using bulk density and a soil depth of 10 cm. All variables were transformed to satisfy normality criteria, but results and figures are presented with untransformed values.

We tested grazing exclusion effects on plant functional groups as describe above, but with a non-parametric multivariate ANOVA (Anderson, 2008) implemented in R (Oksanen et al., 2010). All analyses were done with the R package for Linux (R Development Core Team, 2010).

3. Results and discussion

3.1. Vegetation composition

Grazing exclusion increased the relative abundance of dwarf-shrubs and reduced the proportion of graminoid species as expected (Table 1). Nevertheless, graminoids were not the same between areas, with *Nardus stricta*, *Festuca* spp. and *Agrostis* spp. dominating under grazing conditions, and *Deschampsia* spp., *Agrostis* spp. and *Eriophorum vaginatum* dominating the grazing exclusion. Mosses and other non-flowering plants (mainly *Lycopodium* sp.) were not affected by grazing exclusion (Table 1), although they were an important component of the vegetation on both areas. Seasonal differences in vegetation were marginally significant. These changes in vegetation are widely reported after grazing cessation in uplands of the UK (Marrs et al., 1989), and are attributed to an increase in competitive ability of grazing sensitive dwarf-shrubs when grazing pressure is released.

3.2. Aboveground and belowground biomass, and size of litter horizons

Grazing exclusion effects on aboveground biomass depended on season, being larger in the ungrazed area in May and July 2007

(Table 1). Although it is known that grazing exclusion can cause increases in aboveground biomass, high seasonal variability is generally considered to be more important driver of this measure (Marrs et al., 1989; Milne et al., 2002). Grazing exclusion increased the litter (L) layer by 70%, although this effect was marginally significant (Table 1). Welch and Rawes (1964) reported similar increases in the L horizon after 7-years of grazing exclusion at an upland site in northern England, and Marrs et al. (1989) observed that the size of the L horizon became even larger after 30 years of grazing cessation at the same site. Such increase in the L horizon is likely due to high biomass allocation to litter by the dwarf-shrubs that became dominant after grazing exclusion, and to the lower decomposability of this litter due to its high concentration of condensed tannins and phenolics (Vuuren et al., 1992). Furthermore, grazing increases plant biomass turnover through consumption (Cebrian, 1999), so in its absence dead vegetation will accumulate. Although we did not measure C content of the L horizon, Ward et al. (2007) reported 35% accumulation of C in L layer after 60 years of grazing exclusion in British heather moorland. This, together with our results, highlights the potential to increase the C pool in the L layer in upland ecosystems after long term grazing exclusion. Root biomass was not affected by grazing exclusion, but it varied significantly with season across both areas, being largest in early summer (May 2007) and lowest in winter (January 2008) (Table 1). The F and H horizon was not affected by grazing exclusion, and it did not vary seasonally.

3.3. Soil properties

Consistent with our hypothesis, we observed that grazing exclusion was associated with a slowing-down of soil nutrient cycling. For example, we found that grazing exclusion caused a 20% reduction in microbial activity, measured as soil basal respiration (a measure of soil heterotrophic activity, Table 1). Also, grazing exclusion consistently reduced net NH_4^+ mineralisation, although this reduction was dependent on season being lower in ungrazed than grazed areas for all 2007 sampling dates and in June 2008, and the opposite in January 2008 (Table 1). Finally, grazing exclusion increased the ratio of DON to dissolved inorganic N (DIN) (Table 1), which has been shown to increase as ecosystem productivity decreases (Farrell et al., 2011). Such differences in soil nutrient cycling between ungrazed and grazed ecosystems have been observed across gradients of grazing-management intensification in temperate upland habitats (Bardgett et al., 2001), and other ecosystems (Augustine et al., 2003; McNaughton et al., 1988). Such slowing-down of nutrient cycling is most likely due to the lack of input of animal excreta, which is known to stimulate microbial activity and nutrient cycling in grassland soils (Augustine et al., 2003; Bardgett and Wardle, 2003; McNaughton et al., 1988), and to changes in plant functional composition that lead to increased litter amount and likely reduced litter quality. Grazing exclusion also decreased microbial biomass N by 30%, which adds additional support to our conclusion that the cessation of grazing slowed down nutrient cycling (Table 1). Although microbial biomass N is not a direct measure of ecosystem N cycling, we did detect a positive correlation between this measure and the rate of NH_4^+ mineralisation (Kendall's rank correlation coefficient (τ)=0.17, $z=1.8$, $P=0.06$) and DON ($\tau=0.56$, $z=6.4$, $P<0.001$), which indicates that the availability of inorganic and organic N pools might be regulated by the microbial biomass. The slowing down of N cycling could also be, in part, due to higher moisture content of ungrazed soils. However, soil moisture content was extremely high in both grazed and ungrazed soils, and grazing exclusion did not affect the depth of the soil water table (data not shown). These findings suggest that moisture was not a significant driver of differences

Table 1

Effects of grazing (G+) vs grazing exclusion (G–) on plant functional groups abundance (percentage of aboveground biomass), dry mass pools (g DM m⁻²), soil biogeochemical and soil physical properties. Mineralisation rates in mg m⁻² day⁻¹, ratios are unit-less, and the rest of biogeochemical variables in g m⁻². Soil moisture (%), and soil bulk density (kg m⁻³). Seasonal variability (dd/mm/yr) has been included. All data are means (se). ns: no significant. Main results highlighted in bold.

	G+	G–	14/05/07	19/07/07	19/10/07	24/01/08	17/06/08	G	S	G × S
Plant functional groups										
Full model								$F_{1,48} = 8.0^{**}$	$F_{4,48} = 1.9^{\dagger}$	$F_{4,48} = 1.5^{ns}$
Graminoids	43.2 (4.7)	23.4 (4.7)	47.9 (7.1)	20.1 (5.5)	33.9 (9.1)	21.0 (6.4)	41.4 (9.1)	–	–	–
Forbs	1.9 (0.5)	1.9 (0.9)	3.0 (2.0)	1.9 (1.0)	0.9 (0.4)	3.2 (1.3)	0.6 (0.5)	–	–	–
Shrubs	3.0 (1.1)	20.4 (3.8)	9.8 (4.0)	17.8 (5.7)	10.1 (3.1)	17.1 (8.6)	4.7 (1.5)	–	–	–
Non spermatophytes	52.0 (4.2)	54.3 (5.2)	39.3 (5.6)	60.2 (6.8)	55.1 (7.3)	58.8 (8.7)	53.4 (8.1)	–	–	–
Dry mass pools										
Aboveground biomass	847 (65)	1134 (133)	1271 (218)	1224 (238)	885 (110)	718 (70)	851 (114)	$F_{1,10} = 6.0^{*}$	$F_{4,38} = 3.9^{**}$	$F_{4,38} = 14.6^{**}$
Belowground biomass	2814 (253)	2805 (279)	4283 (570)	3148 (245)	1996 (232)	1715 (329)	3088 (242)	$F_{1,10} = 1.1^{ns}$	$F_{4,36} = 10.6^{**}$	$F_{4,36} = 0.3^{ns}$
Litter horizon mass	592 (67)	981 (122)	1170 (131)	293 (51)	811 (241)	832 (141)	790 (101)	$F_{1,10} = 4.3^{\dagger}$	$F_{4,38} = 15.5^{**}$	$F_{4,38} = 0.1^{ns}$
F+H horizon mass	2118 (202)	2541 (202)	2941 (284)	–	1964 (337)	2294 (310)	2118 (319)	$F_{1,10} = 0.5^{ns}$	$F_{4,38} = 15.5^{**}$	$F_{4,38} = 0.2^{ns}$
Soil biogeochemical										
NH ₄ ⁺ mineralisation	62.9 (11.3)	43.1 (12.3)	119.2 (19.4)	68.2 (18.4)	26.6 (13.9)	48.0 (16.1)	25.4 (17.1)	$F_{1,10} = 2.5^{ns}$	$F_{4,35} = 6.0^{**}$	$F_{1,35} = 4.1^{**}$
NO ₃ ⁻ mineralisation	34.1 (10.6)	47.0 (15.6)	144.2 (38.5)	26.8 (9.5)	27.8 (9.3)	42.9 (14.5)	–4.3 (1.0)	$F_{1,10} = 0.2^{ns}$	$F_{4,39} = 6.2^{**}$	$F_{4,39} = 0.4^{ns}$
DOC	33.1 (4.5)	45.6 (7.6)	89.9 (14.9)	33.9 (4.4)	22.9 (1.7)	29.2 (5.2)	24.3 (2.5)	$F_{1,10} = 0.4^{ns}$	$F_{4,37} = 10.0^{**}$	$F_{1,37} = 1.2^{ns}$
DON	3.9 (0.6)	3.9 (0.5)	6.9 (0.8)	1.9 (0.2)	2.1 (0.2)	1.6 (0.1)	5.9 (0.8)	$F_{1,10} = 0.7^{ns}$	$F_{4,34} = 34.9^{**}$	$F_{1,34} = 0.6^{ns}$
DOC: DON ratio	11.4 (1.2)	13.3 (1.2)	12.7 (1.1)	17.5 (1.8)	11.0 (0.7)	18.0 (2.0)	4.9 (0.8)	$F_{1,10} = 1.8^{ns}$	$F_{4,34} = 19.2^{**}$	$F_{4,34} = 0.4^{ns}$
DON: DIN ratio	3.21 (0.58)	3.33 (0.25)	2.17 (0.23)	1.25 (0.20)	3.50 (0.32)	3.74 (0.78)	5.45 (0.89)	$F_{1,10} = 7.7^{*}$	$F_{4,36} = 54.0^{**}$	$F_{4,36} = 1.7^{ns}$
Microbial C	50.3 (7.5)	58.2 (12.6)	88.7 (14.0)	83.3 (26.8)	38.1 (6.6)	32.2 (7.8)	21.9 (5.3)	$F_{1,10} = 0.02^{ns}$	$F_{4,37} = 18.3^{**}$	$F_{4,37} = 1.2^{ns}$
Microbial N	19.7 (2.1)	14.1 (1.8)	31.8 (2.9)	10.8 (1.6)	9.7 (1.7)	9.4 (1.2)	21.9 (1.6)	$F_{1,10} = 21.1^{**}$	$F_{4,39} = 26.6^{**}$	$F_{4,39} = 1.5^{ns}$
C:N microbial ratio	3.4 (0.5)	3.9 (0.9)	2.7 (0.3)	5.3 (1.4)	5.9 (1.6)	3.0 (0.6)	0.9 (0.2)	$F_{1,10} = 0.001^{ns}$	$F_{4,37} = 6.7^{**}$	$F_{4,37} = 1.7^{ns}$
Soil basal respiration	8.1 (0.8)	6.4 (0.8)	6.9 (0.7)	12.9 (1.3)	5.1 (1.1)	5.1 (0.7)	6.0 (1.1)	$F_{1,10} = 10.9^{**}$	$F_{4,38} = 14.3^{**}$	$F_{4,37} = 1.2^{ns}$
Total soil C	5889 (367)	6238 (444)	–	–	–	–	–	$t_{20} = 0.7^{ns}$	–	–
Total soil N	249 (22)	269 (36)	–	–	–	–	–	$t_{7,4} = 0.2^{ns}$	–	–
Physical										
Bulk density	0.18 (0.09)	0.14 (0.04)	0.13 (0.03)	0.19 (0.10)	0.18 (0.07)	0.15 (0.06)	0.16 (0.04)	$F_{1,49} = 5.4^{*}$	$F_{4,49} = 1.4^{ns}$	$F_{4,49} = 0.6^{ns}$
Soil moisture	469 (29)	563 (26)	557 (43)	474 (42)	516 (52)	561 (53)	472 (33)	$F_{1,10} = 10.9^{**}$	$F_{4,39} = 5.1^{**}$	$F_{4,39} = 1.0^{ns}$

* $P < 0.05$.** $P < 0.01$.† $0.05 < P < 0.10$.

in microbial activity and N cycling between grazed and ungrazed areas.

Despite the slowing down of N cycling and build up of litter on the soil surface, grazing exclusion did not increase the amount of C or N in soil (Table 1). Soil C sequestration is a long term processes, and the lack of response of soil C here might reflect the relatively short time since grazers were excluded, which might have been insufficient for C, and also N, to accumulate. This is consistent with other studies of upland ecosystems that likewise show a lack of response of soil C to grazing removal, even after more than 30 years (Garnett et al., 2000; Marrs et al., 1989; Ward et al., 2007). Also, the lack of response of soil C and N to the cessation of grazing might be due to the lack of impact on mosses, which act as a significant C and N store in upland grasslands, and comprised just over half of the aboveground plant biomass on both areas. This might indicate that other restoration measures that promote dwarf shrubs in expense of graminoids and non-Sphagnum mosses might increase the potential for C sequestration of upland habitats. Our findings, combined with those of others, suggest that removal of domestic herbivores from upland grassland, at least in the short term (i.e. decades), is unlikely to reap benefits for soil C sequestration. Moreover, our findings suggest that current grazing management is unlikely to cause losses in soil C, aside potential losses associated with reductions in the size of the litter layer, which can act as a significant, albeit short term and unstable, C sink (Ward et al., 2007). More research is needed to corroborate these conclusions especially given that responses of soil C to grazing are highly variable across ecosystems (Conant et al., 2001; Bardgett and Wardle, 2010).

Grazing exclusion did not modify other soil properties, including DOC, DON and microbial C, which are key pools in the C and N cycle. However, most soil properties showed strong seasonal variation. Some of them, especially DOC, DON, microbial N and C, had higher values in warm periods in late spring and summer (Table 1), than in the colder periods in winter and autumn. High seasonal variation in these properties is consistent with Ward et al. (2007) who observed that seasonality was more important than grazing in explaining DOC variations in moorland, which they attributed to a combination of environmental factors which alter microbial activity.

4. Conclusion

Overall, our findings suggest that a decade of grazing exclusion has caused a slowing-down of C and N cycling and build up of surface litter in a restored upland ecosystem. However, this has not yet translated into any change in stocks of total C and N in soil. Soil C storage is a key goal of restoration programmes in degraded over-grazed uplands, and our results might suggest that certain levels of grazing intensity do not necessarily compromise the provision of this ecosystem service in uplands, although grazing reduced other C pools such as that in surface litter. Our data suggest that stability of the mosses functional group to grazing management might be responsible for no response in soil C and N. Therefore, moorland restoration schemes which can increase dwarf-shrubs cover in expense of graminoid and non-Sphagnum mosses might increase the potential of upland ecosystem for C sequestration, although further research is needed to support this conclusion.

Acknowledgements

We are grateful to the Mexican Council for Science and Technology (CONACYT) for the PhD. scholarship granted to EMR. We

are also grateful to Colin Newlands and Natural England for kindly allowing us access to the reserve and providing information on land use history of the field sites. We thank the UK Meteorological Office for access to climate datasets, and Simon Oakley, Gerlinde De Deyn and Helen Quirk for help in the field and laboratory.

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Further reading

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