Dear Associate Editor,

here we submit the revised version of our MS.

5 We adapted the text according to our responses to the earlier reviewers comments, dealing with every single issue raised by the reviewers.

Special attention was given to your latest requests:

10 - We included a statement in Material and Methods (Statistics) on the threshold value α and the P-values considered significant and marginally significant, respectively (p. 5, l. 30-31). Now the controversial expression 'marginally significant' is either accompanied by the respective P-value (p. 1, l. 17), or directly referenced to the statistical table (Tab. 3) (p. 6, l. 27) to avoid misunderstandings.

15 - The complex issue of bulk density changes associated with SOC concentration changes is now part of the discussion (p. 8, l. 19 – p. 9, l. 3).

We hope the mark-ups in the submitted manuscript make it easy to follow our adjustments

On behalf of the authors, kind regards, Matthias Volk

Subalpine grassland carbon balance during seven years of increased atmospheric N deposition

Matthias Volk, Jan Enderle & Seraina Bassin

Agroscope Zürich, Air Pollution/Climate Group, Reckenholzstrasse 191, 8046 Zurich, Switzerland

5 *Correspondence to*: Matthias Volk (matthias.volk@agroscope.admin.ch)

Abstract. Air pollution agents interact when affecting biological sinks for atmospheric CO_2, e.g. the soil organic carbon (SOC) content of grassland ecosystems. Factors favoring plant productivity, like atmospheric N deposition, are usually considered to favor SOC storage. In a seven year experiment in subalpine grassland under N- and O_3-deposition treatment, we examined C-fluxes and -pools. Total N deposition was 4, 9, 14, 29 and 54 kg N ha^{-1} yr^{-1} (N4, N9, etc.), annual mean phytotoxic O_3 dose

10 was 49, 65 and 89 mmol m^{-2} projected leaf area. We hypothesized that between years SOC of this mature ecosystem would not change in control treatments and that effects of air pollutants are similar for plant yield, net ecosystem productivity (NEP) and SOC content, leading to SOC content to increase with N deposition. Cumulative plant yield showed a significant N and N × N effect (+38% in N54), but no O_3-effect. In the control treatment SOC increased significantly by 9% in seven years. Cumulative NEP did show a strong, hump shaped response pattern to N deposition with a +62% increase in N14, and only

15 +39% increase in N54 (N effect statistically not significant, N × N interaction not testable). SOC had a similar, but not significant response to N, with highest C gains at intermediate N deposition rates, suggesting an unimodal response with a marginal ($P= 0.09$) N × N interaction. We assume the strong, pollutant-independent soil C sink developed as a consequence of the management change from grazing to cutting. The non-parallel response of SOC and NEP compared to plant yield under N deposition is likely the result of increased respiratory SOC losses, following mitigated microbial N-limitation or priming

20 effects, and a shift in plant C allocation leading to smaller C input from roots.

Keywords: Carbon sequestration, nitrogen deposition, tropospheric ozone, grassland, mountain, ecosystem CO_2 fluxes, SOC, R_{eco}, GPP, NEP

1 Introduction

25 Biological sinks for atmospheric CO_2, including grassland soils, substantially modulate the increase rate of this greenhouse gas (Sitch et al., 2008) and are also viewed as tools for greenhouse gas increase mitigation (Conant, 2010). Grassland soils contain 661 Pg C, equivalent c. 28% of total global soil C (excluding wetlands and frozen soils (Jobbágy and Jackson, 2000)) or >80% of C contained in the atmosphere. Climate change effects were predicted to decrease European grassland soil organic carbon (SOC) stocks by 6-10% during the 21st century (Smith et al., 2005). It is thus necessary to address the factors that

---

Margin annotations (tracked changes):

[1] verschoben

[1] nach oben: Cumulative plant yield showed a highly significant N effect (+38% in N54), but no O_3-effect.

determine the ecosystem carbon (C) sink properties, including air pollution agents, which interact to affect C sinks in grasslands.

Atmospheric N deposition exceeds critical loads in European lowlands, thus increasing ecosystem productivity and affecting biodiversity (Bobbink et al., 2010; Phoenix et al., 2012). Commonly, N deposition is much lower (c. 5 kg ha^{-1} yr^{-1}) at high altitude and in inner-alpine valleys (Rihm and Kurz, 2001) like our study site. Tropospheric O_3 concentrations have reached levels that affect plant growth (Fuhrer et al., 1997; Ashmore, 2005; Volk et al., 2006) and least conservative estimates predict mean annual O_3 background concentrations to rise to 84 ppb by 2100 (Vingarzan, 2004; Derwent et al., 2007). Ozone concentrations are particularly high at high altitude sites, due to reduced nocturnal degradation and deposition, and the natural gradient under free-tropospheric conditions (Chevalier et al., 2007).

C storage in ecosystems begins with plant growth. It is thus reasonable to expect that increased plant growth under increased N deposition has a similarly positive effect on ecosystem C storage (Vitousek et al., 1997). For example Ammann et al. (2009) found higher C sequestration in an intensively managed, high N input (c. 240 kg N ha^{-1} yr^{-1}) grassland, compared to an extensively managed field (no N fertilizer input) and in forests atmospheric N deposition showed a strong positive correlation with C sequestration (Magnani et al., 2007). But additional N input also has the potential to hinder C sequestration by favoring the decomposition rate of soil organic matter more compared to plant productivity (Neff et al., 2002). Meanwhile, a growing awareness of the highly dynamic and complex interactions of both climate parameters (e.g. water availability and temperature) and multiple pollution factors (e.g. N and O_3) has developed. Such scenarios lead to assume that the C sink of terrestrial ecosystems will show a much smaller growth, or even turn into a substantial source of atmospheric CO_2 (Lu et al., 2011; Heimann and Reichstein, 2008).

In the past, the scale for N input was often adopted from agronomic practices. Experimentalists used high, single dose N input treatments, and also extensive analyses did not differentiate effects of high and low N input doses (e.g. Liu and Greaver, 2010), summarizing N inputs from 10 to 650 kg N ha^{-1} yr^{-1}. But in their meta-analysis of fertilization experiments (24-600 kg N ha^{-1} yr^{-1} treatment range) Knorr et al. (2005) concluded that <20 × ambient N deposition or low-quality (high-lignin) litter inhibited decomposition. On the other hand, >20 × ambient N deposition or high-quality (low-lignin) litter stimulated decomposition, suggesting reduced ecosystem C storage in high N deposition environments. Usually, natural and semi-natural vegetation, including grasslands, does not receive management similar to such experimental conditions. Instead, atmospheric N input comes in a variety of doses that are very low by agricultural standards, combined with additional environmental agents like O_3 pollution. To make matters more complicated, different weather conditions in every growing season have the potential to alter quality and quantity of the individual pollutant effects.

Despite their applicability for the prediction of C sink properties under changing environmental conditions, low input dose, multilevel N deposition and multifactor experiments focusing on the carbon budget are rare in (semi-) natural ecosystems. Those available, usually studied one or more potentially beneficial atmospheric inputs, e.g. multi-level N deposition, multi-nutrient application (e.g. Fornara and Tilman, 2012; Fornara et al., 2013; Fang et al., 2014) or N × CO_2 interactions.

An earlier analysis in the same project, applying a fully factorial, multilevel $N \times O_3$ deposition treatment, had shown plant dry matter yield to increase consistently with rising N deposition rates (Volk et al., 2014), but there was no O_3 effect or $N \times O_3$ interaction. A net ecosystem CO_2 balance (net ecosystem productivity, NEP), that was parameterized for an exceptionally warm, single year, had shown C losses at all treatment levels and combinations, but the largest C loss coincided with the

5 highest N deposition treatment applied (Volk et al., 2011). Meanwhile, the full meteorological record for NEP parameterization, belowground yield data and SOC analysis from the destructive final harvest of the experimental plots is available. In the study presented here, we investigate the seven year impact of N and O_3 deposition on the grassland ecosystem C budget, from CO_2 exchange via plant productivity to SOC. We hypothesized that

I) 'Interannual effects', most importantly representing the weather variability, cause the annual NEP to be positively correlated

10 with temperature, reflecting the positive effect of warmer growing seasons on aboveground plant yield in this cold, high altitude environment. SOC content was expected to remain unchanged over seven years. The grassland developed under a low intensity management that was unaltered for decades, if not centuries. We therefore considered SOC to be in steady state in a mid- and long-term perspective.

II) 'Air pollutant effects' (O_3 and N deposition) cause both NEP and SOC to respond analogous to plant productivity. This

15 implies a consistently positive response to rising N input and a negative or zero response to increased O_3 deposition.

The study intends to improve the understanding of the C balance of extensively used, semi-natural grassland under realistic air pollution scenarios.

2 Material and methods

2.1 Experimental site and meteorological conditions

20 The experiment was located at 1990 m a. s. l., on a grassland plateau in the Central Alps (46° 32' N, 9° 39' E; Alp Flix, Sur, Canton Grisons, Switzerland). The natural tree line is at ~2200 m a. s. l.. Low-intensity management prior to the experiment (approximately 60 yrs.) was 3–4 weeks of cattle grazing at ~1.3 livestock units ha^{-1}. No manure or fertilizer was applied. Winters are cold with permanent snow cover from December until April. Annual mean temperature during the seven year experiment was 1.1 °C, with an April to October mean of 6.2 °C. Mean April to October precipitation was 853 mm. Background

25 O_3 hourly mean concentration for the growing period varied from 40 to 50 ppb. Very small diurnal and seasonal fluctuations are typical for this high-elevation site. Background N deposition - calculated from concentration measurements in air, rainwater and snow - amounts to <4 kg N ha^{-1} yr^{-1} (Bassin et al., 2007). A comprehensive description of the physical environment can be found in (Volk et al., 2014).

2.1 Experimental plots and treatment application

30 Plots consisted of 180 intact turf monoliths (L × W × H = 30 × 40 × 22 cm) that were excavated in the fall of 2003, placed in drained plastic boxes and randomly assigned to treatment combinations. Groups of 20 were placed in shallow pits, flush with

---

the surrounding surface in the center of nine O_3 fumigation rings. Comparative measurements showed that monolith soil temperature was 0.25°C cooler on average (not significant) than the soil at the nearby site of origin (Volk et al., 2011). To minimize confounding effects of microclimatic differences between rings, monoliths were rearranged annually between the fumigation rings, but kept within their respective O_3 treatment. Both the O_3 and N treatments were applied during the snow-

5   free period from approximately May to October.

Accounting for 4 kg N ha^{-1} yr^{-1} background deposition, total N loads in the five different treatments were 4 (control), 9, 14, 29, and 54 kg N ha^{-1} yr^{-1}. They are identified below as N4$_{control}$, N9, N14, N29, and N54, respectively. The extra N amounts were applied bi-weekly in twelve doses of ammonium nitrate (NH$_4$NO$_3$) solution in 200 ml of well water per monolith. N4$_{control}$ monoliths received only water. The amount of water added per year is equivalent to 2.8% of the mean precipitation during the

10   same period. Each N level was replicated four times in each fumigation ring.

The free-air ozone fumigation system (for details see Volk et al., 2003) had a control treatment (ambient [O_3], termed O_3 control) and two O_3 enrichment levels that achieved 1.4 × and 1.7 × ambient concentration (O_3+ and O_3++, respectively). Each treatment level was randomly assigned to three out of the total of nine fumigation rings. Ozone was continuously sampled at canopy height in the middle of each ring and measured with Dasibi 1008 series UV analyzers (Dasibi, Glendale CA, USA).

15   Annual mean accumulated exposure over a threshold of 40 ppb O_3 (AOT40) was 9.7, 21.0 and 38.9 ppm.h and phytotoxic ozone dose over the threshold of 0 mmol m^{-2} (POD0) was 48.7, 64.8 and 89.1 mmol O_3 m^{-2} projected leaf area in O_3 control, O_3+ and O_3++, respectively (Volk et al., 2014). For details on the O_3 treatment please refer to Volk et al. (2003).

2.2 Plant yield C content

Aboveground plant biomass was cut annually at peak canopy development (end of July), at 2 cm height. The harvested material

20   was oven dried and weighed to yield dry matter mass (DM). For details please refer to Bassin et al. (2007). Belowground root biomass was assessed from soil cores covering a subset of the monoliths (Volk et al., 2014). DM masses were expressed as g C m^{-2} based on plant biomass C concentration (C/N elemental analyzer measurements (Bassin et al., 2015)). Masses of four plots each were combined and averaged to match the lumping rules developed for soil sampling (compare below). Tests for effects of N- and O_3 deposition on mean plant C concentration yielded no results and a common value of 47% was assumed.

25   **2.3 Net ecosystem productivity**

Net ecosystem CO_2 exchange (NEE) was measured in 17 day + night campaigns, between end of May and end of October 2010 (the seventh treatment year), covering the complete growing season. The subset of 27 monoliths used covered all three O_3 treatments and three out of five N deposition treatments (N4$_{control}$, N14 and N54). The method utilizes a dynamic CO_2 concentration, non-flow-through, transparent acrylic chamber, fit to cover the whole grassland monolith including a fully

30   developed canopy. Reduced assimilation or respiration due to chamber effects (CO_2 substrate depletion, overheating, reduced diffusion gradient) could be safely excluded during the short measurement (2 min. per monolith).

Formatiert: Englisch (USA)

We parameterized seven year cumulative NEP CO_2 balance (NEP_{cum}) for the 2004-2010 period, based on NEE CO_2 flux data from 2010. We used global radiation (GR) and soil temperature at 5 cm depth (T_{soil}) to model gross primary productivity (GPP) and ecosystem respiration (R_{eco}), respectively. All methods are equivalent to Volk et al. (2011), except for an optimized exponential function for R_{eco} after Lloyd and Taylor (1994) and Ammann et al. (2007). System respiration substrate limitation
5   was accounted for based on reduced respiration observed in winter in mountain grassland (Galvagno et al., 2013). Accordingly, temperature normalized R_{eco} during the snow covered period was modelled to drop linearly to 50% of the last fall measurement until Dec. 31st. From Jan. 1st a linear rise from 50% of the first spring measurement to 100% at the date of this first measurement was assumed.

2.4 Soil organic carbon

10   SOC data presented here derive from all 180 monoliths, covering all five N- and three O_3-treatment levels. At the setup of the experiment in October 2003 soil from 0-10 cm and 10-20 cm depth was cut from the walls of the pits that resulted from the excavation of the monoliths. The bulk soil density of these samples is not known. Seven years later, in October 2010, two cylindrical soil cores per monolith were sampled from the surface to 10 cm and from 10-20 cm depth. All samples were dried and sieved (2 mm). For analysis sieved samples from four monoliths each, receiving the same treatment combination, were
15   lumped to get a total of 45 samples per sampling depth.
We measured soil organic C and N contents by elemental analysis (oxidation of C-CO_2 and N-NO_2 in an O_2 stream and subsequent reduction of NO_2-N_2 by a copper-tungsten granule). Separation of CO_2 and N_2 was accomplished by GC-TCD and quantification using acetanilid as an external standard (Hekatech Euro EA 3000, Wegberg, Germany). Samples were free of carbonate, so total C equals organic C. All soil C stock values (kg C m^{-2}) are based on density values established using the
20   2010 samples.

2.5 Statistics

Effects of treatments were tested in a split plot analysis using a linear mixed effect model (Pinheiro and Bates, 1996) with O_3 at the main plot level and N at the sub-plot level. For SOC concentration (45 samples), plant C (45 samples) and NEP_{cum} (27 samples) replication was n = 3 for O_3 on three treatment levels and n = 9 for N on five (SOC, plant C) or three (NEP) treatment
25   levels. Block entered the model as class variable, whereas O_3 and N were continuous variables. The O_3 treatment was quantified as the seven years cumulative POD0. Inference on the significance of variables and selected interactions was made using likelihood ratio tests (Burnham and Anderson, 2002). Data were Box-Cox-transformed when necessary. All analyses were carried out by means of the statistics software R (R Development Core Team, 2011) using the packages nlme, nortest, lattice and car. Probability of equal means for SOC concentrations in 2003 vs. 2010 was tested using a two-sided, paired Student's t-
30   test on samples from the control treatments. We chose a threshold value of α = 5% and probability values (P) ≤ 0.05 were considered significant and P ≤ 0.1 marginally significant. Values in the text and the figures are means ±1 standard error.

[2] nach unten: Values in the text and the figures are means ±1 standard error.

Formatiert: Schriftart: Kursiv

[2] verschoben

Formatiert: Schriftart: Kursiv

3 Results

3.1 Cumulative plant C yield

In the control treatment, the mean plant C contained in total above- and belowground biomass was 591 $\pm$60.3 g C m^{-2}. Of this plant material an annual mean of 74.9 $\pm$7.64 g C m^{-2} yr^{-1} was exported with the grass harvest (Fig. 1). Thus, seven years cumulative aboveground plant C yield ($Yield_{cum}$) in control plots is equivalent to 524 g C m^{-2}.

$Yield_{cum}$ in seven years showed a significant N-effect, but neither an O_3-effect nor an N $\times$ O_3 interaction. The increase of yield with increasing N deposition was continuous, but not linear, as indicated by the significant N $\times$ N interaction (Fig. 2; Tab. 1). Compared to $N4_{control}$ (509 $\pm$15.4 g C m^{-2}) the maximum $Yield_{cum}$ increase was +38% in the N54 treatment (702 $\pm$15.6 g C m^{-2}).

3.2 Net ecosystem productivity

In the control treatment the ecosystem CO_2 exchange balance for single years ranged from a loss of -14.2 $\pm$30.44 g C m^{-2} in warm and dry 2006 to a maximum gain of 68.3 $\pm$28.56 g C m^{-2} in cool and moist 2010 (Fig. 1). Linear regression of NEP over soil temperature yielded a negative correlation with R^2 = 0.82 (P = 0.005). NEP_{cum} for the seven year period was 245 $\pm$210.0 g C m^{-2}.

In NEP neither O_3 nor N caused statistically significant treatment effects or interactions (Tab. 2). Surprisingly, NEP_{cum} is smaller under N54-deposition (Fig. 3) compared to the N14 treatment (lumped over O_3 treatments: $N4_{control}$ 279 $\pm$89.3 g C m^{-2}, N14 452 $\pm$81.4 g C m^{-2}, N54 387 $\pm$96.0 g C m^{-2}). Owing to the low number of treatment levels (three), statistical testing for N $\times$ N interaction is not meaningful, yet the response of NEP_{cum} to rising N deposition suggests a hump shaped pattern.

3.3 Soil organic carbon

From 2003 to 2010 bulk soil C concentration in the 0-20 cm soil layer (control treatments) increased from 61 $\pm$1.9 g kg^{-1} to 67 $\pm$0.6 g kg^{-1} (P = 0.052). The concentration increase was unevenly distributed with +12% in the 0-10 cm layer (2003: 87 $\pm$2.5 g kg^{-1}, 2010: 97 $\pm$1.1 g kg^{-1}; P = 0.019) and +3% in the 10-20 cm layer (2003: 36 $\pm$3.4 g kg^{-1}, 2010: 37 $\pm$1.1 g kg^{-1}; $n.s.$; Fig. 4). The resulting C stock in the 0-20 cm layer grew from 6.9 $\pm$0.18 to 7.5 $\pm$0.02 kg C m^{-2}, equivalent +9% increase. A resampling campaign (0-10 cm depth) in 2012, at the site where the turf monoliths were excavated in 2003 and where the seasonal grazing had continued since, yielded a bulk soil C concentration of 83 $\pm$1.5 g kg^{-1}, slightly less than the first sampling nine years ago.

The N deposition related C gain occurred mostly in the top 10 cm of the soil column (Fig. 5). This soil layer shows a marginally significant N $\times$ N interaction (Tab. 3), indicating a hump shaped response function of SOC to N deposition. But no significant effects or interactions were found for N in the lower layer or for O_3 in either upper or lower soil layer. In the 0-10 cm soil layer all SOC stock increases were higher in the N deposition treatments compared to $N4_{control}$. But at low deposition rates (N9, N14)

SOC kg m^{-2} increased much further than at high deposition rates (N29, N54) (Fig. 5). Changes in the 10-20 cm layer mirror the upper layer, only there is no gain in N4$_{control}$.

4 Discussion

4.1 Cumulative plant C yield

5 Because O_3 did not show an even remotely significant effect on either plant C yield, NEP or SOC, here we do not discuss the highly variable responses, observed in the O_3 treated grassland plots.

Despite a decreasing N effect, Yield$_{cum}$ (Fig. 2) is monotonically increasing over the whole range of rising N deposition rates. This is a typical response across a range of ecosystems, with the response function depending mostly on the productivity of the system (e.g. Yuan et al., 2006). Surprisingly, the positive N effect on plant C yield translated unevenly into higher C values
10 for the parameters that are directly indicative of the ecosystem C budget (compare below).

4.2 Net ecosystem productivity

Despite a substantial gain over seven years, NEP$_{cum}$ suggested smaller C gains than those finally found in SOC. We suspect that a temperature based overestimation of C losses contributed to a small NEP$_{cum}$. In a sensitivity test of R$_{eco}$ a 1°C reduced soil temperature increased NEP$_{cum}$ to 787 ±194.9 g C m^{-2}. Thus, a small error in the temperature parameterization could be one
15 reason for the observed mismatch between NEP$_{cum}$ and SOC content.

GPP parameterization is based on the comparatively small 2010 canopy (cf. yield values Fig. 1) and NEP$_{cum}$ may therefore contain an underestimation of GPP. To explore the potential error, we alternatively scaled GPP linearly with aboveground yield of the respective years (2010 = 100%) and found a NEP$_{cum}$ of 1.61 ±0.23 kg C m^{-2}. This is c. 40% more than the sum of SOC gain (0.6 kg C m^{-2}) plus plant C exported with the harvest (0.5 kg C m^{-2}) and implies that GPP does not grow
20 proportionally with the size of the green canopy.

Although this scaling exercise substantially increased NEP, interannual comparison shows that canopy size and its influence on GPP play only a small role for NEP, compared to the temperature effect on R$_{eco}$. For example two years of very similar sized canopy showed the maximum NEP in 2010 (68 g C m^{-2}), but also the minimum NEP in 2006 (-14 g C m^{-2}). Also, both negative NEPs (2006 and 2009) coincided with the lowest and the highest annual aboveground yield during the experiment
25 (Fig. 1). Thus, contrary to expectations, no correlation could be found between annual NEP and yield. Instead, the lowest (i.e. most negative) NEPs were found in years with highest mean soil temperature, exactly like Marcolla et al. (2011) report (for air temperature) from a seven year mountain grassland eddy covariance study. In our experiment linear regression of NEP over mean T$_{soil}$ for the respective years results in $R^2 = 0.82$. This made T$_{soil}$ the largest single factor determining whether the system was a C sink or source in a given year.
30 Under air pollution treatment we found an unimodal/hump shaped response pattern of NEP$_{cum}$ (Fig. 3), but the significance of the N × N interaction is not testable with three treatment levels. We suggest a mechanism that, at low N deposition rates, favors

assimilatory plant C gains more and, at high N deposition, favors the antagonistic, respiratory soil C decomposition losses more.

The R_{eco} and GPP data, used for NEP_{cum} parameterization, allowed us to test the above proposition: For cumulative C loss from R_{eco} the N effect in the N14 treatment (N14/N4$_{control}$) is +3.3%, but in the N54 treatment (N54/N4$_{control}$) it rises to +5.8%, equivalent of a factor 1.8 increase. For cumulative C gain from GPP on the other hand, the positive effect in N14 is already +7.3%, but the rise in the N54 treatment is only to +8.1%. This is equivalent to an increase of only factor 1.1 between N14 and N54 treatment level. Thus, the response functions developed differently in assimilatory vs. respiratory processes and GPP is not growing substantially, once a comparatively low N deposition rate is exceeded.

Skinner (2013) reports the same pattern from eddy covariance measurements in a grassland fertilization experiment: In a first five year period NEE yielded C gains under low N (+26 kg N ha^{-1} yr^{-1}), compared to C losses under high N deposition (+83 kg N ha^{-1} yr^{-1}) treatment. With substantially increased fertilizer application (low +85, high +217 kg N ha^{-1} yr^{-1}) during the subsequent four year period all NEE balances were negative, indicative of the high N deposition treatment promoting respiration processes more than assimilation processes.

4.3 Soil organic carbon

SOC concentration values observed here are well within the established limits for this vegetation type. In the 0-20 cm layer of 15 grassland soils above 1000 m a.s.l., described by the Swiss National Soil Observatory (Desaules and Dahinden, 2000), SOC ranges from 3.5% to 8.7% (mean 5.4, SD 1.40). In a literature overview Leifeld et al. (2005) calculated a mean of 6.07% (SD 3.31) for >1000 m a.s.l. permanent grassland in Switzerland.

To convert the 2003 SOC concentrations into SOC stocks, we used the bulk soil density values established in 2010. Indeed soil bulk density is often decreasing in parallel with the input of fresh organic matter. This would imply a higher than assumed bulk density in 2003 and a smaller difference between SOC stocks in 2003 and 2010.

Suitable literature values to estimate the potential underestimation of bulk soil density at the start of our experiment were not found, as the bulk of the literature refers to restoration measures for overgrazed areas (e.g. Li et al., 2007), forest regrowth after abandonment (e.g. Guidi et al., 2014) or afforestation (e.g. Hiltbrunner et al., 2013). Therefore, in a thought experiment we follow the equal soil mass concept (Ellert and Bettany, 1995) and assumed a 10% density reduction to coincide with the C concentration increase. In this scenario the 2010 0-20 cm depth sampling campaign would cover only 90% of the soil mass present in the 2003 sample. The resulting error comes from the fact that the 0-20 cm sampling of the reduced density soil of 2010 only goes to a layer that was at ca. 18 cm in 2003. Consequently, to compare C content changes in the 2003 soil mass-equivalent after seven years, the 2010 sampling volume would have to include an extra 10%.or 2 cm depth. At our site the C concentration at 20 cm depth is ≤ 4% (Leifeld and Fuhrer, 2009). Including this extra volume of low C concentration soil into the sample would have reduced the mean concentration from 6.7% to 6.43%. In parallel with the lower C concentration, the C stock of 2010 would have to be reduced by factor 0.96, resulting in a 7.2 kg m^{-2} C stock. The remaining stock-increase from 2003 to 2010 is only ca. 0.30 kg C m^{-2}, compared to 0.6 kg when equal density is assumed. But at the same time the extra soil

volume of a density of ≥ 0.76 would have increased mean bulk density such that the resulting stock would be 7.4 kg C m^{-2}. Using this alternative calculation, introducing a density correction, the 0-20 cm carbon gain would be c. 20% less, only about 0.5 kg C m^{-2} instead of 0.6 kg C m^{-2}.

We assumed the management change did not result in decreased bulk soil density from reduced trampling, because the Leifeld and Fuhrer (2009) study in adjacent plots showed no differences, even 60 years after the meadow/pasture management change. Both grasslands had a $\rho = 0.64 \pm 0.04$ for 0-16cm depth, slightly higher but very similar to the $\rho = 0.60 \pm 0.005$ we found for 0-20cm soil depth in this study.

The small decrease in SOC concentration under continued seasonal grazing, found at the 2012 resampling of the site where the monoliths came from, showed that the effect seen in the trial must have been a treatment effect or a side effect of the experimental management. The monoliths were slightly cooler compared to the original site and our study found that soil temperature is by far the largest single factor determining the C balance of the mountain grassland. But judging from the sensitivity analysis mentioned above, we are confident that the insignificant temperature difference is responsible for a small effect on soil C stock only, if any.

We suggest that the newly introduced cutting at only 2.5 cm above the ground represents a more intensive use, compared to selective grazing by cattle, and that this has created a new SOC source. Because plants maintain a functional root/shoot equilibrium (Poorter et al., 2012), the aboveground harvest will cause a proportional root dieback. In the control treatment the mean annual above- plus belowground live plant C content was >500 g C m^{-2}. If all newly introduced soil C (0.6 kg m^{-2}) in seven years derived from solid plant material, an additional 86 g C m^{-2} yr^{-1} or c. 15% of the plant C stock must have been incorporated in the soil, compared to steady state SOC stock conditions. Indeed, a mean of 74.9 $\pm$7.64 g C m^{-2} yr^{-1} was cut and exported with the grass harvest, an amount quite similar to the required annual SOC input, making proportional root dieback a possible source for the belowground C gain. Similarly, from the eight eddy covariance mountain sites of the Carbomont network it was concluded that NEE for pastures was close to zero, while cut meadows accumulated carbon (Berninger et al., 2015). This 'grazed vs. cut' contrast could be equivalent to the long-term Reeder and Schuman (2002) grassland study, with increasing SOC stocks along a gradient of increasing grazing intensity. Further support comes from the Leifeld et al. (2015) study, showing that root median residence time was approximately halved (from 7.3 to 3.8 years), when grassland management intensity rose from 'low' to 'medium'. This intensity change is similar to what the grassland monoliths in our study experienced, when they were transferred from the occasionally grazed pasture into the experimental cutting regime.

Under N deposition air pollution treatment SOC mirrored the response pattern seen in the seven years NEP parameterization. The strong increase at low, but only small increase at high N deposition rates, demands an asymmetric promotion of C accumulation at different N deposition rates. Both soil layers (Fig. 5) show this characteristic pattern, which makes it likely that the same processes are active, only with a lag phase and at a slower rate in the lower layer. This is because the top soil layer has a higher proportion of recently assimilated, labile C, consisting of plant litter, roots, microbial and fungal biomass, as indicated by the >2 × higher soil C concentration (Fig. 4).

Some studies suggested soil C content to increase by reducing organic matter decomposition following high N deposition, either as a result of abiotic interactions with microbial products or from effects on the decomposer community (Hobbie, 2008). High lignin litter reduced decomposition in Dijkstra et al. (2004), while Waldrop et al. (2004) differentiates accelerated decomposition of easily decomposable litter and reduced decomposition of recalcitrant (high lignin) litter.

5 But the Park Grass Experiment at Rothamsted, UK (Fornara et al., 2011) shows no change of SOC content in N fertilized plots (96 kg N ha^{-1} yr^{-1}) between 1876 and 2005, indicative of the N-input rate to be beyond the positive C-balance point. In an alpine grassland at Niwot Ridge LTER Neff et al. (2002) found N deposition >100 kg ha^{-1} yr^{-1} to accelerate microbial decomposition of labile C pools, leading to a zero soil C balance despite substantially increased plant C input.

Increasing N doses were also found to decrease soil C content via higher microbial activity. Generally speaking, microbial
10 activity or population size may suffer similar nutrient constraints as plant productivity does (cf. 'Law of the minimum'; von Liebig, 1840). N limitation of soil microbial growth has been suggested based on theoretical considerations (Schimel and Weintraub, 2003) and Stone et al. (2012) found increased activity of extracellular hydrolytic enzymes from soil microbiota after long-term 25-35 kg ha^{-1} yr^{-1} N deposition in the field. A meta-analysis of 77 grassland studies on N deposition and CO_2 enrichment effects (Sillen and Dieleman, 2012) concluded that 'moderate' N deposition (equivalent to the maximum N
15 deposition in our experiment) was accelerating decomposition processes, thus reducing the soil C storage potential. Likewise Li et al. (2014) report maximum SOC concentrations at intermediate rather than maximum treatment levels after five years of high N input rates (50, 100 and 150 kg N m^{-2} yr^{-1}) in alpine meadows, finding the hump shaped response at generally high input levels.

Also the quality of available C as a source of energy plays an important role. Fontaine et al. (2003) describe a mechanism that
20 may often stand behind substantial soil C losses in the face of increased plant growth: Newly available carbohydrates in the soil create a competitive stimulation of microbial populations that results in strongly increased decomposition rates (priming effect).

But besides decomposition changes of uncertain direction, plant C allocation plays an important role. In an earlier analysis Volk et al. (2014) had found N deposition to change the Shoot/Root ratio (S/R) only marginally between N4$_{control}$ (0.22 ±0.013)
25 and N14 (0.23 ±0.013). But S/R increased quite dramatically in N54 (0.29 ±0.009), equivalent to a 24% smaller root proportion of total biomass. Thus, despite increasing aboveground plant C mass, a strong allocation shift, away from roots in favor of aboveground plant organs, may have led to smaller soil C input from root turnover.

Lacking low dose, multilevel N deposition treatments, cases of tipping response functions could usually not be found as a consequence of the experimental design. But in the 19 year, multi-nutrient experiment described by Fornara et al. (2013), the
30 effect of multiple nutrients may constitute an analogy to the effect of increasing N deposition levels. Smaller C sequestration was found in NP, PK and NPKMg treatments compared to the N only application, while aboveground plant mass grew consistently with nutrient addition (Control < N < NP < NPKMg). This is indirect evidence that a more effective plant fertilization does not necessarily result in higher SOC sequestration, but instead creates a situation where decomposition or

allocation changes are more favored than assimilation, at higher or combined nutrient addition treatments, with negative effects on soil C content.

5 Conclusion

Annual NEP was negatively correlated with soil temperature, not positively correlated with air temperature like plant yield. Contrary to expectations, soil C concentration in the subalpine grassland substantially increased over time. Thus, as an effect of management change from grazing to cutting, the soil was a strong C sink in the 2003-2010 period. Despite consistently positive responses of aboveground plant yield to increasing N deposition, SOC increased substantially only at low N deposition, but grew less at high deposition rates. NEE measurements suggest, that N deposition caused extra C-losses via higher microbial respiration after mitigation of N limitation or as a consequence of priming, resulting from the increased input of fresh organic plant material. Also plant C-allocation changes may have contributed to reduced soil C input from roots at high N deposition doses. These mechanisms are likely responsible for the hump shaped response of SOC to increasing N deposition. Accordingly, N deposition driven yield increases in low productivity grasslands may not be considered as a valid proxy for parallel ecosystem C-pool increases.

6 Acknowledgements

This study was supported by the Swiss Federal Office for the Environment in the framework of the International Cooperative Programme (ICP Vegetation) under the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP) and the EU project ECLAIRE. Local support by Victoria Spinas, Alfons Cotti and Gemeinde Sur is greatly acknowledged.

7 References

Ammann, C., Flechard, C. R., Leifeld, J., Neftel, A. and Fuhrer, J.: The carbon budget of newly established temperate grassland depends on management intensity, Agriculture, Ecosystems & Environment, 121(1-2), 5–20, doi:10.1016/j.agee.2006.12.002, 2007.

Ammann, C., Spirig, C., Leifeld, J. and Neftel, A.: Assessment of the nitrogen and carbon budget of two managed temperate grassland fields, Agriculture, Ecosystems & Environment, 133(3-4), 150–162, doi:10.1016/j.agee.2009.05.006, 2009.

Ashmore, M. R.: Assessing the future global impacts of ozone on vegetation, Plant, Cell & Environment, 28(8), 949–964, 2005.

Bassin, S., Volk, M., Suter, M., Buchmann, N. and Fuhrer, J.: Nitrogen deposition but not ozone affects productivity and community composition of subalpine grassland after 3 yr of treatment, New Phytologist, 175(3), 523–534, doi:10.1111/j.1469-8137.2007.02140.x, 2007.

Bassin, S., Käch, D., Valsangiacomo, A., Mayer, J., Oberholzer, H.-R., Volk, M. and Fuhrer, J.: Elevated ozone and nitrogen deposition affect nitrogen pools of subalpine grassland, Environmental Pollution, 201, 67–74, doi:10.1016/j.envpol.2015.02.038, 2015.

We conclude that a

Berninger, F., Susiluoto, S., Gianelle, D., Bahn, M., Wohlfahrt, G., Sutton, M., Garcia-Pausas, J., Gimeno, C., Sanz, M. J., Dore, S. and others: Management and site effects on carbon balances of European mountain meadows and rangelands, Boreal Environment Research, 20, 748–760, 2015.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.-W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L. and De Vries, W.: Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis, Ecol. Appl., 20(1), 30–59, doi:10.1890/08-1140.1, 2010.

Burnham, K. P. and Anderson, D. R.: Model selection and multimodel inference: a practical information-theoretic approach, Springer Science & Business Media., 2002.

Chevalier, A., Gheusi, F., Delmas, R., Ordonez, C., Sarrat, C., Zbinden, R., Thouret, V., Athier, G. and Cousin, J.-M.: Influence of altitude on ozone levels and variability in the lower troposphere: a ground-based study for western Europe over the period 2001–2004, Atmospheric Chemistry and Physics, 7(16), 4311–4326, 2007.

Conant, R. T.: Challenges and opportunities for carbon sequestration in grassland systems: A technical report on grassland management and climate mitigation, Food and Agriculture Organization of the United Nations, Rome, 2010.

Derwent, R. G., Simmonds, P. G., Manning, A. J. and Spain, T. G.: Trends over a 20-year period from 1987 to 2007 in surface ozone at the atmospheric research station, Mace Head, Ireland, Atmospheric Environment, 41(39), 9091–9098, doi:10.1016/j.atmosenv.2007.08.008, 2007.

Desaules, A. and Dahinden, R.: Nationales Boden-Beobachtungsnetz: Veränderungen von Schadstoffgehalten nach 5 und 10 Jahren: Messperioden 1985-1991 und 1992-1997, Bundesamt für Umwelt, Wald und Landschaft (BUWAL)., 2000.

Dijkstra, F. A., Hobbie, S. E., Knops, J. M. H. and Reich, P. B.: Nitrogen deposition and plant species interact to influence soil carbon stabilization, Ecology Letters, 7(12), 1192–1198, doi:10.1111/j.1461-0248.2004.00679.x, 2004.

Ellert, B. H. and Bettany, J. R.: Calculation of organic matter and nutrients stored in soils under contrasting management regimes, Canadian Journal of Soil Science, 75(4), 529–538, 1995.

Fang, H. J., Cheng, S. L., Yu, G. R., Yang, X. M., Xu, M. J., Wang, Y. S., Li, L. S., Dang, X. S., Wang, L. and Li, Y. N.: Nitrogen deposition impacts on the amount and stability of soil organic matter in an alpine meadow ecosystem depend on the form and rate of applied nitrogen: Dynamics of organic carbon in alpine meadow soils, European Journal of Soil Science, 65(4), 510–519, doi:10.1111/ejss.12154, 2014.

Fontaine, S., Mariotti, A. and Abbadie, L.: The priming effect of organic matter: a question of microbial competition?, Soil Biol. Biochem., 35(6), 837–843, doi:10.1016/S0038-0717(03)00123-8, 2003.

Fornara, D. A. and Tilman, D.: Soil carbon sequestration in prairie grasslands increased by chronic nitrogen addition, Ecology, 93(9), 2030–2036, 2012.

Fornara, D. A., Steinbeiss, S., McNamara, N. P., Gleixner, G., Oakley, S., Poulton, P. R., Macdonald, A. J. and Bardgett, R. D.: Increases in soil organic carbon sequestration can reduce the global warming potential of long-term liming to permanent

grassland: Liming effects on soil carbon sequestration, Global Change Biology, 17(5), 1925–1934, doi:10.1111/j.1365-2486.2010.02328.x, 2011.

Fornara, D. A., Banin, L. and Crawley, M. J.: Multi-nutrient vs. nitrogen-only effects on carbon sequestration in grassland soils, Global Change Biology, 19(12), 3848–3857, doi:10.1111/gcb.12323, 2013.

5 Fuhrer, J., Skärby, L. and Ashmore, M. R.: Critical levels for ozone effects on vegetation in Europe, Environmental pollution, 97(1), 91–106, 1997.

Galvagno, M., Wohlfahrt, G., Cremonese, E., Rossini, M., Colombo, R., Filippa, G., Julitta, T., Manca, G., Siniscalco, C., Morra di Cella, U. and Migliavacca, M.: Phenology and carbon dioxide source/sink strength of a subalpine grassland in response to an exceptionally short snow season, Environmental Research Letters, 8(2), 025008, doi:10.1088/1748-10 9326/8/2/025008, 2013.

Guidi, C., Vesterdal, L., Gianelle, D. and Rodeghiero, M.: Changes in soil organic carbon and nitrogen following forest expansion on grassland in the Southern Alps, For. Ecol. Manage., 328, 103–116, 2014.

Heimann, M. and Reichstein, M.: Terrestrial ecosystem carbon dynamics and climate feedbacks, Nature, 451(7176), 289–292, 2008.

15 Hiltbrunner, D., Zimmermann, S. and Hagedorn, F.: Afforestation with Norway spruce on a subalpine pasture alters carbon dynamics but only moderately affects soil carbon storage, Biogeochemistry, 115(1–3), 251–266, doi:10.1007/s10533-013-9832-6, 2013.

Hobbie, S. E.: Nitrogen effects on decomposition: A five-year experiment in eight temperate sites, Ecology, 89(9), 2633–2644, doi:10.1890/07-1119.1, 2008.

20 Jobbágy, E. G. and Jackson, R. B.: The vertical distribution of soil organic carbon and its relation to climate and vegetation, Ecological applications, 10(2), 423–436, 2000.

Knorr, M., Frey, S. D. and Curtis, P. S.: Nitrogen additions and litter decomposition: a meta-analysis, Ecology, 86(12), 3252–3257, 2005.

Leifeld, J. and Fuhrer, J.: Long-term management effects on soil organic matter in two cold, high-elevation grasslands: clues 25 from fractionation and radiocarbon dating, European Journal of Soil Science, 60(2), 230–239, doi:10.1111/j.1365-2389.2008.01111.x, 2009.

Leifeld, J., Bassin, S. and Fuhrer, J.: Carbon stocks in Swiss agricultural soils predicted by land-use, soil characteristics, and altitude, Agriculture, Ecosystems & Environment, 105(1-2), 255–266, doi:10.1016/j.agee.2004.03.006, 2005.

Leifeld, J., Meyer, S., Budge, K., Sebastia, M. T., Zimmermann, M. and Fuhrer, J.: Turnover of Grassland Roots in Mountain 30 Ecosystems Revealed by Their Radiocarbon Signature: Role of Temperature and Management, PLoS One, 10(3), e0119184–Article No.: e0119184, doi:10.1371/journal.pone.0119184, 2015.

Liebig, von J. F.: Die organische Chemie in ihrer Anwendung auf Agricultur und Physiologie, Vieweg., 1840.

Li, X.-G., Li, F.-M., Zed, R. and Zhan, Z.-Y.: Soil physical properties and their relations to organic carbon pools as affected by land use in an alpine pastureland, Geoderma, 139(1–2), 98–105, doi:10.1016/j.geoderma.2007.01.006, 2007.

Li, J. H., Yang, Y. J., Li, B. W., Li, W. J., Wang, G. and Knops, J. M. H.: Effects of Nitrogen and Phosphorus Fertilization on Soil Carbon Fractions in Alpine Meadows on the Qinghai-Tibetan Plateau, PLoS ONE, 9(7), e103266, doi:10.1371/journal.pone.0103266, 2014.

Liu, L. and Greaver, T. L.: A global perspective on belowground carbon dynamics under nitrogen enrichment: Belowground
5   C dynamics under N enrichment, Ecology Letters, 13(7), 819–828, doi:10.1111/j.1461-0248.2010.01482.x, 2010.

Lloyd, J. and Taylor, J. A.: On the temperature dependence of soil respiration, Functional Ecology, 315–323, 1994.

Lu, M., Zhou, X., Luo, Y., Yang, Y., Fang, C., Chen, J. and Li, B.: Minor stimulation of soil carbon storage by nitrogen addition: A meta-analysis, Agriculture, Ecosystems & Environment, 140(1-2), 234–244, doi:10.1016/j.agee.2010.12.010, 2011.

10 Magnani, F., Mencuccini, M., Borghetti, M., Berbigier, P., Berninger, F., Delzon, S., Grelle, A., Hari, P., Jarvis, P. G., Kolari, P., Kowalski, A. S., Lankreijer, H., Law, B. E., Lindroth, A., Loustau, D., Manca, G., Moncrieff, J. B., Rayment, M., Tedeschi, V., Valentini, R. and Grace, J.: The human footprint in the carbon cycle of temperate and boreal forests, Nature, 447(7146), 849–851, doi:10.1038/nature05847, 2007.

Marcolla, B., Cescatti, A., Manca, G., Zorer, R., Cavagna, M., Fiora, A., Gianelle, D., Rodeghiero, M., Sottocornola, M. and
15 Zampedri, R.: Climatic controls and ecosystem responses drive the inter-annual variability of the net ecosystem exchange of an alpine meadow, Agricultural and Forest Meteorology, 151(9), 1233–1243, doi:10.1016/j.agrformet.2011.04.015, 2011.

Neff, J. C., Townsend, A. R., Gleixner, G., Lehman, S. J., Turnbull, J. and Bowman, W. D.: Variable effects of nitrogen additions on the stability and turnover of soil carbon, Nature, 419(6910), 915–917, 2002.

Phoenix, G. K., Emmett, B. A., Britton, A. J., Caporn, S. J. M., Dise, N. B., Helliwell, R., Jones, L., Leake, J. R., Leith, I. D.,
20 Sheppard, L. J., Sowerby, A., Pilkington, M. G., Rowe, E. C., Ashmore, M. R. and Power, S. A.: Impacts of atmospheric nitrogen deposition: responses of multiple plant and soil parameters across contrasting ecosystems in long-term field experiments, Global Change Biology, 18(4), 1197–1215, doi:10.1111/j.1365-2486.2011.02590.x, 2012.

Pinheiro, J. C. and Bates, D. M.: Unconstrained parametrizations for variance-covariance matrices, Statistics and Computing, 6(3), 289–296, 1996.

25 Poorter, H., Niklas, K. J., Reich, P. B., Oleksyn, J., Poot, P. and Mommer, L.: Biomass allocation to leaves, stems and roots: meta-analyses of interspecific variation and environmental control: Tansley Review, New Phytologist, 193(1), 30–50, doi:10.1111/j.1469-8137.2011.03952.x, 2012.

R Development Core Team: R: A Language and Environment for Statistical Computing. Vienna, Austria, 2011, URL http://www. R-project. org, 2011.

30 Reeder, J. D. and Schuman, G. E.: Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands, Environmental Pollution, 116(3), 457–463, doi:10.1016/S0269-7491(01)00223-8, 2002.

Rihm, B. and Kurz, D.: Deposition and critical loads of nitrogen in Switzerland, Water, Air, and Soil Pollution, 130(1-4 III), 1223–1228, doi:10.1023/A:1013972915946, 2001.

Schimel, J. P. and Weintraub, M. N.: The implications of exoenzyme activity on microbial carbon and nitrogen limitation in soil: A theoretical model., Soil Biology & Biochemistry, 35(4), 549–563, doi:10.1016/S0038-0717(03)00015-4, 2003.

Sillen, W. M. A. and Dieleman, W. I. J.: Effects of elevated CO_2 and N fertilization on plant and soil carbon pools of managed grasslands: a meta-analysis, Biogeosciences, 9(6), 2247–2258, doi:10.5194/bg-9-2247-2012, 2012.

Sitch, S., Huntingford, C., Gedney, N., Levy, P. E., Lomas, M., Piao, S. L., Betts, R., Ciais, P., Cox, P., Friedlingstein, P., Jones, C. D., Prentice, I. C. and Woodward, F. I.: Evaluation of the terrestrial carbon cycle, future plant geography and climate-carbon cycle feedbacks using five Dynamic Global Vegetation Models (DGVMs), Glob. Change Biol., 14(9), 2015–2039, doi:10.1111/j.1365-2486.2008.01626.x, 2008.

Skinner, R. H.: Nitrogen fertilization effects on pasture photosynthesis, respiration, and ecosystem carbon content, Agriculture, Ecosystems & Environment, 172, 35–41, doi:10.1016/j.agee.2013.04.005, 2013.

Smith, J., Smith, P., Wattenbach, M., Zaehle, S., Hiederer, R., Jones, R. J. A., Montanarella, L., Rounsevell, M. D. A., Reginster, I. and Ewert, F.: Projected changes in mineral soil carbon of European croplands and grasslands, 1990–2080, Global Change Biology, 11(12), 2141–2152, doi:10.1111/j.1365-2486.2005.001075.x, 2005.

Soussana, J. F. and Hartwig, U. A.: The effects of elevated CO_2 on symbiotic N_2 fixation: a link between the carbon and nitrogen cycles in grassland ecosystems, Plant and Soil, 187(2), 321–332, 1996.

Stone, M. M., Weiss, M. S., Goodale, C. L., Adams, M. B., Fernandez, I. J., German, D. P. and Allison, S. D.: Temperature sensitivity of soil enzyme kinetics under N-fertilization in two temperate forests, Global Change Biology, 18(3), 1173–1184, doi:10.1111/j.1365-2486.2011.02545.x, 2012.

Vingarzan, R.: A review of surface ozone background levels and trends, Atmospheric Environment, 38(21), 3431–3442, doi:10.1016/j.atmosenv.2004.03.030, 2004.

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., Schlesinger, W. H. and Tilman, D. G.: Human alteration of the global nitrogen cycle: sources and consequences, Ecological applications, 7(3), 737–750, 1997.

Volk, M., Geissmann, M., Blatter, A., Contat, F. and Fuhrer, J.: Design and performance of a free-air exposure system to study long-term effects of ozone on grasslands, Atmospheric Environment, 37(9-10), 1341–1350, doi:10.1016/S1352-2310(02)01012-9, 2003.

Volk, M., Bungener, P., Contat, F., Montani, M. and Fuhrer, J.: Grassland yield declined by a quarter in 5 years of free-air ozone fumigation, Global Change Biology, 12(1), 74–83, doi:10.1111/j.1365-2486.2005.01083.x, 2006.

Volk, M., Obrist, D., Novak, K., Giger, R., Bassin, S. and Fuhrer, J.: Subalpine grassland carbon dioxide fluxes indicate substantial carbon losses under increased nitrogen deposition, but not at elevated ozone concentration: Grassland CO_2 flux under O_3 and N deposition, Global Change Biology, 17(1), 366–376, doi:10.1111/j.1365-2486.2010.02228.x, 2011.

Volk, M., Wolff, V., Bassin, S., Ammann, C. and Fuhrer, J.: High tolerance of subalpine grassland to long-term ozone exposure is independent of N input and climatic drivers, Environmental Pollution, 189, 161–168, doi:10.1016/j.envpol.2014.02.032, 2014.

Waldrop, M. P., Zak, D. R., Sinsabaugh, R. L., Gallo, M. and Lauber, C.: Nitrogen deposition modifies soil carbon storage through changes in microbial enzymatic activity, Ecological Applications, 14(4), 1172–1177, doi:10.1890/03-5120, 2004.

Yuan, Z.-Y., Li, L.-H., Han, X.-G., Chen, S.-P., Wang, Z.-W., Chen, Q.-S. and Bai, W.-M.: Nitrogen response efficiency increased monotonically with decreasing soil resource availability: a case study from a semiarid grassland in northern China, Oecologia, 148(4), 564–572, doi:10.1007/s00442-006-0409-0, 2006.

Table 1

	L-ratio	p-value
Block	10.5	0.005
O_3	0.3	0.582
N	48.2	0.000
$O_3 \times N$	2.1	0.151
$N \times N$	5.4	0.020

Table 1 Split plot analysis of cumulative seven year plant C yield. Effects of block, ozone (O_3), nitrogen (N) and selected interactions.

Table 2

	L-ratio	p-value
Block	1.8	0.410
O_3	0.0	0.859
N	0.7	0.387
$O_3 \times N$	0.0	0.995

Table 2 Split plot analysis of cumulative seven year net ecosystem productivity (NEP). Effects of block, ozone (O_3), nitrogen

(N) and $O_3 \times N$ interaction.

Table 3

	L-ratio	p-value
Block	1.36	0.51
O_3	0.58	0.45
N	0.25	0.62
$O_3 \times N$	0.47	0.49
$N \times N$	2.85	0.09
$O_3 \times N \times N$	0.02	0.88

Table 3 Split plot analysis of change of soil organic carbon concentration (0-10 cm layer) between 2003 and 2010. Effects of block, ozone (O_3), nitrogen (N) and selected interactions.

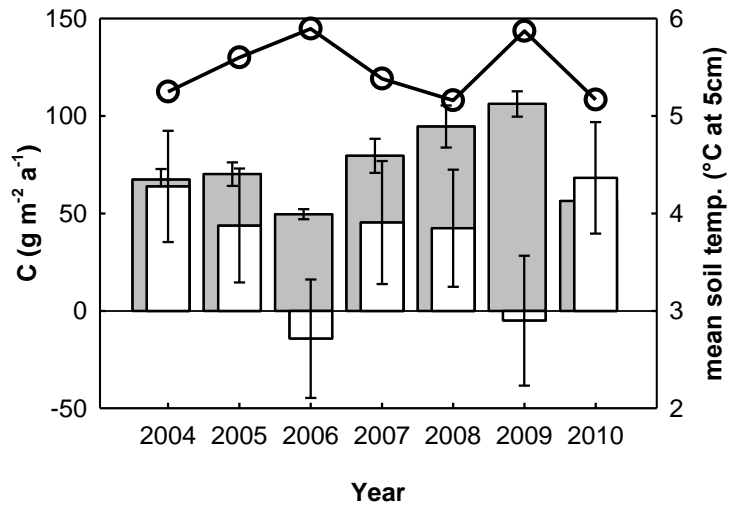
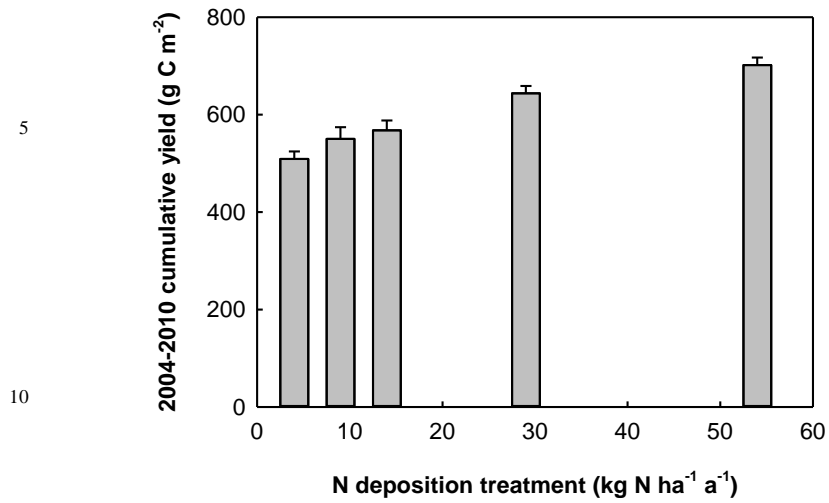



Figure 1 Aboveground plant yield (grey bars) and net ecosystem productivity (NEP; white bars) for the control treatment, and annual mean soil temperature at 5 cm depth (T_{soil}; solid line) for each year during the experiment. Soil temperature accuracy is ±0.18° C (within year systematic error for mean of nine probes used).

15 **Figure 2** Seven year cumulative aboveground plant C yield (O_3 treatments lumped), plotted against total annual N deposition of 4, 9, 14, 27 and 54 kg N ha^{-1} yr^{-1} (atmospheric background plus experimental deposition rates).

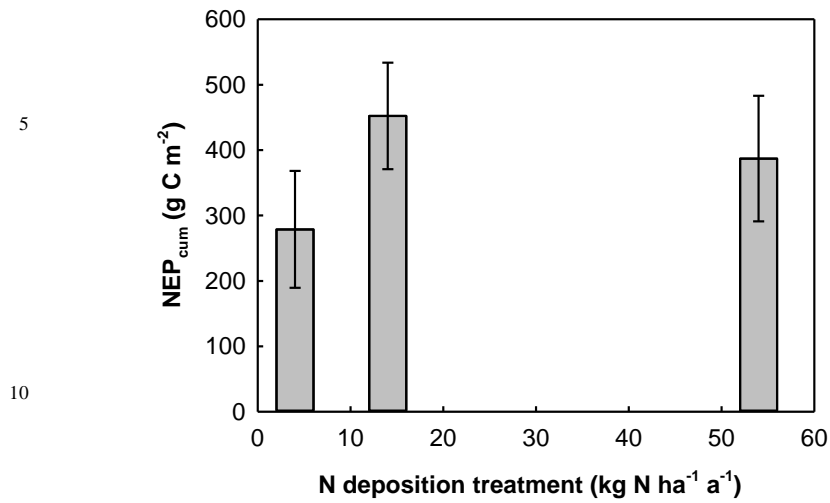
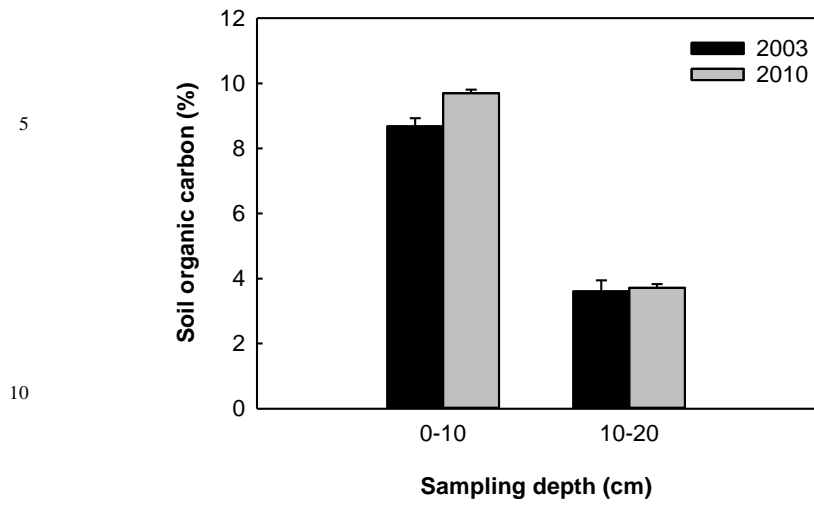



Figure 3 Seven year cumulative NEP carbon gain (O₃ treatments lumped), plotted against total annual N deposition of 4, 14 and 54 kg N ha⁻¹ yr⁻¹ (atmospheric background plus experimental deposition rates).

 Figure 4 Bulk soil (control treatment) organic carbon concentration in 2003 (grey bars) and 2010 (black bars) in the 0-10

cm and 10-20 cm soil layer.

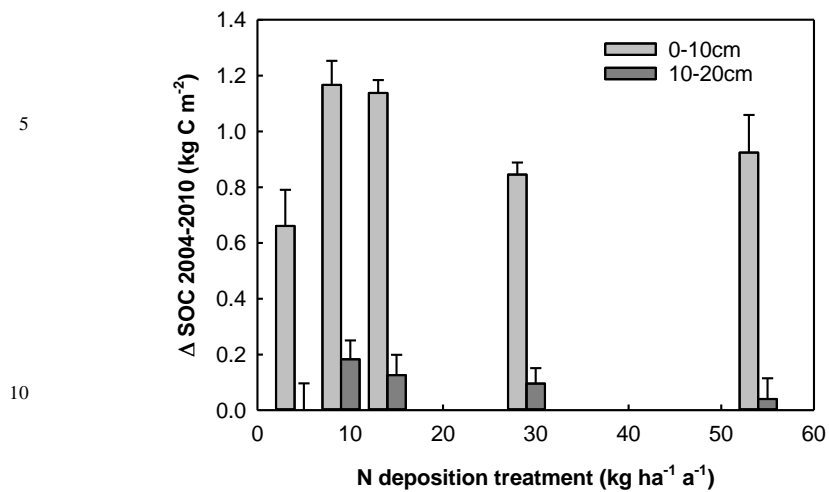



Figure 5 Absolute soil organic C stock gains during seven years (2003-2010) in the 0-10 cm (grey bars) and the 10-20 cm layer (black bars). O_3 treatments are lumped in N-treatment levels of 4, 9, 14, 27 and 54 kg N ha^{-1} yr^{-1} (atmospheric background plus experimental deposition rates).