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Impact of past and present land-management on the C-balance of a grassland in the Swiss Alps

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Abstract

Grasslands cover about 40% of the ice-free global terrestrial surface, but their quantitative importance in global carbon exchange with the atmosphere is still highly uncertain, and thus their potential for carbon sequestration remains speculative. Here, we report on CO_2 exchange of an extensively used mountain hay meadow and pasture in the Swiss pre-Alps on high-organic soils (7–45% C by mass) over a 3-year period (18 May 2002–20 September 2005), including the European summer 2003 heat-wave period. During all 3 years, the ecosystem was a net source of CO_2 (116–256 $g\,C\,m^{-2}\,yr^{-1}$). Harvests and grazing cows (mostly via C export in milk) further increased these C losses, which were estimated at 355 $g\,C\,m^{-2}\,yr^{-1}$ during 2003 (95% confidence interval 257–454 $g\,C\,m^{-2}\,yr^{-1}$). Although annual carbon losses varied considerably among years, the CO_2 budget during summer 2003 was not very different from the other two summers. However, and much more importantly, the winter that followed the warm summer of 2003 observed a significantly higher carbon loss when there was snow ($133 \pm 6\,g\,C\,m^{-2}$) than under comparable conditions during the other two winters (73 ± 5 and $70 \pm 4\,g\,C\,m^{-2}$, respectively). The continued annual C losses can most likely be attributed to the long-term effects of drainage and peat exploitation that began 119 years ago, with the last significant drainage activities during the Second World War around 1940. The most realistic estimate based on depth profiles of ash content after combustion suggests that there is an 500–910 $g\,C\,m^{-2}\,yr^{-1}$ loss associated with the decomposition of organic matter. Our results clearly suggest that putting efforts into preserving still existing carbon stocks may be more successful than attempts to increase sequestration rates in such high-organic mountain grassland soils.

Keywords: CARBOMONT, CO_2 exchange, drainage, eddy covariance flux measurements, land-management, mountain regions, pastoral grazing ecosystems, peatland, respiration

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Introduction

Carbon cycling in terrestrial ecosystems has not only attracted considerable attention by scientists in recent years, but also that of policy makers who are mainly interested in its potential for sequestering atmospheric CO_2 (IPCC, 2000; Rosenberger & Izauralde, 2001). The general understanding is that it might be possible to increase the quantity of organic matter in ecosystems and soils via appropriate management practices. This could be achieved by either offsetting a proportion of the anthropogenic fossil fuel CO_2 emissions, or via prevention of the decomposition of soils with high-organic carbon contents (Drewitt *et al*., 2002; Leifeld *et al*., 2005). To account for carbon sequestration in national greenhouse gas budgets, reliable information on current carbon stocks is essential, and potential sinks and changes in soil carbon must be measurable and verifiable (Smith, 2004). Leahy *et al*. (2004) have shown that grasslands can be either a source or sink of CO_2. Novick *et al*. (2004) collected information on annual

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grassland net ecosystem exchange (NEE) estimates based on eddy covariance (EC) measurements and the Bowen Ratio Energy Balance technique and reported values varying from a net source of $+400\,\mathrm{g\,C\,m^{-2}}$ to a net sink of $-88\,\mathrm{g\,C\,m^{-2}}$. Two reviews of available data (Janssens *et al*., 2003; Jones & Donelly, 2004) have shown that large uncertainties remain in resolving whether grassland ecosystems function as a CO_2 source or sink. This uncertainty is primarily attributable to the sensitivity of grasslands to interannual climate variability and associated biomass dynamics (Meyers, 2001; Flanagan *et al*., 2002) and incomplete process understanding of grassland assimilation and respiration (Nösberger *et al*., 2006).

The objective of this study that was part of the European CARBOMONT project was to quantify the carbon budget of a managed pre-Alpine grassland which is used as combined pasture and meadow, representing the traditional Swiss system of mountain agriculture. In particular, the NEE and its response to short-term climate variability was of primary interest to assess whether changes in land-management practices have the potential to increase the ecosystem carbon sink, or whether measures to reduce or stop decomposition of the high-organic matter of drained soils could be more efficient to positively affect the national greenhouse gas budget.

To achieve this goal, we selected the Rigi Seebodenalp site, an extensively used grassland in the Swiss pre-Alps on high-organic soils that were drained for agricultural use starting in the late 19th century. This type of historical land-use change from blanket peat bog wetlands that were drained for agricultural use that began in the mid-19th century is very typical for Switzerland and is considered to influence the local and regional climate well beyond the extent of the drained areas (see Schneider & Eugster, 2007 and references therein).

Material and methods

Site description

The Rigi Seebodenalp flux site was established in May 2002 as part of the CARBOMONT network and collected data until September 2005. This subalpine grassland is located on a flat shoulder terrace of Mount Rigi (47°03′31″N, 8°27′27″E) in Central Switzerland (Fig. 1) at an altitude of 1025 m above sea level (a.s.l.). A high spatial variability in soil properties and in plant composition is found at Rigi Seebodenalp. A detailed description of the site can be found in Rogiers *et al*. (2005) and in Rogiers (2005).

The wetland area (Fig. 1) is statutory protected and the vegetation is mown once per year in late summer, at the end of the growing season. The grassland area (Fig. 1) is currently used as a hay meadow followed by a pasture in late summer. The timing and intensity of land-management at Rigi Seebodenalp differed among years (Table 1). The first cut was always carried out in early to mid-June.

Of special interest is the drainage history of Rigi Seebodenalp because of its possible profound influence on the development of the soil, on the vegetation, on current land-management practices, and consequently also on the CO_2 exchange of the site. The current terrain

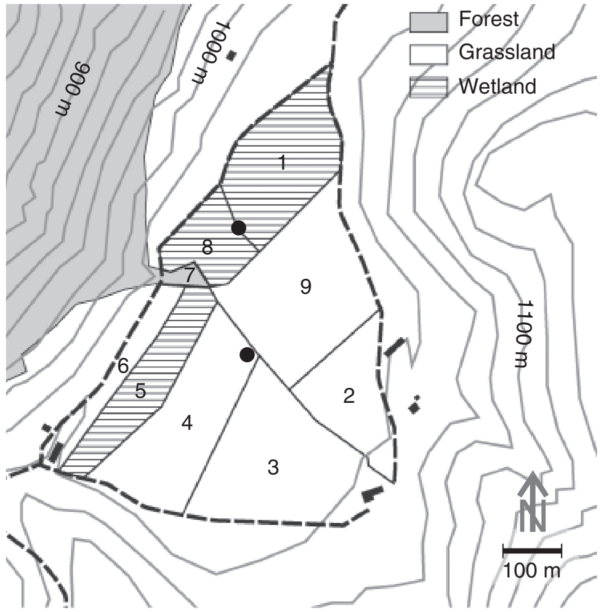

Fig. 1 Topographical map of Rigi Seebodenalp, a flat area on the north slope of mount Rigi (contour interval 20 m). The individual fields with uniform land use are shown with thin boundaries and numbers. Broken lines indicate unpaved private access roads. The position of the continuous grassland EC tower (field 4) and the episodic wetland tower to the northeast are shown with filled circles. The flux footprint area of the grassland tower used here covers fields 2, 3, 4 and 9.

Table 1 Timing of the first and second grass cut at Rigi Seebodenalp 2002–2005 at fields 4 and 9 at the grassland area (Fig. 1)

	Grass cut 1		Grass cut 2	
Year	Date	DoY	Date	DoY
2002	11 June	162	11 August	223
2003	11 June	162	31 July	212
2004	6 June	158	17 July	197
2005	16 June	168	–	–

In 2004, only half of field 9 was harvested.

is the bottom of a former but vanished lake formed during the last glaciation (Vogel & Hantke, 1989) with a thick sedge peat layer on top. The main drainage of peatlands in Switzerland was carried out in two phases from 1885 to 1949 (Eidgenössisches Meliorationsamt Bern, 1954). Cultivation of Rigi Seebodenalp was started in 1886 with the digging of the first drainage channels through the site (Wyrsch, 1988). During the Second World War, the drainage was intensified. The lowest part of Rigi Seebodenalp (hatched fields in Fig. 1) is still the wettest area, which is now statutory protected as a wetland. In this part, the peat layer is still thick, whereas in the other parts of Rigi Seebodenalp the soil has developed to a normal organic soil. The soils can be classified as folic Histosols (drystic) in the wetland and as stagnic Cambisols in the grassland area (Müller, 2004) – classification according to the World Reference Base for Soil Resources (WRB, 1998). Especially high concentrations of soil organic carbon characterize the site. At the grassland site, the soil organic carbon content amounts to 7–15% by mass, which is substantially lower than that of the wetland (20–45%).

Eddy covariance fluxes

CO_2 and water vapor fluxes were measured with the EC technique and calculated as described in Rogiers *et al*. (2005). There, detailed information on the data screening, filtering, data correction and gapfilling procedure of the EC data can be found together with a list of instruments used for the micrometeorological measurements. Here, we restrict ourselves to a brief summary. NEE was measured at the EC tower (Fig. 1) using an open-path infrared gas analyzer (IRGA; LI-7500, LI-COR Inc., Lincoln, NE, USA) in combination with a three-dimensional ultrasonic anemometer–thermometer (Solent HS, Solent-Gill Ltd, Lymington, UK). A positive flux-sign indicates a net loss of CO_2 from the site, whereas negative signs denote net uptake. In addition to what was done by Rogiers *et al*. (2005), we recomputed all net fluxes with an additional heat flux correction term (Burba *et al*., 2006a, b) due to instrument heating. The 'final' flux F_c obtained after damping loss correction (Eugster & Senn, 1995) and the Webb *et al*. (1980) density flux correction was further modified to yield

$$F_{c,corr} = F_c + \xi \frac{(T_s - T_a) q_c}{r_a (T_a + 273.15)} \left(1 + 1.6077 \frac{\rho_v}{\rho_d}\right), \quad (1)$$

where ξ is the fraction (range 0–1) of the heat flux produced by the open-path instrument relevant for the correction, T_s and T_a are instrument surface and air temperatures, respectively (°C), q_c is mean ambient CO_2 density ($\mu mol\, m^{-3}$), r_a is aerodynamic transfer resistance ($s\, m^{-1}$), and ρ_v and ρ_d are densities of water vapor and dry air ($kg\, m^{-3}$), respectively. The empirical approximation $T_s = 0.0025 T_a^2 + 0.90 T_a + 2.07$ (Burba *et al*., 2006a) was used, and for ξ an empirically determined value of 0.05 was chosen.

NEE was partitioned into gross primary production (GPP) and ecosystem respiration (R_{eco}) using a light response curve approach. Briefly, for days where there was a clear response of NEE to photosynthetically active radiation, total R_{eco} was calculated as the weighted average of dark night-time respiration and daytime respiration. Mean night-time (photosynthetic photon flux density PPFD $< 10\ \mu mol\, m^{-2}\, s^{-1}$) CO_2 fluxes were assumed to represent ecosystem respiration throughout the night (Wofsy *et al*., 1993; Schmid *et al*., 2000). Daytime respiration was determined by fitting a rectangular hyperbolic light response curve (Ruimy *et al*., 1995; Gilmanov *et al*., 2003) to the data points for each day when the vegetation was photosynthetically active. Gaps in daytime and night-time respiration were fitted by statistical interpretation using a 3-days relationship between night-time NEE and shallow soil temperature (5 cm depth) (e.g. Schmid *et al*., 2000). Daily GPP was then calculated as the difference between NEE and R_{eco}. The uncertainty of annual means is based on the standard error of the mean daily fluxes.

Footprint areas of the flux measurements were determined with the Kormann & Meixner (2001) footprint model as detailed in Rogiers *et al*. (2005). Results showed that the footprint was mostly restricted to the respective fields in the upwind direction.

Meteorological data

Additional meteorological data for the period January 1992 to June 2005 were provided by the National Air Pollution Monitoring Network (NABEL). This station is located about 1000 m NNE of the CARBOMONT flux site (47°04′10″N, 8°27′56″E, 1030 m a.s.l.). The years 1992–2001 were used as the reference period to which the years 2002–2005 can be compared. Precipitation data were only available for 1994–2005. Snow data were provided by the nearby measurement station Oberiberg (1090 m a.s.l.). The quantitative snow data from this station, operated by the Swiss Federal Institute for Forest, Snow and Landscape Research (WSL), correspond rather well with the qualitative information on snow cover provided by local people from Rigi Seebodenalp, and from this we conclude that Rigi Seebodenalp experienced the same snow events as the Oberiberg station.

Air temperatures in 2002 and 2003 were above normal (+0.64 and +1.06 °C, respectively; Table 2, Fig. 2). The year 2003 was the warmest year in many places in

Central Europe since the beginning of instrumental observations (see, e.g. Luterbacher *et al*., 2004; Schär *et al*., 2004; Ciais *et al*., 2005). The year 2002 was the wettest year since 1994, while 2003 was the driest. Rainfall distribution for the year 2002 shows some deviation from the average in which the rainfall surplus occurred in the second half-year. In contrast, 2003 was dry from the beginning, and the precipitation deficit increased monotonically until the end of the year. Volumetric soil moisture content (SWC, measured with ECH_2O-20 dielectric probes; Decagon Devices, Pullman, WA, USA), however, shows the high buffering capacity of the soil and that only shows very dry conditions from August to early September 2003 (Fig. 3). The year 2004 started normally with respect to precipitation and accumulated a precipitation deficit in the second half-year, which, however, did not strongly affect SWC. The first half of 2005 was again relatively dry, and SWC responded in a similar way as in late summer 2003, but much earlier in the year.

Table 2 Basic climatological values for Rigi Seebodenalp

Variable	1992–2001	2002	2003	2004
Temperature (°C)	7.32	7.96	8.38	6.95
Total precipitation (mm)	1327*	1746	991	1111
Days with snow cover	–	109	126	144

Data were taken from the nearby NABEL (Swiss National Air Quality Network) site.
*Mean of years 1994–2001.

Total carbon budget

To estimate the temporal change in soil organic carbon for a managed grassland, different components have to be considered: (1) the CO_2 exchange between ecosystem and atmosphere measured with the EC system; the carbon which is (2) exported by harvesting grass and (3) by weight gain and milk production of grazing cattle, and finally (4) the addition of carbon by manure applications. In the present case, there was no manure import to the grassland besides the droppings of grazing cattle. Carbon losses caused by the water cycle [i.e. dissolved organic carbon (DOC) or dissolved inorganic carbon (DIC)] were not investigated in detail based on the expectation that these losses should be relatively small. We will, therefore, use a preliminary value from a similar nearby site for completeness of our total carbon budget. The livestock influences the carbon budget in several ways. Besides the export of C by grazing (total weight gain and milk production), there are changes in the C-budget through excreta (in form of organic C, CO_2 and CH_4) and respiratory and ruminal fermentation losses from cattle (CO_2 and CH_4). The influence of grazing cattle on the C-budget was estimated based on literature data (BUWAL, 1998) using emission factors for different cattle age and size classes. A detailed

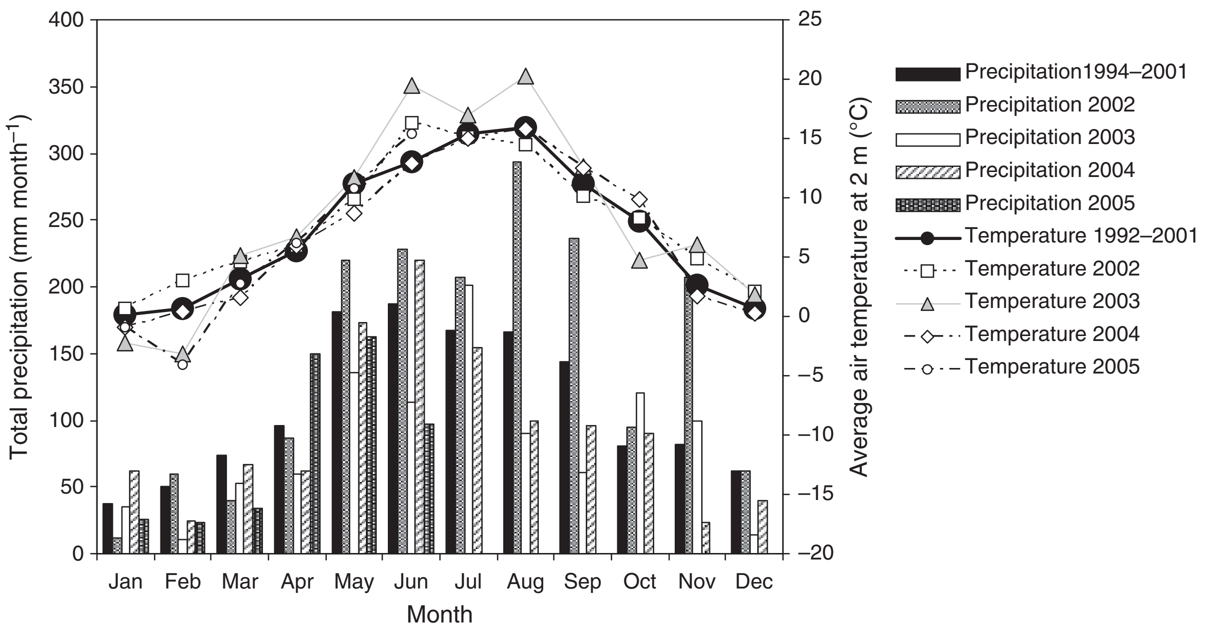

Fig. 2 Monthly average air temperatures at 2 m above ground (symbols and lines) and monthly total precipitation (bars) at Rigi Seebodenalp for the years 2002, 2003, 2004 and the first half of 2005 compared with the respective long-term average. Data were obtained from the Swiss National Air Quality Monitoring Network (NABEL).

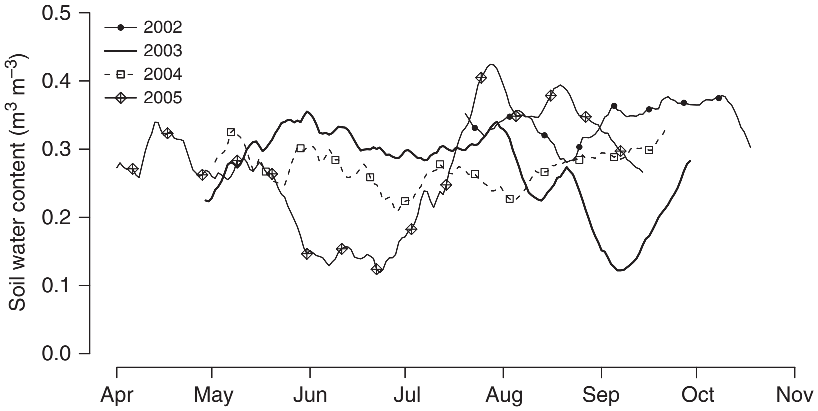

Fig. 3 Volumetric soil moisture content at −0.1 m depth, 18 May 2002–20 September 2005. Lines represent 14-day running averages of daily means. Values during winter when soils were snow-covered or frozen were screened out.

description of the calculations can be found in Studer (2003). The carbon export by grazing cattle was mainly due to milk production, which was estimated by average milk production per livestock unit (i.e. a cow weighing 600 kg and producing 3000 L of milk per year) multiplied by livestock density and total grazing period. In Switzerland, 1 L of milk contains 59 g C on average (BUWAL, 1998).

Estimation of historical soil carbon loss by soil core analysis

Besides climatic conditions and present land-management regime, additional site-specific factors can influence the carbon budget of a given site. The Rigi Seebodenalp site is strongly influenced by historical land-management (i.e. draining of the site). Integrated carbon losses since the beginning of drainage of the peat-containing areas were estimated from excess ash content in the upper peat layers. A rough estimate of net mean annual CO_2 emission can only be made under the following assumptions: (1) ash content (mass ash/mass dry peat) has been the same at all depths before drainage; (2) the peat surface started to oxidize after drainage and is continuing to do so; (3) oxidized C is mainly lost in form of CO_2, directly from the surface of the site; (4) ash from oxidized peat remains on site and accumulates in the surface layer and (5) ash content at greater depth is still the same today as before drainage.

Four replicate soil cores were sampled in the beginning of May 2005 near the center of field 8 (Fig. 1) within a radius of 3 m. According to the footprint analysis, field 8 was the major contributor to the EC measurements at the wetland area (Rogiers *et al.*, 2005). Samples were taken with a 5.4-cm-diameter corer with internal plastic liner (Giddings Machine Company, Windsor, CO, USA) to a depth between 50 and 60 cm. Compaction was corrected for, assuming linear compaction with depth. The ratio between the length of the compacted core and the depth of the hole left (which corresponds with the uncompacted soil core volume) was used in this correction step. Cores were cut into sections corresponding to uncompacted 3 cm depth intervals. These were dried at 40 °C for 96 h. Dry weight was measured and bulk density was calculated. For determining ash content, complete samples were pulverized in a wolfram mill and subsamples of around 3 g were combusted at 600 °C overnight. During the first few hours, oven temperature was lower (400–450 °C) to avoid hazardous conditions.

Integrated carbon losses since drainage (C lost; in $g\,m^{-2}$) were then estimated from (1) the mass of ash per unit surface area in the surface layer exceeding background mass of ash in deeper layers (excess ash mass; in $g\,m^{-2}$), (2) the proportion of ash in the background samples [background ash; (dimensionless)] and (3) the proportion of C in decomposed material. This was assumed to be the same as the proportion of C in material subjected to the loss-on-ignition method for organic C determination in peat samples (0.52, range 0.50–0.56; value taken from Bhatti & Bauer, 2002). Thus, carbon oxidation losses (C lost) were calculated as

$$\text{C lost} = \frac{\text{excess ash mass}}{\text{background ash}} \times 0.52. \qquad (2)$$

This calculation was done separately for each 3 cm layer of the upper 27 cm of each soil core. Average ash contents of the individual soil cores below 27 cm were considered background values, not yet influenced by drainage-induced oxidation. Estimated C losses in the upper 27 cm were then cumulated for every soil core, giving four replicate estimates from which mean and standard error for the site were estimated.

Results

Annual CO_2 losses due to historical land-management

Relatively high carbon losses were measured with the EC system at Rigi Seebodenalp (Table 3; Fig. 4). In order to evaluate the reliability of these results – which will be detailed later – and to assess the influence of historical land-management, the EC measurements were compared with our laboratory estimates of annual CO_2 losses from the wetland since drainage began. A soil pit and coring on field 8 indicated a peat layer thickness of over 1.5 m. From the surface to a depth of around 20 cm, peat was decomposed, crumbly, with no visible structure of the original plant material left and with a dark brownish-black color. Below, intact plant material of a lighter brownish color without clear signs of decomposition was found. This material was homogenous in structure and appeared from about 20 cm depth to about 1.5 m. The soil cores taken to a depth between 50 and 60 cm can thus be assumed representative for the upper 1.5 m. No trace of a mineral phase was found. It can be assumed that the top 1.5 m at this site is blanket peat, a Histosol that has received mostly atmospheric inputs.

Mean dry bulk density (Fig. 5, left panel) was around $0.16\,g\,cm^{-3}$ at 10–20 cm depth and only around half of that value below 27 cm. Mean ash contents (Fig. 5, right panel) decreased steadily from 13.3% at the surface to 3.4% at 27 cm depth. Below 27 cm depth, ash contents remained almost constant with mean values around 2.1%. Thus, ash contents below 27 cm were considered background values representative for ash contents in the entire profile before drainage and subsequent oxidation of the surface layers. Accumulation of excess ash in the upper 27 cm was calculated individually for each of the four cores and so was total C loss since drainage began.

Using Eqn (2), mean C loss since drainage began was estimated at $59.2 \pm 4.8\,kg\,C\,m^{-2}$ (mean ± SE, $n = 4$ soil cores). First drainage activities for the site were in 1886. Further improvements on drainage were made around 1940. Therefore, our lower and upper best estimates for mean annual C loss in form of CO_2 are 500 and $910\,g\,C\,m^{-2}\,yr^{-1}$ for a cultivation period of 119 and 65 years, respectively.

Three years of EC measurements

Data coverage. The performance of the EC system was good (90% availability), but the final data coverage of 46% for the 3-year measurement period after rigorous screening (e.g. Mauder et al., 2008) was only moderate. This is rather a low fraction as compared with the

Table 3 Best estimates and standard errors (based on daily data) of the cumulative budgets of net ecosystem exchange (NEE) of CO_2, ecosystem respiration (R_{eco}) and gross primary production (GPP) for the 3 years of measurements at Rigi Seebodenalp

Year	NEEa ($g\,C\,m^{-2}$)	R_{eco}a ($g\,C\,m^{-2}$)	GPPa ($g\,C\,m^{-2}$)	NEEs ($g\,C\,m^{-2}$)	R_{eco}s ($g\,C\,m^{-2}$)	GPPs ($g\,C\,m^{-2}$)	Fs ($g\,C\,m^{-2}\,day^{-1}$)	Ds (d)	TS5s (°C)
2002–2003	120 ± 36	1321 ± 46	−1201 ± 50	73 ± 5	131 ± 8	−57 ± 9	0.9 ± 0.1	78	1.83 ± 0.0
2003–2004	256 ± 42	1614 ± 60	−1358 ± 62	133 ± 6	180 ± 7	−47 ± 5	1.3 ± 0.1	100	1.42 ± 0.0
2004–2005	116 ± 44	1371 ± 57	−1253 ± 60	70 ± 4	103 ± 5	−31 ± 4	0.8 ± 0.1	90	0.88 ± 0.0

The total annual budgets (suffix a) from 18 May to 17 May of the following year, and the budgets during snow cover in winter (before 21 March; suffix s) are listed. Fs is the mean daily CO_2 flux from snow pack, Ds is the number of days with snow coverage and TS5s is the mean soil temperature under snow cover at 5 cm depth.

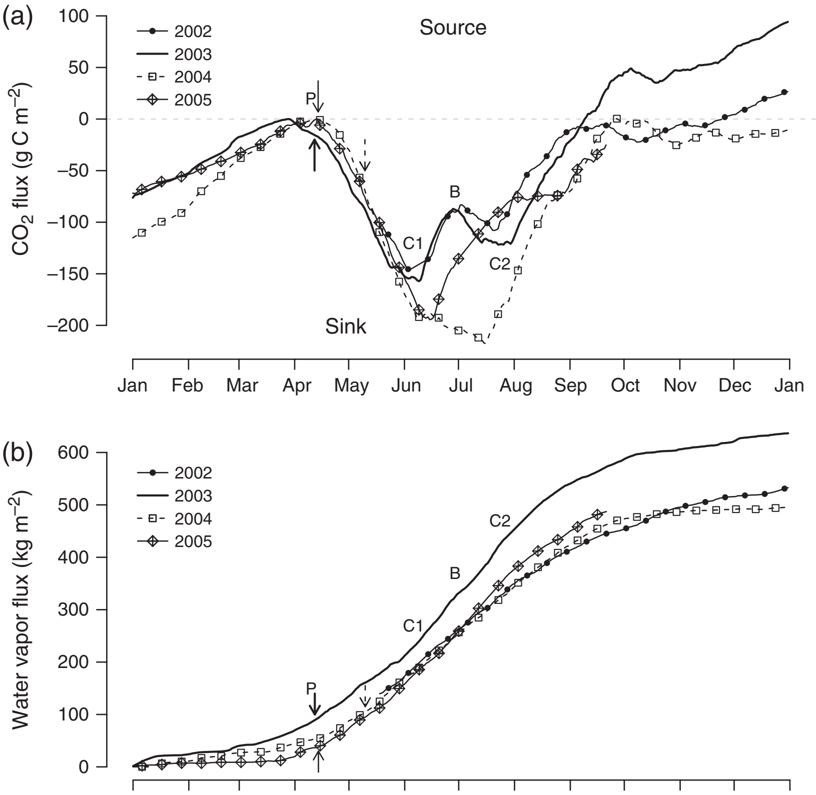

Fig. 4 Cumulative CO_2 (a) and water vapor fluxes (b) from 18 May 2002 to 20 September 2005 based on gapfilled data of the extensively used grassland at Rigi Seebodenalp. The cumulative CO_2 fluxes were drawn with respect to P, the first peak in the year, which is the end of the winter net carbon loss period. C1 and C2 refer to the first and second grass cut, respectively, within one growing season, and B indicates the date when the disturbed vegetation has regenerated after harvest. Water vapor fluxes are cumulative sums since 1 January of each year. The starting point of the curves from year 2002 corresponds to the mean of years 2003–2005. Arrows indicate the date by when the last late snow has melted away.

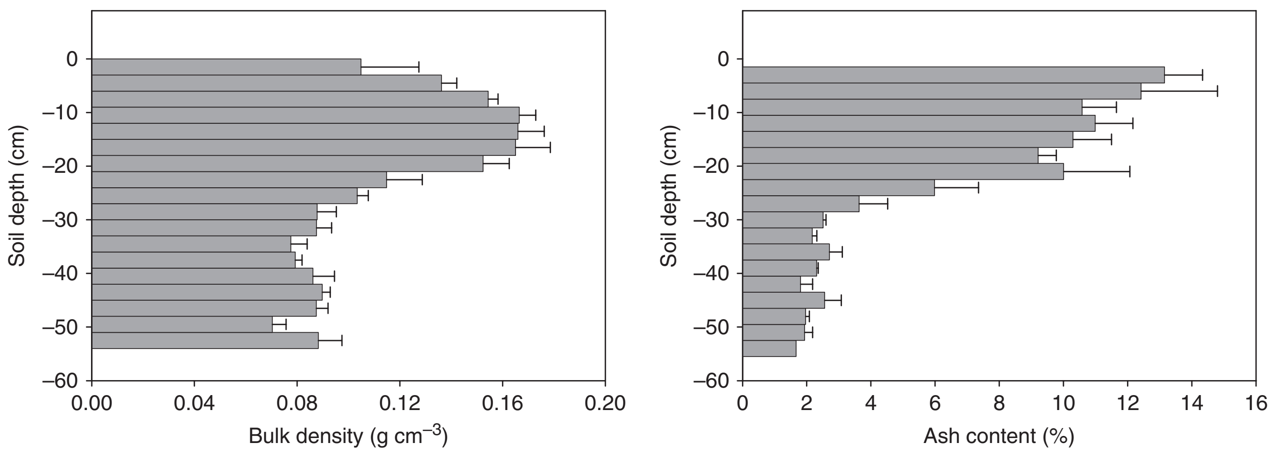

Fig. 5 Mean dry bulk density (left panel) of peat and mean ash contents (right panel) in dry material (in % by mass) from the surface to a depth of 54 cm ($n = 4$; bars indicate $\pm$ 1 SE).

average data coverage of 65% at the FLUXNET sites (Falge *et al.*, 2001), but comparable with the 45% reported from the Belgian FLUXNET site 'De Inslag' (Carrara *et al.*, 2004). In total, the data coverage was 46% high-quality data, 10% unavailability of instruments, 17% rejected due to rain, dew or snow on the open-path

IRGA and 27% rejected due to inadequate turbulence conditions (nonstationary turbulence conditions or momentum flux was not directed from the atmosphere towards the surface; Eugster *et al*., 2003). Gaps were filled using site-specific functional relationships between micrometeorological variables and CO_2 fluxes (Falge *et al*., 2001). According to Falge *et al*. (2001), a data coverage as low as 40% is still sufficient to produce defensible annual sums. Thus, we can assume that our dataset allows us to make a good estimate of the CO_2 and carbon budget of Rigi Seebodenalp.

Cumulative fluxes. The cumulative CO_2 and water vapor fluxes were calculated by summing up the gapfilled fluxes from 18 May 2002 until 20 September 2005 (Fig. 4). We followed the convention that positive fluxes indicate a net upward transport from the vegetation to the atmosphere, whereas negative values signify surface uptake. The cumulative curves of the CO_2 and water vapor fluxes measured at Rigi Seebodenalp between May 2002 and September 2005 (Fig. 4) show a similar seasonality in all years. However, a direct comparison of the 3 measurement years from June, the start of land-management at Rigi Seebodenalp, is somewhat complicated because of the differences in timing and magnitude in land-management interventions (Table 1).

In early spring, the cumulative CO_2 fluxes reach a peak (P in Fig. 4), meaning that the growing season started and assimilation and respiration were in equilibrium. The cumulative net CO_2 curves culminate later than the timing of snow melt, because at the beginning of the growing season, respiration losses still exceed assimilation uptakes, but late spring snowfall occurred in all years during the early season growth period (last event shown with arrows in Fig. 4).

After maximum carbon losses in spring and before the first grass cut (C1 in Fig. 4) in the beginning of June, the slopes of the integrated CO_2 fluxes had similar steepness, indicating that the increment in net daily CO_2 flux is rather constant during the three spring periods.

During the three growing seasons 2002, 2003 and 2004, the first grass cut (C1 in Fig. 4) at Rigi Seebodenalp was executed in the beginning of June. Cutting the grass considerably reduced the leaf area of the vegetation and thus also the assimilative capacity, resulting in net carbon losses from the grassland. In 2002 and 2003, the same fields situated in the daytime footprint were cut (Table 1), resulting in a similar pattern in the cumulative curves. At the breaking point (B in Fig. 4), plants had regenerated and a net carbon uptake was measured again. After the first cut in 2002, it took 23 days until a net CO_2 uptake was measured, in 2003, when it was warmer, the vegetation regenerated faster and a net uptake was already registered after 18 days. In 2004, only half of the fields lying in the daytime footprint were cut (Table 1). Therefore, there is only a reduction in the steepness of the cumulative courses visible (reduced sink strength compared with the end of May 2004), but the site still acted as a net sink for carbon.

The second grass cut in mid-summer (Table 1; C2 in Fig. 4) changed the direction of the curves again, and after that only carbon losses were measured. Additional interventions after the second grass cut like cattle grazing also contributed to the reduction in assimilation and thus to the persistence of losing carbon from the ecosystem.

Carbon losses subsided towards the end of September (Fig. 4). At that time, cattle were fetched away from the site. The vegetation could partially recover, inducing a higher photosynthetic activity under the same micrometeorological conditions as during the disturbances due to land-management. Moreover, the heat input into the grassland ecosystem was already strongly reduced compared with the months before, which resulted in smaller respiration rates and thus in smaller EC carbon losses.

During snow cover, there was no photosynthetic activity measured above the snow cover. The small values of GPP during snow cover (GPPs in Table 3) result from photosynthetic activity during snow melt periods were the footprint of the EC tower comprised an assemblage of snow-covered patches and areas with green vegetation. Substantial respiration losses from microbial activity were detected, due to favorable soil temperatures under the snow pack (TS5s in Table 3). The highest carbon losses were recorded from snow-covered grassland in the cold season 2003–2004 ($1.3 \pm 0.1\,g\,C\,m^{-2}\,day^{-1}$ or $133 \pm 6\,g\,C\,m^{-2}$ for 100 days with snow cover). The CO_2 flux from snow pack was on average less in winter 2002–2003 ($0.9 \pm 0.1\,g\,C\,m^{-2}\,day^{-1}$ or $73 \pm 5\,g\,C\,m^{-2}$ for 78 days with snow cover) and smallest in winter 2004–2005 ($0.8 \pm 0.1\,g\,C\,m^{-2}\,day^{-1}$ or $70 \pm 4\,g\,C\,m^{-2}$ for 90 days with snow cover).

The carbon budgets for the measurement years 2002–2003, 2003–2004 and 2004–2005, calculated by integrating the CO_2 flux data from the first day of measurements [i.e. 18 May until 17 May of the following year (Table 3)], have shown that the site is a net source of CO_2 during all 3 years. In 2002–2003 $120 \pm 36\,g\,C\,m^{-2}$, in 2003–2004 $256 \pm 42\,g\,C\,m^{-2}$ and in 2004–2005 $116 \pm 44\,g\,C\,m^{-2}$ was lost from the ecosystem.

In contrast to the cumulative CO_2 fluxes, the management interventions during the different growing seasons were not detectable in the total measured evapotranspiration (Fig. 4b). High evapotranspiration rates were measured during the growing season and nearly zero evapotranspiration rates were detected during the cold season. The highest amount of water vapor was lost via evaporation and transpiration in 2003–2004 ($581 \pm 7\,kg\,H_2O\,m^{-2}$) corresponding with the highest heat input in summer 2003 (Table 2; Fig. 2). In 2002–2003 $570 \pm 6\,kg\,H_2O\,m^{-2}$ and in 2004–2005 $486 \pm 6\,kg\,H_2O\,m^{-2}$ was lost from the ecosystem.

Total carbon budget for 2003

Figure 6 summarizes all relevant fluxes that contribute to the total carbon budget of year 2003. With the EC system, an annual net CO_2 loss of $172 \pm 40\,g\,C\,m^{-2}\,yr^{-1}$ was measured at Rigi Seebodenalp, the difference of $1447 \pm 61\,g\,C\,m^{-2}\,yr^{-1}$ GPP and $1619 \pm 61\,g\,C\,m^{-2}\,yr^{-1}$ R_{eco}. The carbon removed from the ecosystem by harvesting amounted to $152 \pm 30\,g\,C\,m^{-2}\,yr^{-1}$ and the presence of grazing livestock caused an additional carbon loss of $31 \pm 6\,g\,C\,m^{-2}\,yr^{-1}$. There was no manure application. Because we did not measure neither CH_4 fluxes nor DIC and DOC losses from the site, we present two budgets in Fig. 6. Budget I only includes measurements and best estimates based on local data, whereas budget II also includes somewhat speculative estimates of the two unmeasured components.

Our estimate of CH_4 fluxes is partially based on detailed measurements carried out by Jones *et al.* (2006) at a Scottish site. Based on initial estimates from soil incubation, we expect a small dominance of CH_4 oxidation over CH_4 production on an annual time scale, because soils are drained and often well-aerated at least in the topsoil. DIC and DOC losses (Fig. 6) were assumed to be comparable with the preliminary values obtained at a nearby site (courtesy of R. Kindler, J. Siemens, A. Heim) with similar quaternary deposits, similar climate and same elevation.

It should, however, be recalled that we assumed that peat decomposition is mainly increasing ecosystem respiration – which is included in EC flux measurements – for our estimate of C losses due to peat decomposition (see 'Estimation of historical soil carbon loss by soil core analysis'). If this is a correct assumption, then DIC/DOC losses should actually be negligibly low. If we, however, assume that an important fraction of the carbon losses from peat decomposition is lost via DIC and DOC export, then a relatively high loss could be expected, because our soil core-based peat loss estimate would then most likely be too low. Figure 6 expresses these uncertainties as follows. Each component of the budget is shown as a rectangle with an error bar that corresponds to the range of two standard errors or the 95% confidence interval. The uncertainties in the budget values I and II were modeled using Monte Carlo simulation ($N = 10\,000$). Because our estimate for peat loss due to management history is not a separate term in the budget, but helps explain the large annual net losses, we present two variants to the right of line A in Fig. 6: the upper as the reality and the lower as a hypothetical budget that excludes the special effect of peat decomposition. Without peat decomposition, the

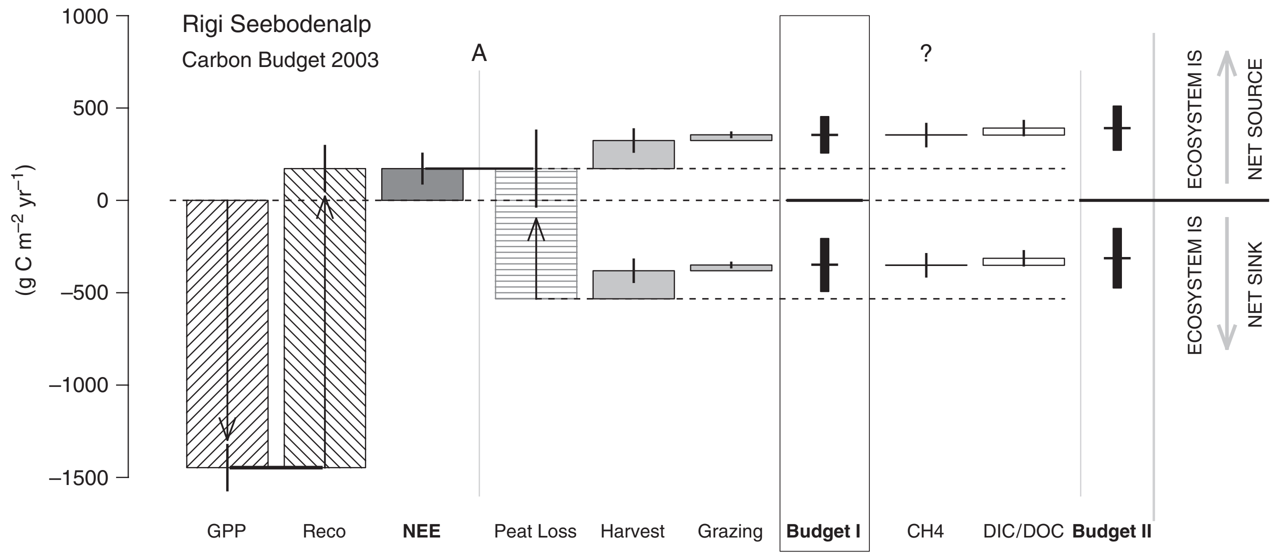

Fig. 6 Full budget of carbon fluxes to and from the site for year 2003. Eddy covariance-based measurements are shown left of line A. The first bar on the left (gross primary productivity) starts at zero, all subsequent bars begin where the previous bar ended (corresponding to 'stacked bars'). On the right hand side of line A, the lower variant shows the influence of peat losses to the overall budget, whereas the upper variant represents reality. Error bars show the 95% confidence intervals (mean ± 2 SE). See text for details.

annual budget II would fall in the range of net uptake similar to typical, well-fertilized grasslands and pastures. The measured conditions, however, show a best estimate for budget I of 355 $g\,C\,m^{-2}\,yr^{-1}$ (95% confidence interval 257–454 $g\,C\,m^{-2}\,yr^{-1}$), which might be even a larger loss of 391 $g\,C\,m^{-2}\,yr^{-1}$ (budget II; 95% confidence interval 271–513 $g\,C\,m^{-2}\,yr^{-1}$) if our speculative estimates for CH_4 and DIC/DOC fluxes are included. Similar additional losses can be expected from other years. which add to the measured NEE net losses (Table 3). Thus, it is highly certain that the Rigi Seebodenalp site was a net carbon source not only in 2003 but during the whole CARBOMONT project as well.

Discussion

Net CO_2 losses measured with the EC tower only account for half of the total carbon losses from the grassland ecosystem at Rigi Seebodenalp. Among all CARBOMONT sites, Rigi Seebodenalp is the strongest source of carbon (Frank Berninger, personal communication). CO_2 budgets (NEE) at other CARBOMONT sites ranged between a net C uptake of $-190\,g\,C\,m^{-2}$ (Amplero site, Italy) and a net source of $+80\,g\,C\,m^{-2}$ (Matra site, Hungary). The extreme case of Rigi Seebodenalp is thus indicative of specific conditions that differ from other sites. Because Rigi Seebodenalp is the only site with such high-organic soils, we focused on peat losses associated with the specific land-use history of the site as a possible explanation. Leifeld *et al*. (2003) provided an overview of available data on expected annual losses of organic carbon due to peat oxidation under similar Swiss climatic conditions and found a mean oxidative peat loss of 952 $g\,C\,m^{-2}\,yr^{-1}$. Our laboratory results with soil samples from Rigi Seebodenalp (500–910 $g\,C\,m^{-2}\,yr^{-1}$ loss) are thus in good agreement with literature data. The largest uncertainty in our estimated losses is caused by the fact that no detailed written documents exist on when and where the first drainage pipes were installed in 1886. Thus, it remains unclear, whether the renewal of the drainage system in the 1940s was a pure maintenance intervention (thus, the relevant time period to be considered starts in 1886), or whether this was an improvement and extension of the drainage system that could be considered the initial time point when peat oxidation started. In total, drainage has probably reduced peat thickness by 1.3 m, a value similar to that reported by Aerni (1980) for the Swiss Seeland region from 1880 to 1960 (see, e.g. Schneider & Eugster, 2007 for more details on these land-use changes).

With respect to the overall R_{eco} losses of 1435 $g\,C\,m^{-2}\,yr^{-1}$ (mean of 3 measurement years, see R_{eco}a in Table 3), the fraction of respiration that can be related to drainage activities varies between 35% and 63%. Thus, about half the respiratory carbon losses at Rigi Seebodenalp can be explained by historical land-management. Other possible contributors to the carbon losses are autotrophic respiration, soil respiration due to present land-management practices and respiration losses from grazing cattle. Advective losses which can be important components in forest ecosystem C-budgets are, however, not expected to be significant at such a site with short-statured vegetation (see, e.g. Hiller *et al*., 2008).

Peat decomposition from drained former wetlands is a slow and long-term process. Thus, the short-term influence observed during the summer 2003 heat wave is of interest, because it could have invalidated our hypothesis of peat losses had we measured large anomalies in respiration only during 2003 but not in other years. This is however not the case. Although Europe experienced an exceptionally hot summer 2003 (Luterbacher *et al*., 2004; Schär *et al*., 2004; Ciais *et al*., 2005), we did not observe any substantial deviation of the cumulative CO_2 flux-curves from year 2002 until early September (Fig. 4a). Although extremely high air temperatures were reached in summer 2003 and precipitation was generally lower than average, soil water stocks were still partially replenished by night-time rainfall events until early August (Fig. 3). Only in September, when SWC finally dropped below $0.15\,m^3\,m^{-3}$ could we see a marked difference in respiratory losses in comparison with other years (Fig. 4a). Before September, a reduction in photosynthetic capacity was observed, but also a decrease in daytime and dark respiration, such that the influence of drought on the net CO_2 flux was stinted. In this respect, Rigi Seebodenalp behaved similar to most other ecosystems affected by the heat anomaly, where drought not only decreased assimilation substantially but also ecosystem respiration (Reichstein *et al*., 2007). But as an important difference, NEE was almost unaffected during the drought. Increased respiration, however, followed in autumn and winter (September 2003–March 2004; Fig. 4a).

In particular, the special conditions observed in summer 2003 might have caused an increment in dead organic material providing additional substrate that did not directly increase respiration rates, but with a delay in the following winter (Franzluebbers *et al*., 2000). CO_2 losses from below the snow pack were 40–60% higher per day in winter 2003–2004 compared with losses in 2002–2003 and 2004–2005 (Fs, TS5s and R_{eco}s in Table 3). This finding is, however, partially confounded by the fact that the winter that followed the 2003 heat wave had a 10–30% longer duration of snow cover (Ds in Table 3) than the other two winters.

In contrast to CO_2 fluxes, our evapotranspiration measurements do not show any strong response to grass cuts (Fig. 4b). This suggests that evaporation of soil water must have increased after grass cuts, whereas transpiration must have been reduced due to the decrease in leaf area index caused by the grass cut. The increase in evaporation can be easily explained by the fact that a soil surface without dense canopy receives more net radiation and is heated up much more than a soil surface covered by plants, thereby having more energy available for evaporating water. Harvesting changed the contribution of transpiration and evaporation to the total measured water vapor fluxes in such a way that the magnitude of the total evapotranspiration flux before and after the grass cut did not change significantly.

Our results point to a phenomenon that could have severe consequences in the wider context of high carbon soils and global change. This is the delayed response of soil respiration to a warmer than usual season (cf. Barford *et al*., 2001; Hollinger *et al*., 2004). Although drought may somewhat restrict increases in respiration during particularly hot summers, there seem to be memory effects that can lead to massively increased respiration rates in the cold season following such an event. We do not know the mechanisms behind such effects but they seem much stronger than what we would expect from simple considerations of the short-term response of respiration to temperature (e.g. Q_{10} value of 2). The summer 2003 at our site was about 4.1 °C warmer than usual (18.9 °C vs. 14.8 °C). Nevertheless, it resulted in almost a doubling of net CO_2 loss from the snow-covered ecosystem during the following winter. Temperatures in arctic regions, where 20–30% of the global soil carbon is stored, are expected to increase by 0.8–2.6 °C during the next 50 years (Gundelwein *et al*., 2007). From this study, we have seen that it is particularly important to consider the possibility of delayed responses to unusual warming and to follow carbon fluxes in such ecosystems well beyond the duration of an extreme event.

Conclusions

EC fluxes measured at an extensively managed pre-Alpine grassland between 18 May 2002 and 20 September 2005 were presented. During all 3 years, the ecosystem acted as a net source of CO_2, although significant differences in the CO_2 budgets were found between the years.

Carbon losses due to historic drainage of the wetland were assessed by analyzing soil core composition. This yielded an annual net carbon loss estimate due to drainage of the wetland that ranged between 500 and 910 $g\,C\,m^{-2}\,yr^{-1}$. The relatively high carbon loss measured at Rigi Seebodenalp with the EC tower was unique among the CARBOMONT flux sites. Without the ongoing decomposition of the remaining drained peat biomass the site, however, would most likely have shown the typical net sink strength of grasslands (e.g. Gilmanov *et al*., 2007).

The annual carbon budgets were influenced by differences in land-management practices and intensities, as well as differences in weather conditions. With respect to temperature, the years 2002 and 2004 were close to the 10-year average, whereas year 2003 was clearly warmer and drier than average. The carbon budget during summer 2003 was not dramatically different from the other two summers, but significantly higher carbon losses were observed during the winter period 2003–2004. Besides the CO_2 exchange measured with the EC system, other components such as carbon export by livestock and harvesting are relevant when calculating the total carbon budget of a managed grassland. At Rigi Seebodenalp, such exports roughly doubled the carbon losses via CO_2 exchange that were measured by EC in 2003.

Our results provide evidence that carbon losses from decomposing peat are and will be a substantial and important component in the carbon budget of this grassland site on high-organic soils, whereas the intra-annual variation of NEE is much smaller. This suggests that measures to prevent further carbon losses resulting from ongoing decomposition of organic matter in the soils have a higher potential to reduce overall ecosystem carbon losses than could be expected from management changes that directly affect NEE.

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