Chapter 14
Eddy Covariance Measurements over Wetlands

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14.1 Introduction

Wetland ecosystems can be classified according to various systems, one of which defines three major groups: (1) northern peatlands (with a total area of 350×10^6 ha), (2) freshwater swamps and marshes (204×10^6 ha), and (3) coastal wetlands (36×10^6 ha) (Mitsch et al. 2009). Depending on the definition, wetlands cover 3–6% of the Earth's land surface. This chapter concentrates on northern peatlands, which constitute a highly important component of the global biogeochemical cycling, as these boreal and arctic mires have accumulated about one-third of the global organic soil carbon (Gorham 1991). Turunen et al. (2002) estimated the size of this carbon pool as 270–370 Tg C, while tropical peatlands, which are also a very important source of greenhouse gases (GHGs) to the atmosphere, are estimated to store about 50 Tg C in peat (Hooijer et al. 2006).

Peatlands in the boreal zone may originate from different processes, but the main prerequisite for their development is a surplus of water. Mires start to grow in lowlands where draining is poor, and usually precipitation exceeds evaporation (Kuhry and Turunen 2006). The poorly aerated conditions due to the high water table result in a slow decay of plant litter. The accumulation of carbon results mainly from the inhibited decomposition in anoxic conditions, rather than high photosynthetic uptake rates. On the other hand, the reduction reactions prevailing in these conditions lead to the microbially mediated production of CH_4 (Limpens et al. 2008).

Mires affect the radiative forcing of the atmosphere in two opposite ways: (1) cooling induced by the uptake of CO_2 from the atmosphere, acting on a timescale of millennia, and (2) warming induced by CH_4 emissions on a timescale of decades

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(Frolking et al. 2006). The stability of these large organic carbon masses largely depends on the hydrological conditions, which may alter as result of climate change (Griffis et al. 2000; Lafleur et al. 2003), and which also in turn depend upon the scale of human intervention, such as the draining of mires for agriculture and forestry.

The GHG exchange of mires has interested the scientific community for a long time, and different approaches have been used to measure the exchange rates. The long-term apparent accumulation of carbon in peat during the Holocene or over a shorter period reflects the CO_2 exchange and its net uptake (Clymo et al. 1998; Turunen et al. 2002; Schulze et al. 2002). Chamber measurements have provided important information on the GHG exchange on the scale of plant communities and on the environmental responses of these fluxes (Moore and Knowles 1990; Alm et al. 1999; Riutta et al. 2007). Utilization of the micrometeorological eddy covariance (EC) method has been a great step forward for the understanding of the present-day GHG exchange of wetlands, because EC measurements provide continuous nonintrusive observations on an ecosystem scale. It is nowadays feasible to make year-round EC measurements and thus obtain direct observations of the current annual GHG balances.

This chapter first provides a brief history of EC-based GHG studies and summarizes the current network of wetland measurement sites. As throughout the chapter, the emphasis of this site survey is placed on the natural northern mires, but examples are also shown of wetlands located in lower latitudes. After that, some special features related to the application of the EC technique in mire ecosystems are highlighted. This is followed by an outline of the ancillary and complementary measurements that would support the interpretation of EC flux data and a discussion of the additional challenges of carrying out flux measurements in the winter conditions encountered in northern latitudes. Finally, both the determination of the total carbon balance of an ecosystem and its related climate impacts are discussed as examples of the application of EC-based flux data.

14.2 Historic Overview

For the measurement of CO_2 fluxes using the EC technique, fast-response nondispersive infrared (NDIR) sensors have been available for about three decades. In present-day EC systems, the CO_2 fluxes are measured using perfected open- or closed-path instruments (Sect. 2.4). Compared to CO_2, the measurement of CH_4 is technically more difficult, due to its lower atmospheric concentration and less favorable absorption spectrum. Thus, the development of user-friendly analyzers for the EC field measurements of CH_4 fluxes has been slower, and has involved more diverse (laser absorption) techniques. The first of these instruments was based on the Zeeman-split HeNe laser (Fan et al. 1992), but the tunable diode laser (TDL) enabled a better selection of the absorption peak (Verma et al. 1992; Zahniser et al. 1995). While lead-salt TDL spectrometers were already commercially available in the 1990s, they suffered from instability and were laborious to maintain, as they require cooling by liquid nitrogen. The quantum cascade laser (QCL) spectrometers offered an alternative that was more stable and accurate (Faist et al. 1994; Kroon et al. 2007). More recently, with the introduction of the cavity ring-down method, the sensitivity and stability of the analyzers has improved markedly. Instruments based on QCL or narrow-band industrial lasers have also proved feasible for CH_4 detection at room temperature, making the maintenance of field measurement sites much more convenient (Hendriks et al. 2008).

The first micrometeorologists to study the ecosystem–atmosphere exchange of GHGs were originally already attracted by the remote wetlands that guaranteed the acquisition of original data. The first micrometeorological CO_2 flux measurements were conducted over wet meadow tundra in Alaska in 1971 by means of the gradient method (Coyne and Kelly 1975). The first wetland measurements of CO_2 and CH_4 fluxes with the EC method were made in 1988 at a tundra site in Alaska by Fan et al. (1992), who used an NDIR instrument for CO_2 and both a HeNe laser spectrometer and a total hydrocarbon detector for CH_4 concentrations. The first EC measurements of CO_2 above a raised bog were made in the Hudson Bay lowlands in July 1990 by Neumann et al. (1994). Within the same study, Edwards et al. (1994) measured CH_4 fluxes using a TDL-based instrument. On a more southern bog in Minnesota, Verma et al. (1992) demonstrated the applicability of their newly developed TDL sensor for the EC measurements of CH_4 fluxes. Multiyear measurements at the same site showed that an ecosystem can act either as a CO_2 sink or a source, depending on the meteorological and hydrological conditions during the growing season (Shurpali et al. 1995).

In Europe, EC measurements on mires started later than in North America. The first measurements of CO_2 fluxes were probably carried out on a disturbed bog in the Netherlands in 1994–1995 using commercial instruments (Nieveen et al. 1998). The first European measurements on a pristine wetland were conducted on a subarctic fen in 1995. The CO_2 fluxes from this measurement campaign were reported by Aurela et al. (1998) and the CH_4 fluxes by Hargreaves et al. (2001). Measurements at this site, Kaamanen, located in northern Finland, have continued since 1997. Another long-term (since 1998) EC-based time series of CO_2 fluxes has been collected on an ombrotrophic bog near Ottawa, Canada (Lafleur et al. 2001). These sites demonstrate the feasibility of the EC technique for continuous multiyear measurements in wetland ecosystems, providing data that have proved most useful for studying the environmental responses, also enabling consideration of climate change effects on the GHG exchange (Lafleur et al. 2003; Aurela et al. 2004).

During recent years, a growing number of EC measurements have been commenced on different wetland ecosystems. The first measurements within the vast wetland areas of northern Russia were conducted in 2000 by Arneth et al. (2002). The importance of these areas, especially in relation to the possible melting of permafrost soil, has subsequently served as a motivation for several EC-based studies (Corradi et al. 2005; Kutzbach et al. 2007; van der Molen et al. 2007; Sachs et al. 2008; Laurila et al. 2010). Figure 14.1 and Table 14.1 present a survey of wetland sites at which EC measurements have been conducted. The majority

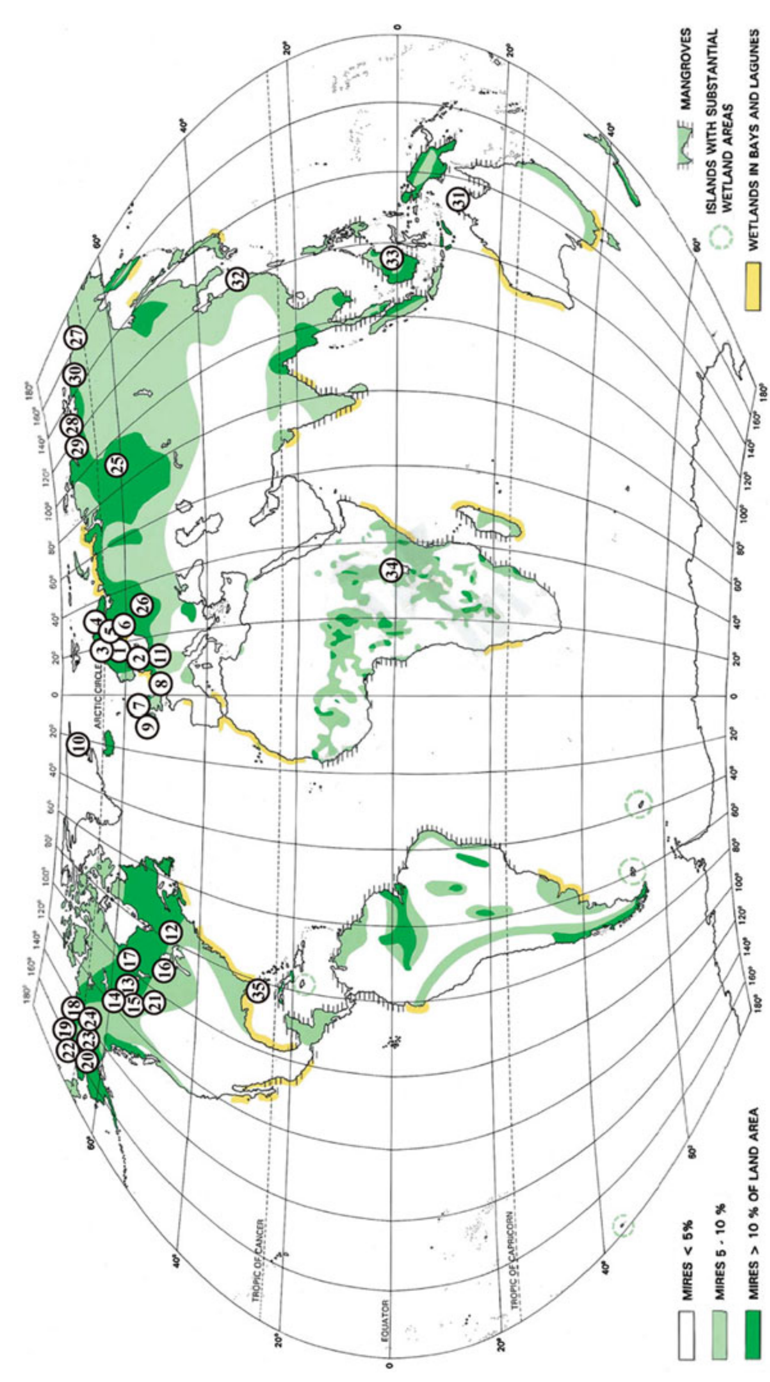

Fig. 14.1 Global distribution of mires (Lappalainen 1996) together with the locations of present and earlier eddy covariance measurement sites (*numbered circles*) in the world's various wetland ecosystems. The numbers refer to Table 14.1, in which additional information is presented for each site

Table 14.1 Eddy covariance measurement sites in various wetland ecosystems

ID	Site name	Coordinates	Seasonal coverage	Wetland type	Measured GHGs	Gas analyzer	Sonic anemometer	Reference
Europe								
1	Degerö	64°11′N, 19°33′E	A	Fen	CO_2	LI6262	Gill R2	Sagerfors et al. (2008)
2	Fäjemyr	56°15′N, 13°33′E	A	Bog	CO_2	LI6262	Gill R3	Lund et al. (2007)
3	Stordalen	68°20′N, 19°03′E	A	Fen	CO_2, CH_4	Aerodyne TDL, LI7500	Gill R3	Jackowicz-Korczyński et al. (2010)
4	Kaamanen	69°08′N, 27°17′E	A	Fen	CO_2, CH_4	LI7000, Aerodyne TDL, LGR RMT-200	ATI SWS-211, METEK USA-1	Aurela et al. (2004), Hargreaves et al. (2001)
5	Siikaneva	61°50′N, 24°11′E	A	Fen	CO_2, CH_4	LI7000, Campbell TGA100, LGR RMT-200	METEK USA-1	Rinne et al. (2007), Aurela et al. (2007)
6	Lompolojänkkä	67°60′N, 24°13′E	A	Fen	CO_2, CH_4	LI7000, LGR RMT-200	METEK USA-1	Aurela et al. (2009)
7	Auchencorth Moss	55°48′N, 3°15′W	A	Bog	CO_2, CH_4	LI7000, LGR RMT-200	Gill R2	Dinsmore et al. (2010)
8	Fochtelooer area	53°01′N, 6°24′E	A	Bog	CO_2	LI6262, LI7000	Gill R2	Nieveen et al. (1998)
9	Ireland	51°55′N, 9°55′W	A	Bog	CO_2	LI7500	Campbell 81000	Sottocornola and Kiely (2005)
10	Zackenberg	74°28′N, 20°34′W	GS	Tundra	CO_2	LI6262	Gill R2	Soegaard and Nordstroem (1999)
11	Rzecin	52°46′N, 16°31′E	A	Fen	CO_2	LI7500	Gill R3	Chojnicki et al. (2007)

(continued)

Table 14.1 (continued)

ID	Site name	Coordinates	Seasonal coverage	Wetland type	Measured GHGs	Gas analyzer	Sonic anemometer	Reference
North America								
12	Mer Bleue	45°25′N, 75°31′W	A	Bog	CO_2	LI6262, LI7000	Gill R2	Lafleur et al. (2003)
13	Canada-WP1	54°57′N, 112°28′W	GS	Treed fen	CO_2	LI7000	Gill R3	Syed et al. (2006)
14	Canada-WP2	55°32′N, 112°20′W	GS	Fen	CO_2	LI7500	Campbell CSAT3	Glenn et al. (2006)
15	Canada-WP3	54°28′N, 113°19′W	GS	Fen	CO_2	LI7500	Campbell CSAT3	Glenn et al. (2006)
16	Minnesota	47°32′N, 93°28′E	GS	Bog	CO_2, CH_4	LI6251, Unisearch Associates TDL	Kaijo-Denki, DA-600	Shurpali et al. (1995), Verma et al. (1992)
17	Manitoba	55°54′N, 98°24′W	GS	Fen	CO_2	LI6252	ATI SWS-211	Joiner et al. (1999)
18	Atqasuk	70°28′N, 157°25′W	A	Tundra	CO_2	LI7500	Gill R3	Kwon et al. (2006)
19	Barrow[a]	71°19′N, 156°38W	A	Tundra	CO_2	LI7500, Advanet E009a	Gill R3, Kaijo-Denki DA-600	Harazono et al. (2003), Kwon et al. (2006)
20	Ivotuk	68°29′N, 155°45′W	A	Tundra	CO_2	LI7500	Gill R3	Walter Oechel and Olaf Vellinga, personal communication (2011)
21	Saskatchewan	53°57′N, 105°57′E	GS	Fen	CO_2	LI6262, LI7000	ATI Sx	Suyker et al. (1997)
22	Happy Valley	69°09′N, 148°51′W	GS	Tundra	CO_2	LI6262	ATI SWS-211	Vourlitis and Oechel (1999)
23	U-Pad	70°17′N, 148°53′W	GS	Tundra	CO_2	LI6262	ATI SWS-211	Vourlitis and Oechel (1997)
24	24-Mile	69°56′N, 148°48′W	GS	Tundra	CO_2	NOAA/ATDD open-path	ATI SWS-211	Vourlitis and Oechel (1997)

Russia								
25	Zotino	60°45′N, 89°23′E	GS	Bog	CO_2	LI6262	Gill R3	Arneth et al. (2002)
26	Fyodorowskoye	57°27′N, 32°55′E	GS	Bog	CO_2	LI6262	Gill R3	Arneth et al. (2002)
27	NE Siberia	68°37′N, 161°20′E	GS	Tundra	CO_2	LI6262	Gill R3	Corradi et al. (2005)
28	Lena River Delta	72°22′N, 126°30′E	GS	Tundra	CO_2, CH_4	LI7000, Campbell TGA100	Gill R3	Kutzbach et al. (2007), Sachs et al. (2008)
29	Tiksi	72°22′N, 126°30′E	A	Tundra	CO_2, CH_4	LI7000, LGR RMT-200	METEK USA-1	Laurila et al. (2010)
30	Kytlyk	70°50′N, 147°30′E	GS	Tundra	CO_2	LI7500	Gill R3	van der Molen et al. (2007)
Tropical wetlands								
31	Fogg Dam	12°54′S, 131°31′E	A	Floodplain	CO_2	LI7500	Campbell CSAT3	Jason Beringer, personal communication (2011)
32	Dongtan	31°31′N, 121°58′E	A	Estuarine	CO_2	LI7500	Campbell CSAT3	Yan et al. (2008)
33	Indonesia	2°20′S, 114°2′E	A	Swamp	CO_2	LI7500	Campbell CSAT3	Hirano et al. (2007)
34	Jinja	0°24′N, 33°11′E	A	Papyrus	CO_2	LI7500	Campbell CSAT3	Saunders et al. (2007)
35	Everglades	25°22′N, 81°04′W	A	Mangrove	CO_2	LI7500	Gill RS50	Barr et al. (2010)

The ID number of the site refers to the symbols on the wetland map (Fig. 14.1). The annual coverage of measurements is presented as two categories: (A) annual cycle and (GS) growing season.

[a] At Barrow there have been several eddy covariance systems for CO_2 fluxes on different tundra ecosystems

of these sites are located on boreal and arctic peatlands, which are the focus of this chapter, but examples are also shown of the measurements in tropical wetland ecosystems.

14.3 Ecosystem-Specific Considerations

Many of the special features identified in connection with EC measurements over grasslands (Chap. 15) apply to open mires as well. The microtopography of mires is typically not as even as that of grasslands, but a relatively low (3–5 m) measurement height is normally sufficient, especially as the typical mire vegetation is rather short, consisting of mosses, shrubs, grasses, and sedges. On the one hand, this reduces the importance of the storage flux and the inherent uncertainties related to its determination, but on the other hand increases the importance of the corrections required for the imperfect frequency response of the measurement system (Sects. 1.5.4 and 4.1.3). Since mires are usually located in a flat landscape, advection problems generated by a sloping terrain are diminished. However, in small mires the turbulent flow field may be influenced by land cover types surrounding the area of interest; this disturbance should be evaluated with footprint models, as described in Chap. 8. The low measurement height also significantly simplifies the design of the measurement tower/mast (Sect. 2.2). In the case of forested wetlands, such as the treed fens and tropical swamps that fall outside the scope of this chapter, many of the EC specifics are common with those of forests in general (Chap. 11).

While the design and operation of measurement sites are discussed in great detail in Chap. 2, there are certain questions specific to wetlands that require further attention. A large proportion of wetlands are located in remote areas, which introduces additional requirements for logistics and site infrastructure. Mires often form extensive complexes, within which the most attractive sites, from the micrometeorological point of view, are difficult to access by car. It is advantageous, if a strip of land having mineral soil extends close to the measurement site. The installation of the measurement system and access to the site are complicated by the fact that peat soil is not firm and occasionally may become inundated. To ensure the stability of the measurement mast, it should be erected on a steady foundation, which can be constructed as a platform with supporting poles extending deep into the peat or, if possible, down to the mineral soil. Similarly, boardwalks are typically necessary for accessing the site and minimizing perturbations to the ecosystem during the maintenance. As mire vegetation is fragile, walking should be restricted to the boardwalks only. This also prevents disturbances by ebullition. An example of a measurement site established on wetland is shown in Fig. 14.2.

One of the main practical issues is the availability of mains power. Mains power is the ideal solution when CO_2 fluxes are measured with a closed-path analyzer in cold conditions, in which case the heating of the inlet tubes and the instrument cabin are necessary for most of the year. Mains power is usually also needed for the pump of a closed-path CO_2 analyzer, and practically always for the present closed-path CH_4

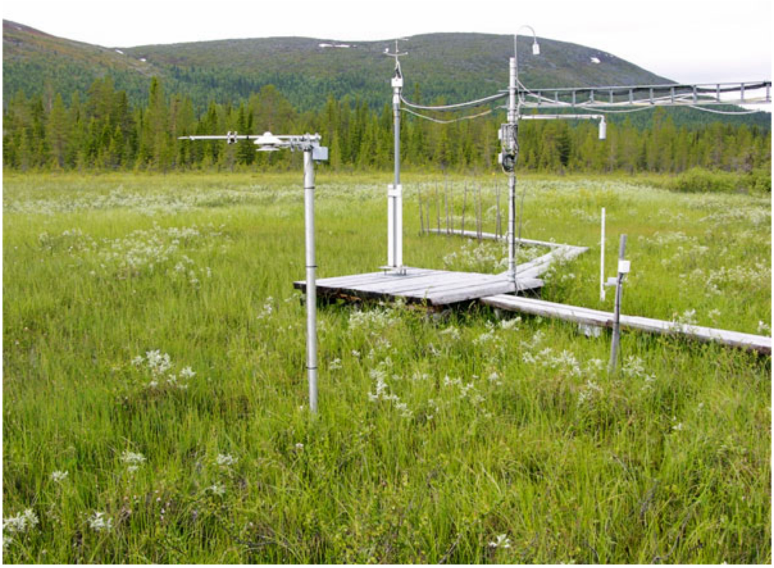

Fig. 14.2 Flux measurement site at the Lompolojänkkä fen located in northern Finland (67°59.832′N, 24°12.551′E). The micrometeorological mast is erected on a supported platform. Radiation sensors are mounted on a separate mast. The horizontal support carries a heated inlet tube from the sonic anemometer to the analyzer

analyzers, which require more powerful pumps. Nowadays, open-path instruments with lower power requirements are also available for both CO_2 and CH_4. However, solar radiation may not be sufficient throughout the year, which compromises the seasonal coverage of the measurements. In addition, in practice it is necessary to use a sonic anemometer that is equipped with sensor-head heating, in order to obtain year-round gap-free data in cold conditions. A generator-powered system requires constant maintenance, which is often difficult to accomplish at remote sites. If a generator is used, the effect of its exhaust gases on the trace gas measurements must be minimized.

Some of the specific solutions suggested above have implications for the quality assurance of the measurements. The necessary constructions at the site, especially the instrument shelter or cabin, potentially disturb the flux and other measurements, which should be taken into account when situating the sensors (Fig. 14.2). Wind-direction-based screening of the flux data may be necessary to minimize such disturbances. A similar screening procedure may also be needed for fetch considerations, if the measurement mast is located close to the edge of the mire, for example, to optimize the flux footprint in certain flow directions, or for the logistic reasons mentioned above (e.g., Aurela et al. 1998). The same applies to the avoidance of generator exhaust.

The northern location of many mires (Fig. 14.1) implies specific winter-related measurement questions, including the detection of small GHG fluxes, which are discussed below in a separate section. However, as a summer-specific final note on practicalities, it should be emphasized that additional filtering of the inlet flow is crucial in order to avoid a rapid blocking of the standard inlet filters due to an abundance of small insects, such as black flies, in wetland areas.

14.4 Complementary Measurements

Net radiation, global and reflected solar radiation, and global and reflected photosynthetically active photon flux density (PPFD) are the basic radiation components that should be measured as ancillary data for the EC fluxes. At boreal and arctic sites, the albedo decreases rapidly as the snow melts and the dark mire surface is exposed. With the appearance of new plants, the albedo gradually increases, but the reflected PPFD remains low. The ratios between the outgoing and incoming shortwave radiation, as well as the corresponding PPFD terms, can be used to trace the emergence and senescence of vegetation defining the growing season (Huemmrich et al. 1999).

In biological terms, boreal and arctic mires are characterized by a variation of habitats having differing plant communities, often corresponding to hydrological conditions that vary with microtopography and result in a mosaic of wet hollows, drier hummocks, and intermediate zones. Ancillary measurements should represent these different surfaces. Thus, the environmental measurements complementing the EC data should include more than one soil temperature profile. In northern regions, soil temperatures remain close to freezing point for extended periods. The soil temperature measurement system, including the data logger as well as the sensors, should be able to resolve variations of 0.1 K, and physically should tolerate inundated and icy conditions.

Continuous measurement of the water table depth (WTD) by a submerged pressure sensor is usually needed at one point at least. The reference level for WTD is usually at the peat surface, but it should be noted that the level of this surface may itself vary with a fluctuating WTD. Thus, it is useful if an additional WTD sensor can be anchored to the mineral soil to measure the absolute WTD changes. Above the WTD, the peat moisture content can be measured by time-domain reflectometer (TDR) probes. The TDR probes also serve as sensors for detecting the freezing of the soil, because dielectricity drops strongly when soil freezes, while it is difficult to distinguish the phase transitions from temperature measurements alone. Snow-depth measurements should also be made, employing continuously registering instruments.

Soil heat flux is normally measured using heat flux plates, which are basically designed for mineral soils. A poor or varying contact between the plate and the surface, as well as variations in the soil moisture, can cause erroneous signals. In mires, the WTD varies, and the heat flux sensor may be above or below it, but in either case the conditions are difficult for a heat flux plate. Below the WTD, the heat transport due to water flow may totally contaminate the measurement, while in the other case problems are caused by the large variations in the peat moisture as well as the limited contact between porous peat and the plate. To some degree, these problems may be overcome using so-called self-calibrating heat flux plates (Ochsner et al. 2006). However, a more reliable way of estimating the soil heat flux would be to employ high-resolution temperature profiles and to calculate the heat flux from temperature variations.

In addition to the physical parameters discussed above, measurements of biochemical variables would provide useful information for both the characterization of the soil and for understanding the soil processes. For example, ombrotrophic bogs are characterized by acidic soil water with low pH (3.0–4.5), while minerotrophic fens usually have a higher pH (4.5–8.0); oxic acrotelm and anoxic catotelm show opposite redox potential characteristics; O_2 concentration and redox potential vary seasonally responding to possible ice and snow cover in winter and flooding in spring; heavy rain events cause variation in acidity and nutrient status. For detecting variations of this kind, sensor packages are currently available for continuous measurements of O_2, redox, soil water pH, temperature, and pressure; these sensors can be permanently installed in the peat. These instruments provide exciting information on the temporal variation of biogeochemical processes, complementing the data on ecosystem–atmosphere exchange.

The atmospheric fluxes of CO_2 and CH_4 vary markedly between the microrelief elements described above. CO_2 exchange is most intense at the hummocks, while CH_4 emissions are highest from the wet surfaces. For this reason, it is laborious to obtain spatially representative flux data on the scale of a mire ecosystem by employing the chamber measurement technique. In contrast, the EC method provides a weighted measurement of the surface exchange within the observation footprint or the "field of view" of the flux sensor (Chap. 8), thus averaging over the small-scale heterogeneity related to habitat variability. However, for understanding the functioning of the ecosystem, it may be necessary to determine the response of a plant community to environmental variables, such as temperature, WTD, and VPD, separately for each microrelief. While a footprint analysis provides additional information for the interpretation of the EC measurements (e.g., Aurela et al. 2009; Laine et al. 2006), this small-scale variation cannot be fully extracted from the spatially averaged EC flux data, but entails the use of additional chamber measurements and ancillary meteorological measurements representing the individual surface types.

Measurements based on the chamber technique have previously been considered as an alternative to the micrometeorological flux measurements. However, the complementary nature of the EC and chamber measurements over low-vegetation mires is evident from those studies in which both these techniques have been employed. For example, the measurements on a homogeneous blanket bog (Laine et al. 2006), an even boreal fen (Riutta et al. 2007) and a subarctic fen with a pronounced microtopography (Maanavilja et al. 2011) all showed similar results. All these sites, while being highly variable in terms of plant communities, can be considered rather homogeneous at the ecosystem scale resolved with the EC method. However, the chamber measurements indicated that the CO_2 exchange varied markedly (twofold to sixfold) between the different plant communities of these mires, and that the variation was similar for respiration and photosynthesis.

In addition to revealing the differences between the microsites, the chamber data facilitate the partitioning of the measured net ecosystem exchange (NEE) into its photosynthesis and respiration terms. For the EC measurements, this partitioning can be calculated using different approaches described in Sect. 9.3. However, chambers enable a more direct partitioning based on the simultaneously measured photosynthesis and respiration fluxes, although introducing a potential representativeness problem with respect to EC data. Thus, it is recommended to integrate the two partitioning approaches.

The studies discussed above illustrate the role of chamber measurements as a complementary source of information supporting the EC flux measurements. However, it must be kept in mind that the gas exchange of porous peat measured in an enclosure may differ greatly from that taking place under the influence of turbulent atmospheric flow, which calls for nonintrusive flux measurement methods such as EC (Sachs et al. 2008). It should also be noted that a vegetation inventory and continuous leaf area measurements are essential, even if no chamber measurements are performed. A special feature of mires is the abundance of mosses, which in such an inventory should be characterized by their coverage and green biomass.

14.5 EC Measurements in the Wintertime

Mires are predominantly to be found in the northern high latitudes (Fig. 14.1). In these boreal and arctic areas, the winter season is relatively long, and snow covers the ecosystems for a significant part of the year. During the wintertime, GHG fluxes are typically small as compared to the growing season. The soil temperature under the insulating snow cover remains close to zero irrespective of the possible harshness of the air temperature. In several studies it has been found that the microbial activity in the soil continues even in subfreezing temperatures (Flanagan and Bunnell 1980; Coxon and Parkinson 1987; Zimov et al. 1993), keeping the formation of CO_2 and CH_4 going throughout the winter. Several long-term studies have shown that these wintertime emissions contribute significantly to the annual CO_2 balances. For example, in a southern boreal bog in Canada, the wintertime CO_2 flux was estimated to represent 25–35% of the annual CO_2 balance (Lafleur et al. 2003), while in a subarctic fen in northern Finland the wintertime efflux is actually greater than the annual uptake (Aurela et al. 2002). For CH_4, the wintertime fluxes are not as important for the annual sum. At a southern boreal fen in Finland, the wintertime CH_4 emission was about 5% of the annual total efflux (Rinne et al. 2007). An example of the seasonal variation of CO_2 and CH_4 fluxes observed at a subarctic fen is shown in Fig. 14.3.

The wintertime decrease in the GHG formation, and consequently in the magnitude of the fluxes, results in an increased noise in the data measured with the EC technique. This is typically manifested by negative values in the flux time

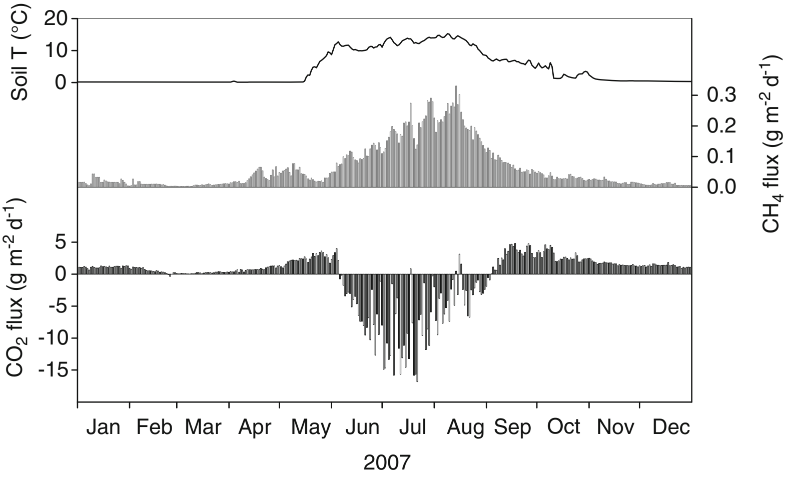

Fig. 14.3 Mean daily exchange of CO_2 (*black bars*) and CH_4 (*gray bars*) together with the soil temperature (at −10 cm) (*line*) at the Lompolojänkkä fen in northern Finland (67°59.832′N, 24°12.551′E)

series, corresponding to apparent uptake in conditions in which no uptake can be expected. As this variation can be considered to be random noise, averaging these data over longer periods produces unbiased estimates. Therefore, it is important to pay attention to the data screening protocols, in order to avoid systematic errors due to data selection. For example, discarding all the negative values as nonphysical would definitely bias the estimated averages and, consequently, the long-term GHG balances.

The decrease of the signal-to-noise ratio also has an indirect influence, especially on the measurement of GHG fluxes with a closed-path instrument, for which the lag time related to the air travel in the tubes must be taken into account in the flux calculations (Sect. 3.2.3.2). When the fluxes are sufficiently high and the turbulent flow is well-defined, the lag-covariance relationship shows a clear (absolute) maximum. As the fluxes decrease, however, this relationship becomes noisier, with many local maxima within the lag search window. If this variation is even partly due to random noise, a systematic error is induced into the long-term mean flux, if the highest value is invariably selected. Thus, the lag algorithm should be chosen with care when the fluxes to be measured are small, such as the GHG emissions from northern mires in winter. In this case, it is recommended to define a rather narrow search window, and apply a predefined constant lag value, if a window limit is exceeded.

In addition to diminished fluxes, the snow cover also brings about other effects that complicate the interpretation of the data measured in winter. As the snowpack acts as a buffer, the observed GHG fluxes are partly decoupled from the actual formation of these gases related to the microbial activity taking place in the different peat layers. This has an influence on the parameterization of the relationships between fluxes and their drivers, needed for the partitioning of the fluxes into the productivity and respiration components, for example (Sect. 9.3). During the summertime, the observed fluxes can be related to concurrent soil or air temperatures, while during the snow-cover season air temperature variations have a lesser influence on the decomposition processes, and thus a weaker correlation with the observed fluxes. However, as the thermal conditions beneath the snow are rather constant, it is still possible to capture the longer-term variations with a simple temperature-response model by using appropriate soil temperatures (Aurela et al. 2002).

Another phenomenon complicating the winter fluxes is their dependence on the atmospheric flow, appearing in practice as a correlation with wind speed or friction velocity. Events where the measured CO_2 efflux increases with increasing wind speed are often observed over snow-covered ecosystems (Goulden et al. 1996; Aurela et al. 2002). This is usually caused by ventilation of the snowpack, in which GHG concentrations tend to accumulate under conditions of slow diffusion into the atmosphere (i.e., with low wind speeds) (Massman et al. 1997). In principle, it would be possible to include this phenomenon in the CO_2 flux parameterizations, but the dependence is often nonlinear, even in the short term. While ventilation increases the fluxes, it decreases the storage, and thus the effect gets weaker in time. Usually, this flow-dependency is not taken into account in gap-filling models, as its effect becomes negligible over longer averaging periods.

14.6 Carbon Balances and Climate Effects

In terms of their GHG balances, wetlands differ from many other ecosystems in that CH_4 emissions play a central role, while N_2O fluxes are typically insignificant. As CH_4 analyzers have become more affordable and easier to deploy in the field, an increasing number of flux stations have started measuring CH_4 alongside with CO_2 fluxes. Figure 14.3 illustrates a typical annual cycle of CO_2 and CH_4 exchanges observed at a northern fen, with the characteristic seasonal dynamics that collectively define the annual balances. As discussed above, in winter the fluxes are small but clearly non-negligible. The increased soil temperature after the snowmelt enhances both the CO_2 and CH_4 effluxes. During the melting process, CH_4 fluxes often show an additional enhancement that cannot be directly related to soil temperature variations or CH_4 formation processes. This so-called spring pulse is caused by the release of a CH_4 reservoir that has accumulated beneath the frozen peat layer during winter (Hargreaves et al. 2001). The growing-season fluxes of CO_2 are co-controlled by the simultaneous photosynthetic uptake and respiration processes. The seasonal cycle of CH_4 appears simpler, being controlled mainly by soil temperature and vegetation phenology. It is thus much easier to fit a simple site-specific temperature-response function to the CH_4 fluxes; this function can then be used for gap-filling the measurement time series. CH_4 fluxes can even be parameterized as daily averages, something that is not possible for CO_2 fluxes.

In addition to the net exchanges of CO_2 and CH_4 that can be measured with the EC technique, there are also other components in the net carbon balance of a mire ecosystem: the export and import by lateral water flow of total organic carbon (TOC), dissolved inorganic carbon (DIC) and CH_4, as well as the import of carbon through precipitation. In practice, not all of these components are significant, but some of them may play an important role in the total balance. For example, at a boreal oligotrophic minerogenic mire in Sweden, the largest term of the balance was carbon gain by net CO_2 uptake (48 g C m^{-2} in 2005), followed by CH_4 emission (14 g C m^{-2}) and the TOC export by a stream (12 g C m^{-2}) (Nilsson et al. 2008). On the other hand, the stream export of DIC and CH_4 (3.1 and 0.1 g C m^{-2}, respectively) and TOC deposition (1.4 g C m^{-2}) were of lesser importance. These components add up to a total annual carbon accumulation of 20 g C m^{-2}, which is 42% of the NEE of CO_2.

It is interesting to compare an estimate of the current carbon balance with the carbon accumulation in the peat profile over a longer time horizon, which is possible if either a certain peat layer (using, for example, the initiation of tree growth; Schulze et al. 2002) or the peat bottom is dated. The latter provides the long-term apparent carbon accumulation (LARCA) over the life span of the mire during the Holocene. The current carbon accumulation rate can be estimated from the LARCA based on an accumulation model (e.g., Clymo et al. 1998), which for the Swedish mire discussed above resulted in an accumulation rate that is very close to the current balance derived from the EC and other measurements (Nilsson et al. 2008).

The climate effects of GHG fluxes are profoundly altered, if the natural ecosystems are subjected to an intervention by human management (e.g., Lohila et al. 2010). For example, the draining of a natural mire for agriculture or forestry typically stops CH_4 emissions, while for CO_2 the peat soil may turn from a sink into a source. While a knowledge of both the CO_2 and CH_4 fluxes is essential for the carbon balance and climate effects to be estimated, the measurement of the third major GHG, that is, N_2O, is usually not needed, because cool, inundated conditions do not favor its production (Regina et al. 1996). However, N_2O emissions are usually high, if the peat soil has been prepared for agriculture (Augustin et al. 1998). This effect may last a very long period. For example, a bog that had been afforested three decades earlier (Lohila et al. 2007), after being first drained for agriculture, still emitted large amounts of N_2O (Mäkiranta et al. 2007). As practically all the histosol sites in Europe, excluding the outskirts of the continent, have experienced human influence during the past centuries, the management history of a potential flux measurement site should be explored to identify measurement needs, and to facilitate the subsequent interpretation of the results obtained. The importance of the non-CO_2 GHGs is highlighted by the fact that, compared to CO_2, they have much higher radiative forcing efficiencies per unit emission, and also different atmospheric lifetimes, complicating the analysis of climate effects of these ecosystems.

14.7 Concluding Remarks

This chapter has concentrated on natural mires in boreal and arctic environments. However, it is important to briefly note the significance of wetlands located in lower latitudes, where high fluxes with sometimes unexpected variations may be observed, as well as the part played by peatlands that have experienced management or are influenced by climate change effects. For example, 3 years of EC data from a drained tropical swamp ecosystem showed very high respiration (3,870 g C m^{-2} $year^{-1}$), resulting in a significant net loss of CO_2 to the atmosphere (430 g C m^{-2} $year^{-1}$) (Hirano et al. 2007). A mangrove forest in Florida was a large sink of CO_2 (1,170 g C m^{-2} $year^{-1}$) due to the low respiration in comparison with the high photosynthetic uptake (2,340 g C m^{-2} $year^{-1}$) (Barr et al. 2009). At these sites, the potential technical problems, such as those related to the impact of high humidity and temperature, thunderstorms and hurricanes, certainly differ from those discussed in this chapter. Thus, these measurements should be considered outstanding accomplishments, further demonstrating the applicability of the EC technique.

Another environment worth paying more attention to is the permafrost region that is thawing due to the warming climate. In the degrading permafrost area, the landscape consists of flarks, lawns, and strings with different vegetation types and non-vegetated peat molds and ponds. This comprises an extremely heterogeneous surface with hotspots of fluxes with opposite directions. This variability is true not only for CH_4 and CO_2, but also for N_2O, for which emissions have been observed from a bare peat surface in a subarctic wetland (Repo et al. 2009). As another example, the area of wet flarks has been observed to increase in a thawing wetland in alpine northern Sweden, enhancing the CH_4 emissions (Christensen et al. 2004). In these environments, EC measurements provide an invaluable instrument for tracing the impacts of climate change on GHG exchange on a long-term basis.

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