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Invited review

Greenhouse gas emissions from soils—A review

Cornelius Oertel [a], Jörg Matschullat [a,*], Kamal Zurba [a], Frank Zimmermann [a], Stefan Erasmi [b]

[a] Institute of Mineralogy, Chair for Geochemistry and Geoecology, Technische Universität Bergakademie Freiberg, Brennhausgasse 14, 09599 Freiberg, Germany
[b] Geographisches Institut, Abt. Kartographie, GIS & Fernerkundung, Universität Göttingen, Goldschmidtstr. 5, 37077 Göttingen, Germany

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ABSTRACT

Soils act as sources and sinks for greenhouse gases (GHG) such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Since both storage and emission capacities may be large, precise quantifications are needed to obtain reliable global budgets that are necessary for land-use management (agriculture, forestry), global change and for climate research. This paper discusses exclusively the soil emission-related processes and their influencing parameters. It reviews soil emission studies involving the most important land-cover types and climate zones and introduces important measuring systems for soil emissions. It addresses current shortcomings and the obvious bias towards northern hemispheric data.

When using a conservative average of 300 mg CO_2e m^{-2} h^{-1} (based on our literature review), this leads to global annual net soil emissions of ≥350 Pg CO_2e (CO_2e = CO_2 equivalents = total effect of all GHG normalized to CO_2). This corresponds to roughly 21% of the global soil C and N pools. For comparison, 33.4 Pg CO_2 are being emitted annually by fossil fuel combustion and the cement industry.

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* Corresponding author.
E-mail address: matschul@tu-freiberg.de (J. Matschullat).

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1. Main greenhouse gas source and sink aspects of soils

Political agendas of individual countries and international initiatives proclaim greenhouse gas (GHG) neutrality, e.g., by the year 2050 (G7, 2015; Law and Harmon, 2011; UBA, 2013; Willson and Brown, 2008). Whether such declarations can be and must be seen as realistic cannot be an issue in this review. Yet, it is of vital interest to properly assess soils as GHG-sources and to develop a better understanding for the source and sink strengths of this important and critical environmental compartment. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are important climate-relevant trace gases. While being highly unequally distributed (Δ of 3–4 orders of magnitude for C), about 1500 Pg of total C and 136 (92–140) Pg of total N are stored in the uppermost meter of the global soil layer, representing the largest terrestrial carbon and nitrogen pools (Batjes, 1996; Kutsch et al., 2009; Nieder and Benbi, 2008; Schaufler et al., 2010; Schlesinger and Andrews, 2000). However, soil structure changes can influence its source and sink function (Jungkunst and Fiedler, 2007) and the total storage capacity is limited. As an example, a review of the European terrestrial GHG balance (Schulze et al., 2009) showed that the fluxes (flow rate per unit area) of CO_2, CH_4 and N_2O between 2000 and 2005 were near neutral; agricultural CH_4 and N_2O emissions being largely offset by the CO_2 sink of grasslands and forests. Yet, the authors show that the trend towards more intensive agriculture and logging "*is likely to make Europe's land surface a significant source of greenhouse gases*". Bahn et al. (2010) compiled and evaluated global data across vegetation types and biomes, concluding that soils emit about 98 Pg C a^{-1} (resulting in a roughly 15-year turnover time for the reservoir). This is considerably more than emissions from fossil fuel consumption. Older sources suggest lower fluxes with 68–77 Pg C a^{-1} (Raich and Potter, 1995; Raich and Schlesinger, 1992). Bahn et al. (2010) nevertheless emphasized the still high uncertainty with various assumptions and in upscaling attempts.

GHG emissions from soils need to be better quantified for global budgets, since 35% CO_2, 47% CH_4, 53% N_2O, and 21 % nitric oxide (NO) of the respective total annual emissions relate to soil degassing (IPCC, 2007). The annual global NO emissions from soils are within the range of NO emissions from fossil fuel combustion (Butterbach-Bahl et al., 2009). The increase of GHG emissions from soils mainly stems from CH_4 and N_2O since the onset of industrialization in the mid 18th Century and has been caused by agricultural practice (Forster et al., 2007). This makes GHG emissions from soils a key topic in global change issues, in climate research, and for agricultural and forestry management.

First measurements of CO_2 emissions from soils were performed in laboratories in the 19th Century (Boussingault and Levy, 1853), motivated by the tracer function of soil respiration for soil fertility (Russell and Appleyard, 1915). Today climate change-related topics dominate the study of CO_2, N_2O and CH_4 behaviour (Cox et al., 2000). Nitric oxide (NO) has been studied because of its role in acidic precipitation, ozone formation and destruction (Kampfl et al., 2007). Field measurements with chambers were introduced at the beginning of the 20th Century (Lundegardh, 1927). Determinations of other trace gas emissions started later, as gas analyzers became available (N_2O in the 1950s: Arnold, 1954; NO in the 1970s: Galbally et al., 1987; CH_4 in the 1980s: Holzapfel-Pschorn et al., 1985).

1.1. Main gas species: properties and processes

CO_2 fluxes can be separated into three types:

1) Soil respiration includes root, anaerobic and aerobic microbial respiration. Root respiration contributes on average up to ~50% of the total soil respiration, yet may vary between 10 and 95% subject to season and vegetation type (Hanson et al., 2000),
2) Ecosystem respiration additionally includes aboveground plant respiration,
3) Net ecosystem exchange (NEE) is the difference between photosynthesis and ecosystem respiration. A positive NEE indicates a CO_2 source, whereas a negative NEE reveals a CO_2 sink.

CH_4 in soils is produced by methanogenesis under anaerobic conditions and is consumed by methanotrophic microorganisms that use O_2 and CH_4 for their metabolism under aerobic conditions (Dutaur and Verchot, 2007).

N_2O and NO releases are driven by nitrification (oxidation of NH_4^+ to NO_3^- via NO_2^-) and denitrification (reduction of NO_3^- to N_2O and N_2). Nitrous oxide (N_2O) is produced mainly by denitrification under anaerobic conditions, where the Water-Filled Pore Space (WFPS) is >50% (Ussiri and Lal, 2013). The released NO

amount is negligible, compared to the NO production of nitrification (Brümmer et al., 2008). N_2O is produced by nitrification under aerobic conditions (Ussiri and Lal, 2013). Still, quite a few open questions remain, particularly with respect to processes and their controls (Butterbach-Bahl et al., 2013) that cannot be comprehensively addressed in this work.

1.2. Review context and objectives

Analyzing the influence of the different parameters on soil emissions remains challenging due to the multitude of interacting processes. Laboratory studies help in solving this problem (Schaufler et al., 2010)—see Section 3.3. Montzka et al. (2011) reviewed the non-CO_2 GHGs from a global perspective, showing the fluxes, the relevance of atmospheric residence time and flux rate alterations, and their potential in climate change mitigation.

This paper discusses soil emission-related processes and their influencing parameters. It reviews soil emission studies from the most important land-cover types and climate zones and introduces important methods for measuring soil emissions. Thus this work can be seen as complementary to important preceding reviews, e.g., Butterbach-Bahl et al. (2013), Dalal and Allen (2008), Montzka et al. (2011), Raich and Tufekcioglu (2000), Schaufler et al. (2010), and Snyder et al. (2009).

We focus on CO_2, CH_4 and N_2O because available high-quality data for these gases cover a wide spectrum of land-use types, crops and climates. Standard deviations are given in the text, when these were mentioned in the original publications. Emission rates have been recalculated throughout to obtain uniform units ($\mu mol \times m^{-2} s^{-1}$ [CO_2] and h^{-1} [CH_4 and N_2O]). To enable a direct comparison between the individual GHGs, the individual gas flux rates have been re-calculated to CO_2 equivalents (CO_2e) and normalized to 100 years, considering their respective atmospheric residence times (IPCC, 2013).

1.3. Drivers of soil emissions

Microbial activity, root respiration, chemical decay processes, as well as heterotrophic respiration of soil fauna and fungi produce GHGs in soils (Chapuis-Lardy et al., 2007). Related emission flux rates largely depend on soil water content (humidity), soil temperature, nutrient availability and pH-value (Ludwig et al., 2001), plus land-cover-related parameters. Thus, meteorological and climatological parameters as well as land-use management information are paramount (Fig. 1). Drivers can be separated into proximal drivers that influence soil emissions in the direct environment (e.g. local climate, soil type) and distal drivers that effect soil emissions on larger scales (e.g. temperature, humidity); Robertson (1989).

1.3.1. Humidity

Soil humidity is the single most important soil parameter for soil gas emissions, since it controls microbial activity and all related processes (Fig. 1). Nitrifying bacteria require oxygen residing in soil pores. Soils with less water-filled pore space (WFPS) show higher emissions by nitrification, with a maximum at 20% WFPS (Ludwig et al., 2001). Nitric oxide (NO) emissions decrease in soils having below 10% WFPS due to inhibited nutrient supply (Brümmer et al., 2008). Nitrification yields a higher potential for NO production than for N_2O production (Fowler et al., 2009). In contrast, CH_4- and N_2O-producing bacteria require anaerobic conditions. N_2O production is optimal around 60% WFPS and lowest when WFPS is below 30% (Gao et al., 2014). Even an increase of WFPS above 80% still leads to an exponential increase of N_2O emissions (Keller and Reiners, 1994). CH_4 production requires strictly anaerobic conditions and correlates positively with soil humidity (Gao et al., 2014; Smith et al., 2003). Soils are CH_4 sinks under aerobic conditions (Fiedler et al., 2005). Wetlands and rice paddies are strong CH_4 sources (Dutaur and Verchot, 2007). Long periods of drought can significantly reduce soil emissions. Soils may then turn into a net sink for N_2O (Goldberg and Gebauer, 2009). Grain-size distribution influences soil moisture. Soils with a high proportion of large pores retain less water and therefore foster the emission of gases produced under aerobic conditions (van der Weerden et al., 2010). NO emissions are highest in soils with coarse soil textures (McTaggart et al., 2002). Soils with dominant fine pores support the formation of CH_4 and N_2O produced under anaerobic conditions (Dutaur and Verchot, 2007; Gu et al., 2013). Higher CO_2 emissions were encountered with fine textured soils, especially compared to sandy soils during warm dry periods (Dilustro et al., 2005). Stable soil aggregates (concretions, crusts) lead to lower soil emissions since C and N are less available for soil microbes (Kögel-Knabner et al., 1998).

Precipitation after extended dry periods causes the pulsing or Birch effect (Birch, 1958). Emissions increase within some minutes or hours after the onset of precipitation and return to background levels within a few days (Sponseller, 2007). This is driven by the renewed mineralization and the availability of easily decomposable material (Borken and Matzner, 2009) for the metabolism of reactivated microbes (Ludwig et al., 2001). The Birch effect decreases with higher frequencies of wet–dry cycles (Borken and Matzner, 2009).

1.3.2. Temperature

Soil temperature is important to explain the variations of trace gas emissions from soils (Fig. 1). Soil moisture and soil temperature can explain 74% and 86% of the variations of NO and N_2O emissions, respectively (Schindlbacher et al., 2004). An increase of soil temperature leads to higher emissions and to higher soil respiration rates as a positive feedback response of increased microbial metabolism. Methane and N_2O emissions are additionally forced by increasing soil respiration rates with increasing soil temperatures, leading to decreasing O_2 concentrations in the soil (Butterbach-Bahl et al., 2013). The positive temperature effect may be overlain by soil water stress, since water is needed as a transport medium for nutrients required by microbes (Fowler et al., 2009). Nitric oxide and CO_2 emissions increase exponentially with temperature (Fang and Moncrieff, 2001; Ludwig et al., 2001; Tang et al., 2003). Soil respiration from bacteria was observed down to soil temperatures of −7 °C (Brooks et al., 1997).

The temperature dependency of gas emissions from soils can be described with the temperature sensitivity factor Q_{10}. It expresses the rate of change in a chemical or biological system with a temperature change of 10 °C (Berglund et al., 2010) and increases with soil depth (Tang et al., 2003). Q_{10} is 2.4 with a range of 1.3–3.3 for soil respiration; based on a data review by Raich and Schlesinger (1992). The average values are confirmed by current studies of Hu et al. (2015) with a range of 1.7–2.5 and Jiang et al. (2015) with an average of 2.2. The response of CH_4 emission to temperature increases with a Q_{10}-value of ~4 (Dalal and Allen, 2008). N_2O emissions increase with temperature up to about 37 °C; thereafter denitrification and N_2O emissions decrease. Reported Q_{10}-values for N_2O range from 1.7 to 9.3 (compiled by Abdalla et al., 2009a).

Under field conditions, moisture and temperature effects overlap, which may make it difficult to observe clear correlations (Fang and Moncrieff, 2001). Temperature is important for the regulation of freeze–thaw events, forcing gas emissions from soils (Holst et al., 2008), and may be responsible for up to 50% of the total annual N_2O emissions (Groffman et al., 2009). In contrast, winter CO_2 emissions are considered less important for the annual emission budget since root respiration is low in temperate or more polar environments (Groffman et al., 2006). Therefore, soil water content has to be close to saturation to reduce the O_2 content (Groffman et al., 2009). During freeze–thaw cycles, additional nutrients are released

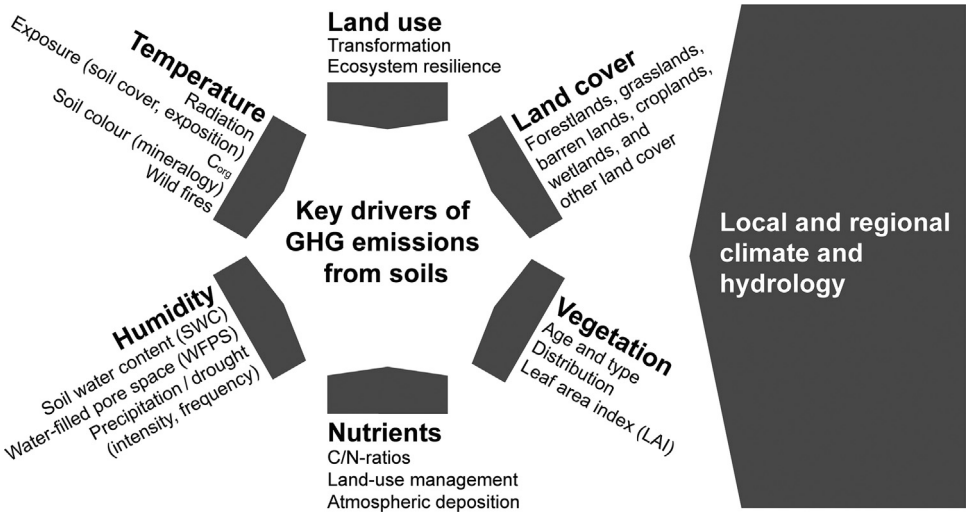

Fig. 1. Key drivers of GHG emissions from soils.

for microbial metabolism through the disaggregation of soil particles (Christensen and Christensen, 1991). After thawing, dead organic material (e.g., dead plant roots) forces increased microbial soil respiration and N_2O emissions (Mørkved et al., 2006). We argue that winter emissions are relevant for the temperate climate zone. Independently, maximum soil emissions result in spring.

1.3.3. Exposure and air pressure

Site exposure (elevation, morphological position, plant cover) influences soil temperature and moisture. Nitrous oxide emissions are higher in depressions than on slopes and ridges due to higher soil moisture (Izaurralde et al., 2004). Lower air pressure supports higher soil emissions due to reduced counter pressure on the soil (Reicosky et al., 2008).

1.3.4. Vegetation fires

Fires in ecosystems can affect the greenhouse gas balance of soils, depending on temperature and duration of the fire, with burned areas showing lower CO_2 and N_2O fluxes than non-burned reference sites for around one month after burning (Kim, 2013; Table 1). This is caused by reduced root respiration in the absence of plant cover and the related pH change. Lower N_2O emissions are caused by charcoal (Kim, 2013). After burning, soil temperatures increase due to missing canopy, while soil moisture does not change since lowered plant transpiration compensates for missing or reduced plant canopy (Castaldi and Fierro, 2005). No or minor changes were observed for CH_4 uptake in soils (Kim, 2013). Thermal ammonification causes ammonium increases after a forest fire. Nitrate decreases with time due to suppressed nitrification (MacKenzie et al., 2004).

1.3.5. Soil pH-values

Microbial activity is influenced by soil pH. Therefore, management practices such as liming influence soil emissions; additional carbonate can be released as CO_2 (Snyder et al., 2009). Acidic soil conditions lead to lower soil emissions. The optimal pH-value for methanogenesis (CH_4 production) lies between pH 4 and 7 (Dalal and Allen, 2008). CO_2 emissions were observed to be highest at neutral pH-values (Cuhel et al., 2010). N_2O emissions decrease only under acidic soil conditions. Nitrification increases with higher pH-values, since the equilibrium between NH_3 and NO_3^- shifts to ammonia (Nugroho et al., 2007). However, no significant correlations between NO and N_2O emissions and pH-value were found (Pilegaard et al., 2006). NO emissions are caused by denitrification under acidic soil conditions, whereas alkaline conditions foster NO emissions produced by nitrification (Remde and Conrad, 1991).

1.3.6. Nutrients

Nutrient availability is paramount to microbial and plant respiratory processes. Hence, natural N and C content in soil, as well as atmospheric deposition, manure or fertilizer applications play an important role. N_2O emissions negatively correlate with the C/N-ratio (Pilegaard et al., 2006), with N_2O emissions being lowest at C/N-ratios $\geq$30 (limited disintegration of organic material) and highest at a C/N-value of 11 (optimum disintegration and humus build-up); Gundersen et al. (2012a,b). Combined with drought and low pH-values, N_2O emissions can be significantly suppressed at C/N-ratios <20 (Gundersen et al., 2012a,b). Emissions of CO_2 and CH_4 positively correlate with the C/N-ratio (Shi et al., 2014; Weslien et al., 2009). Yet, depending on the availability of other electron donors such as Fe_3^+, Mn_4^+, SO_4^{2-} and NO_3^-, the CH_4 production in soils may be reduced (Achtnich et al., 1995; Dalal and Allen, 2008; Fumoto et al., 2008; Kögel-Knabner et al., 2010; Sahrawat, 2004), which is of particular relevance in many (sub)tropical soils and particularly in rice paddies.

Increasing soil N content generally leads to higher soil respiration and to higher net ecosystem exchange (NEE), if carbon is not limiting (Niu et al., 2010; Peng et al., 2011). With limited C availability, N fertilizer application has limited influence on soil respiration (Micks et al., 2004). The application of N fertilizers leads to a higher sensitivity of soil respiration against soil moisture and to a lower sensitivity to soil temperature (Peng et al., 2011). In short-term studies, application of NO_3^- or NH_4^+ leads to a decrease or to no change in the soil respiration rate of forest soils (Micks et al., 2004). Soil respiration decreases during long-term experiments of N addition (Bowden et al., 2004). The application of liquid manure (urea) led to higher N_2O emissions under aerobic soil conditions, whereas NH_4^+ fertilizers caused higher N_2O emissions under saturated conditions (Tenuta and Beauchamp, 2003).

To minimize N_2O emissions from agricultural lands, fertilizer application rates need to be adapted to plant needs since not all forms of nitrogen can be taken up by plants. Non plant-available N amounts lead to increasing N_2O emissions (McSwiney and Robertson, 2005). Cover crops increase CO_2 and N_2O fluxes after their incorporation and positively influence soil respiration during their lifetime (Sanz-Cobena et al., 2014b). Using controlled-release fertilizers or denitrification inhibitors prevents increased N_2O emissions (Shoji et al., 2001); nevertheless this effect can be

Table 1
Soil emission rates in μmol CO_2 m^{-2} s^{-1} and μmol X m^{-2} h^{-1} for CH_4 and N_2O from grassland under temperate, mediterranean and (sub)tropical climates. Rates are given without standard deviations. Ranges have been rounded for clarity (see original literature and text for more details).

	Temperate		Mediterranean		(Sub)Tropical	
Species	Rate	Comment	Rate	Comment	Rate	Comment
CO_2	1.54–1.61	Perennial grassland[15]	1.4–2.7	Summer–spring[5]	7.9	Urban[12]
	0.48–0.57	Tundra[15]	0.54–0.57	Annual avg.[17]	0.71–0.92	Unburnt Cerrado[14]
	1.26; 3.09	Non f.; fertilized sites; summer[13]	1.99	Grassland[1]	0.75–1.33	Burnt Cerrado[14]
	4.30	European sites[20]				
	2.96	Grassland[1]				
	7.98	Peatland[9]				
	3.18–5.30; 2.60–3.18	Reclaimed coal mining, fertilized; non-fertilized[6]				
	4.41	Reclaimed lignite mining[25]				
	1.82–7.95; 11.6–25	Seasonal avg; peak values: peat, sandy, clay soil[8]				
Range CO_2	0.5–8.0 (max. 25)		0.5–2.7		0.7–1.3 (max. 7.9)	
CH_4	0.04	European sites[20]	−0.9 to −0.5	Annual avg.[17]	−2.21; −0.93	Degraded pasture; active pasture[22]
	−2.36	Upland steppe[7]			3.0	Unburnt Cerrado[14]
	1.35–1.60; −3.54– −1.63	Reclaimed coal mining: fertilized; non-fertilized[6]			−35.1 to −11.4	Burnt Cerrado[14]
					−1.3	Cerrado, annual avg.[3]
Range CH_4	−3.5–0.04 (max. 1.6)		−0.9– −0.5		−35.1– −0.9 (max. 3.0)	
N_2O	9.74	European sites[20]	0.02–0.15	Annual avg.[17]	0.10; 0.02	Active; degraded pasture, annual avg.[23]
	0.06	Grassland steppe[21]			0.1–0.59	Undisturbed grassland[19]
	0.03	Steppe (spring), intense grazing[26]			0.21	Grassland[16]
	0.45	Steppe (spring), ungrazed[26]			0.17–0.41 ann. avg. (Peaks ~11)	Grassland[2]
	<18–180	Intensively managed grassland[11]			0.3	Various grassland, median[3]
	0.11	Typical managed grassland[10]			0.04–0.18	Burnt Cerrado[14]
	1.30–3.10	Boreal grassland[24]			0.06	Unburnt Cerrado[14]
	1.52	Peatland[9]				
	0.41–0.82; 1.63–2.44; 4.07–4.89	Ungrazed; sheep; dairy grazed[18]				
	0.68	Sheep grazing[4]				
	0.63; 0.85	Reclaimed coal mining: fertilized; non-fertilized[6]				
Range N_2O	0.03–9.7 (max. 180)		0.02–0.15		0.02–0.6 (max. 11)	

[1] Bahn et al. (2010) temperate: various European, Mediterranean: Catalonia.
[2] Brümmer et al. (2008) Burkina Faso.
[3] Castaldi et al. (2006).
[4] Choudhary et al. (2002) New Zealand.
[5] Correia et al. (2012) Portugal.
[6] Jacinthe and Lal (2006), USA.
[7] Liu et al. (2009a): inner Mongolia.
[8] Lohila et al. (2003) Finland.
[9] Maljanen et al. (2004) Finland.
[10] Merbold et al. (2013) Switzerland.
[11] Merbold et al. (2014) Switzerland.
[12] Ng et al. (2014) urban grassland Singapore.
[13] Peng et al. (2011) China.
[14] Poth et al. (1995) unburnt and (freshly) burnt Cerrado soils, Brazil.
[15] Raich and Potter (1995).
[16] Rees et al. (2006) Zimbabwe.
[17] Ryals and Silver (2013) California.
[18] Saggar et al. (2008) New Zealand.
[19] Sanhueza et al. (1990) Venezuela.
[20] Schaufler et al. (2010) avg. of sites in United Kingdom, Switzerland and Hungary.
[21] Stehfest and Bouwman (2006).
[22] Verchot et al. (1999a) Eastern Amazonia.
[23] Verchot et al. (1999b) Eastern Amazonia.
[24] Virkajärvi et al. (2010) Finland.
[25] Weiß (2011) Germany.
[26] Wolf et al. (2010) P.R. China with unclear observation time (here calculated with 8 weeks).

disturbed by heavy precipitation events (Venterea et al., 2012). Soil water content is important for selecting the fertilizer type to inhibit increased N_2O emissions (Sanz-Cobena et al., 2014a). Fertilizer applications are also influenced by the tillage system. Using urea, N_2O emissions were higher under no-till and conservation tillage, while no differences could be observed using urea-ammonium nitrate fertilizers (Venterea et al., 2005). However, there are discordant findings on the influence of the tillage system: N_2O emissions decreased with no-till practice (Omonode et al., 2011) and were explained by lower soil temperatures (Six et al., 2002), while others found a positive effect of no-till on N_2O emissions and explained this with higher microbial activity (Baggs et al., 2003). Higher soil moisture during no-till practice leads to higher N_2O emissions. Higher N_2O emissions under no-till cannot be balanced by higher C sequestration and CH_4-uptake rates (Li et al., 2005; Six et al., 2002). Yet, no information about the starting point of the no-till practice for this study was included. This is essential, since conventional and no-till approaches can only be compared once an equilibrium has been established after extended periods of time. Six et al. (2004) reviewed long-term studies on the influence of no-till practices on N_2O emissons. They found that N_2O emissions were only elevated during the first ten years after shifting from conventional tillage to conserving no-till farming practice. Soil compaction caused by the use of farming machines leads to lower soil emissions (Mordhorst et al., 2014).

A reduction of N deposition led to decreasing soil NO emissions at forest sites, whereas N_2O emissions were not affected by the reduction (Eickenscheidt and Brumme, 2012).

1.3.7. Vegetation

Vegetation age and tree type (deciduous, coniferous, and species) influence soil respiration. The highest soil respiration rates were found for young spruce forest stands, when comparing 10-, 15-, 31- and 47-year-old stands (Saiz et al., 2006). Soil respiration decreased with stand age, caused by a lower fine root biomass. The decrease levelled out with stand age since lower root respiration rates in old forest were partly compensated for by higher microbial respiration due to higher organic inputs (Saiz et al., 2006). Similar results were found for 45- and 250-year-old pine stands (Law et al., 1999), 20- and 40-year-old fir stands (Klopatek, 2002), and for 5- and 15-year-old poplar plantations (Gong et al., 2012). A high biodiversity with a balanced ratio of *Leguminosae* and C3 and C4 plants (carbon fixation) at grassland sites resulted in an increased C-sequestration potential (Fornara and Tilman, 2008). On agricultural sites, N_2O emissions from legume-N were significantly lower than fertilizer-N derived N_2O emissions (Schwenke et al., 2015).

Vegetation affects CH_4 emissions and positively correlates with net ecosystem production (Dalal and Allen, 2008). Elevated CO_2 concentrations in soils can also be caused by higher root mass due to elevated atmospheric CO_2 concentrations (Dorodnikov et al., 2009). Furthermore, soil moisture increases, since the opening time of stomata can be reduced. Hence, conditions for N_2O and CH_4 emissions improve (Kim, 2013) and drive denitrification (Ding et al., 2003; Smart et al., 1997). Soil temperatures decrease due to the higher leaf area and related shade (Kim, 2013).

1.3.8. Land-use change

Land-use change is very important for GHG emissions from soils, particularly when forests, grass and peat lands are being converted to agricultural land (Fig. 1). Within the first 30 years after turning forest into agricultural land, 30–35% of the soil carbon stored in the top 7 cm is lost, while below plough depth no changes are recognizable (Degryze et al., 2004). The global C-sequestration potential of agricultural soils amounts to 0.73–0.87 Pg C yr^{-1} (Lal and Follett, 2001). These aspects are further elucidated in Section 2.

2. Effects of land-use and land-cover types on soil GHG emissions

The global land surface of 149,430,000 km^2 is covered by 31.5% grassland and pastureland (including shrubs, herbaceous and sparse vegetation; ca. 47,070,500 km^2), followed by 27.7% tree-covered areas (including forestland; ca. 40,794,000 km^2), 15.2% barren land (including [semi]arid areas; ca. 22,713,000 km^2), 12.6% cropland (with annual and perennial crops; ca. 18,828,000 km^2), 9.7% snow and glaciers, 2.7% water bodies, wetlands and mangroves (ca. 4035 km^2), and 0.6% artificial surfaces (including urban environment), according to FAO (2015) and Latham et al. (2014). The most prominent land-cover types are examined in this paper—and include the latest discussion on wetlands and their supposedly particularly important level of GHG release. Neither snow- and glacier-covered areas, nor sealed artificial surfaces deliver respiration rates of any global relevance. Thus, for some calculations these areas are excluded and an effective land surface of 135.39 Mio km^2 is being used (Bahn et al., 2010). Fig. 2 presents a world map of the locations of the studies reviewed in this work and indicates the land-cover types. That map suggests a certain bias with respect to representative geographical coverage of the compiled data. The vast majority of studies have delivered data from Europe, followed by P.R. China and the United States. The southern hemisphere is comparatively underrepresented.

Soil organic carbon (SOC) accumulation largely depends on vegetation cover. Any change in land use may significantly alter related source or sink characteristics for atmospheric CO_2 and other GHGs (Poeplau and Don, 2013; Sainju et al., 2008). Plant species differ in root depth and spatial distribution (Breecker et al., 2012). Therefore, soil-flux data from studies involving various land use, land cover, and different climate regions need to be evaluated separately in order to properly characterize soil degassing (Bahn et al., 2010). In addition, agricultural practices such as tillage and fertilization have to be considered. Food systems alone – everything from growing plants to the disposal of biomass – contribute to 19–29% of global anthropogenic GHG emissions. Of this, 80–86% relate to agricultural production (including indirect emissions associated with land-cover change), albeit with significant regional variation (Vermeulen et al., 2012). Short rotation forestry (SRF) has been integrated into the cropland section, since sites are harvested at 2–3-year intervals and replanting is done about every 20 years.

Wetlands (albeit with limited land cover: 2.7%) show the highest average absolute emission rates; considerably higher than all other land cover types. Forestland, grasslands, croplands and barren lands follow in this order with decreasing emission rates. Yet, individual data of all land-cover types show a high variance (Fig. 3). This reflects climate zone and land management-related conditions and clearly demonstrates the key drivers for soil degassing. Using a conservative average of 300 mg CO_2e m^{-2} h^{-1} for all land-cover types combined as compiled from our literature review, global annual soil emissions of $\geq$350 Pg CO_2e result, corresponding to roughly 21% of the estimated global soil C and N pools (Fig. 3).

2.1. Grassland, including pastureland and savannahs

Permanent grasslands cover about 31.5% of the total global land area and about 70% of the total agricultural expanse (FAO, 2014, 2015). This high grassland cover percentage and the above average GHG emissions underline the potential of grassland and pastureland in mitigating global warming. Table 1 clearly corroborates the inadequate data situation and the strong bias towards temperate climate regions.

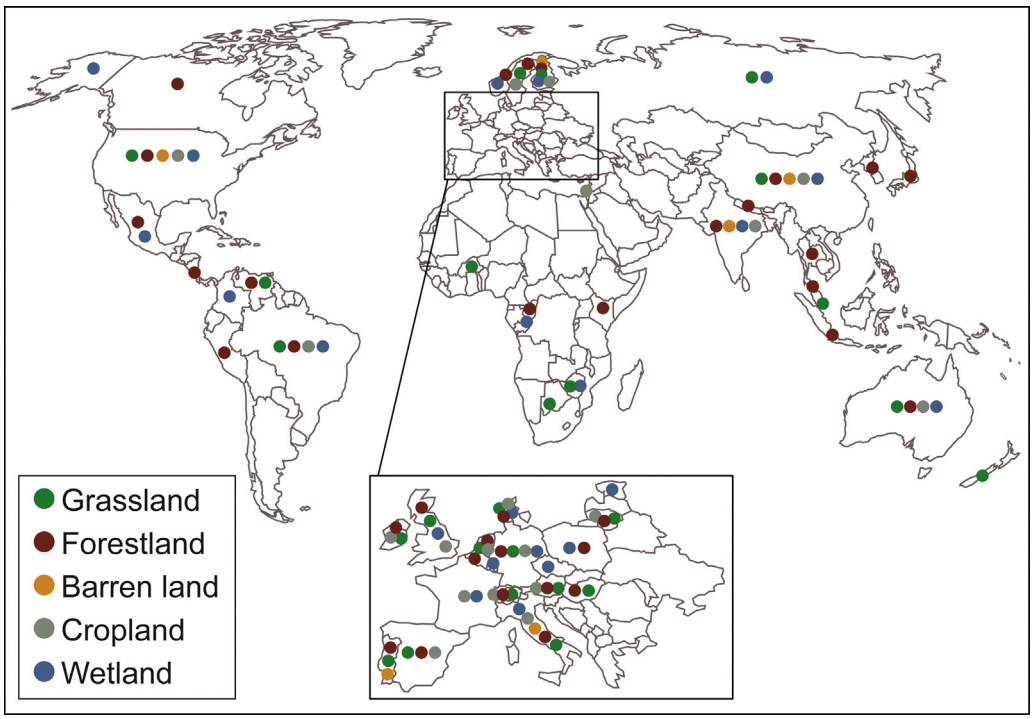

Fig. 2. Study sites worldwide of GHG soil emissions reviewed in this paper, indicating the respective land-cover type.

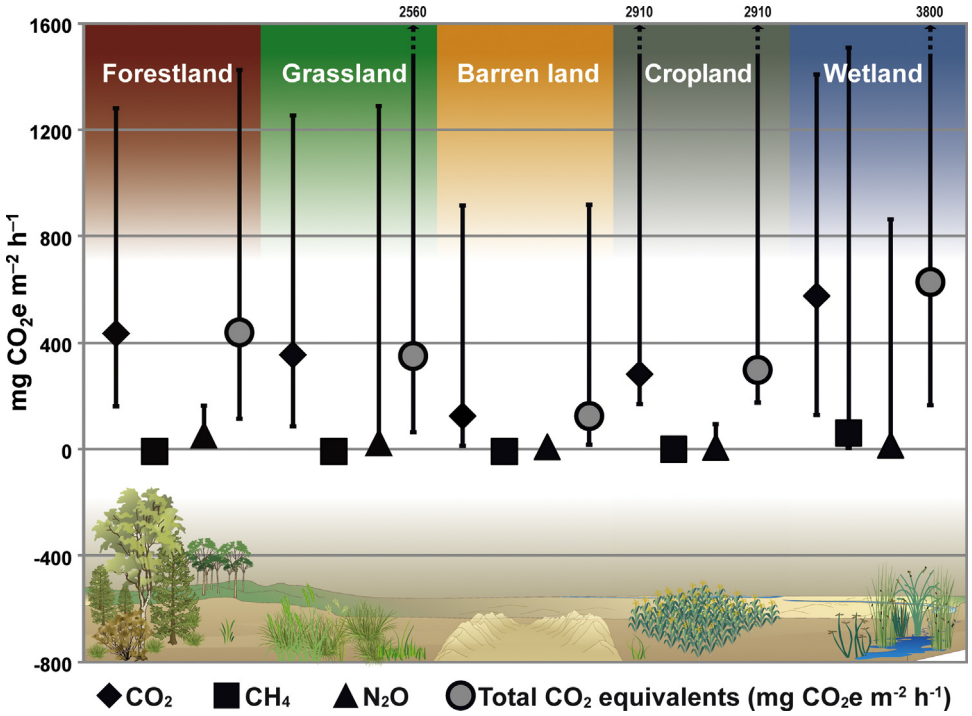

Fig. 3. GHG emissions (CO_2-eq) of CO_2, N_2O and CH_4 from soils with different land cover: grassland (n = 47), forestland (n = 22), barren land (n = 17), cropland (n = 41), and wetland (n = 67). Median values for the sub-collectives are shown with the symbols, the range is indicated with solid lines. Only consistent data discussed in this paper have been integrated. Some maximum values exceed the upper concentration limits (see text).

2.1.1. CO_2-soil respiration

CO_2-emission rates from perennial grasslands and from Tundra were estimated at 1.54–1.61 and 0.48–0.57 μmol CO_2 m^{-2} s^{-1}, respectively (Raich and Potter, 1995). Under a semi-arid, temperate climate (grassland in Inner Mongolia), the highest CO_2-emission rates occurred during summer (3.09 ± 0.23 μmol CO_2 m^{-2} s^{-1}) and the lowest rates during the snow-covered winter period (1.26 ± 0.15 μmol CO_2 m^{-2} s^{-1}) with Q_{10} values ranging between 1.63 and 1.74, depending on the fertilization system (Peng et al., 2011). WFPS plays an important role in CO_2 emission rates; 20–40% WFPS in grassland soils results in maximum emissions (Schaufler et al., 2010). Higher soil respiration rates were measured at managed grassland sites in Finland (peak values 11.6–25 μmol CO_2 m^{-2} s^{-1}; seasonal average 1.82–7.95) with the higher values

recorded in peat soils and the lower ones in sandy and clayey soils (Lohila et al., 2003). Fertilizing grasslands enhances soil respiration. A 24–57% increase was measured in the annual total soil respiration in the first year after fertilizing under semi-arid temperate climate conditions (Inner Mongolia; Peng et al., 2011).

2.1.2. N_2O emissions

The influence of livestock grazing on pastureland showed that the type of animals grazing on pasture does influence soil N_2O emissions. Sheep-grazed pasture sites emitted less N_2O than cattle-grazed ones, while the lowest emissions were reported from non-grazed pastures (1.63–2.44, 4.07–4.89 and 0.41–0.82 $\mu mol\ N_2O\ m^{-2}\ h^{-1}$; Saggar et al., 2008). However, Wolf et al. (2010) found during a year-round study in Inner Mongolia on steppe grassland that the pulse of spring–thaw emission was lowest at sites with intense grazing ($0.03 \pm 0.03\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$) and highest on non-grazed sites ($0.45 \pm 0.03\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$). This applies also for the annual N_2O-emission budget, since spring–thaw emissions contribute most to the annual budget. A sheep-grazing site (permanent pasture) showed significantly lower N_2O emission rates ($0.68\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$) than conventional tillage (CT) and no-tillage (NT) sites under winter oat ($3.75\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$) and fodder maize cultivations ($4.89\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$) and did not significantly differ between CT and NT sites (Choudhary et al., 2002). For a good compilation of N_2O and CH_4 emissions from savannahs and seasonally dry ecosystems, see Castaldi et al. (2006) who generally found low emissions for N_2O and an uptake for CH_4.

2.1.3. CO_2, N_2O, and CH_4 dynamics

Establishing grasslands on drained organic soils enhanced GHG emissions (Kasimir-Klemedtsson et al., 1997). Emission rates from this land-use type in Finland were $7.98\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$ and $1.52\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$ (Maljanen et al., 2004). Using reclaimed coal mine areas to establish managed grassland (fertilized and harvested hay fields) in south-eastern Ohio (USA) emitted 3.18–5.30 $\mu mol\ CO_2\ m^{-2}\ s^{-1}$, 1.35–1.60 $\mu mol\ CH_4\ m^{-2}\ h^{-1}$ and $0.63\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$ (Jacinthe and Lal, 2006). Our group determined similar values for soil respiration on a winter rye field (measurements took place after seeding between plant rows with plant height of 5 cm) at a former lignite site in Lusatia, Germany, with $4.41 \pm 1.15\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$ (Weiß, 2011). The non-fertilized meadow at the reclaimed mine in south-eastern Ohio (USA) showed emission and uptake rates from 2.6 to 3.18 $\mu mol\ CO_2\ m^{-2}\ s^{-1}$, −3.54 to −1.63 $\mu mol\ CH_4\ m^{-2}\ h^{-1}$ and $0.85\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$ (Jacinthe and Lal, 2006).

Insignificant CH_4 fluxes were reported from most of the studied grassland sites (less than $\pm 9.5\ E^{-4}\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$; Skiba et al., 2009). The Inner Mongolian steppe ecosystem was a CH_4 sink (average uptake $-2.36\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$; Liu et al., 2009a).

From the compiled data, it can be calculated that grassland soils emit on average 350 mg $CO_2e\ m^{-2}\ h^{-1}$—the third highest flux rate after wetlands (which represent a small area only) and forestlands (Fig. 3). Given the wide range of results that partly cover several orders of magnitude and the distinct lack of data for Mediterranean and (sub)tropical environments, particularly in the southern hemisphere, it appears premature to compile reliable global assessment data from the existing studies – considerable effort is needed to bridge these gaps.

2.2. Tree-covered land, including forestland

Tree-covered land including forestland covers about 28% of the terrestrial surface. Yet, tree cover and related leaf area indices vary widely, from extremely sparse vegetation (e.g., Caatinga in north-eastern Brazil and similar savannah-type vegetation—which we discuss due to their dominant characteristics in the preceeding section on grassland) to multi-storey tropical rainforests with extremely dense foliage. A similarly high variability occurs with the respective soil carbon and nitrogen pools. Dixon et al. (1994) compiled global forest ecosystem pool data. Their ranges extended from 471 Pg at high latitudes via 100 Pg at mid latitudes to 216 Pg at low latitudes, with a total of 787 Pg. Available data are by far not yet sufficient to obtain a highly representative and precise picture of related GHG emissions, yet decent estimates are available (Table 2). Particular focus is given to longer-term measurements, since many studies deliver relatively short (and potentially less representative) data series.

2.2.1. Global flux estimates from forests

2.2.1.1. CO_2-soil respiration. Bond-Lamberty and Thompson (2010) summarized soil-respiration (R_S) measurements from 818 studies with 3379 records for the years 1961–2007. Mean (±1 s.d.) R_S was 1.01 ± 0.60 for boreal (ca. 15 Mio. km^2), 1.97 ± 1.11 for temperate (ca. 7 Mio. km^2), and $3.40 \pm 1.67\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$ for tropical ecosystems (ca. 19 Mio. km^2)—resulting in a global average of $2.28 \pm 50\%\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$ for the global forestland. A more detailed review of the CO_2 balance of forests is available from Luyssaert et al. (2007), based on publicly available data and including measurement data and model results (Table 2). Schulze et al. (2010) reviewed flux measurements of trace gases of CO_2 and other gases within the CarboEurope network. Forest mean autotrophic respiration was around $1.56 \pm 0.23\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$ and heterotrophic respiration was determined at $0.97 \pm 0.28\ CO_2\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$. Oertel et al. (2015a) present $3.5\ \mu mol\ CO_2\ m^{-1}\ s^{-1}$ (autotrophic + heterotrophic) for forests in Saxony, Central Europe, in summer. Monson et al. (2006) compiled winter forest soil respiration fluxes between 0.05 and $0.25\ \mu mol\ CO_2\ m^{-2}\ s^{-1}$.

2.2.1.2. N_2O-soil emissions. Dalal and Allen (2008) reviewed N_2O emission data from forest soils. They found that the mean N_2O-emission rate from tropical forest soils ($1.23 \pm 2.09\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$) was at least 2–3 times higher than those from temperate forest soils ($0.41 \pm 0.15\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$). These results were based on 22 studies in tropical forests and 15 studies in temperate forests. The dataset was very limited for boreal forest soils; only four studies were included. The average annual flux of N_2O emissions from boreal forest soils was small with $0.10 \pm 0.08\ \mu mol\ N_2O\ m^{-2}\ h^{-1}$.

2.2.1.3. CH_4-soil uptake. Forests are generally sinks for CH_4. This is the key result of CH_4 flux measurements for different ecosystems, as reviewed by Dalal and Allen (2008). Mean tropical forest CH_4 consumption was given with $-2.75 \pm 0.42\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$ (based on 13 studies). Temperate forests consumed on average $-3.42 \pm 0.44\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$ (18 studies), with deciduous forest soils showing higher CH_4 consumption rates than coniferous forest soils and undisturbed forest soils showing higher CH_4 consumption rates than disturbed forests or regrowth and fertilized forests (Smith et al., 2000). Studies in boreal forests are still scarce (4 studies). Their annual average CH_4 flux is given by $-1.38 \pm 2.80\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$, with a range from $-9.38\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$ to $3.53\ \mu mol\ CH_4\ m^{-2}\ h^{-1}$. The latter value represents one boreal aspen forest in Canada that produces CH_4 (Simpson et al., 1997).

2.2.2. Results from long-term soil flux measurements

The interannual variability of GHG emissions is controlled by climate variability and ecosystem-internal parameters (Phillips et al., 2010; Urbanski et al., 2007; Fig. 1). Therefore, soil flux datasets comprising only sporadic measurements or not covering complete seasons and/or years may not be representative for a site (Luo

Table 2
Mean soil respiration fluxes in μmol CO_2 m^{-2} s^{-1} with standard deviations (recalculated after Luyssaert et al., 2007).

Biome	Ecosystem respiration R_e	Autotrophic respiration R_a	Heterotrophic respiration R_h
Boreal humid evergreen	2.18 ± 0.30	1.29 ± 0.22	1.01 ± 0.11
Boreal semiarid evergreen	1.94 ± 0.10	1.43 ± 0.09	0.65 ± 0.07
Boreal semiarid deciduous	2.72 ± NA	2.00 ± 0.08	0.73 ± 0.08
Temperate humid evergreen	3.53 ± 0.15	2.51 ± 0.30	1.11 ± 0.08
Temperate humid deciduous	2.77 ± 0.17	1.78 ± 0.23	1.02 ± 0.07
Temperate semiarid evergreen	2.92 ± 0.69	1.32 ± 0.15	0.79 ± 0.04
Mediterranean warm evergreen	2.94 ± 0.26	1.62 ± NA	1.52 ± 0.26
Tropical humid evergreen	8.08 ± 0.43	6.13 ± 0.38	2.32 ± 0.25

et al., 2012). Long-term datasets are scarce, but are very important to understand the biosphere-atmosphere exchange of C and N compounds. Multi-year datasets are mostly available for temperate forests, e.g., Harvard forest, USA (Phillips et al., 2010), Höglwald spruce site, Germany (Luo et al., 2012) and Solling, Germany (Borken et al., 2002). Here we focus on the results of these long-term measurements to obtain information about the relationships between environmental parameters and soil trace gas fluxes.

2.2.2.1. CO_2 fluxes. Cumulative values of soil respiration CO_2 fluxes tended to be highest in years with below average annual mean soil temperatures and high N_2O emissions at the Höglwald site (e.g., the years 1995/1996 and 2005/2006 with cold winters and springs; Luo et al., 2012). Similar findings were reported for the Harvard Forest site (Phillips et al., 2010). It may thus be concluded that climatic conditions during the dormant or early growing season might determine annual soil respiration rates in temperate forest ecosystems. Increased soil respiration rates may also result from increased plant carbon assimilation and associated litter production in the previous year (Luo et al., 2012). Their data also showed a close correlation of gross primary production and soil respiration. Years with extended summer droughts revealed reduced annual soil respiration rates at all three sites (Borken et al., 1999; Luo et al., 2012; Savage and Davidson, 2001).

2.2.2.2. NO and N_2O emissions. Rather high NO (2.44–4.34 μmol NO m^{-2} h^{-1}) and N_2O (0.08–1.22 μmol N_2O m^{-2} h^{-1}) emission rates were found for the Höglwald site (Luo et al., 2012). These high fluxes are related to the elevated nitrogen deposition at the Höglwald Forest, lying in a range of 25–30 kg N ha^{-1} yr^{-1} for more than three decades (Luo et al., 2012). Similar results were found for NO fluxes at the Speulderbos Douglas fir forest site in the Netherlands, also receiving high loads of atmospheric N input (Pilegaard et al., 2006), whereas sites with low atmospheric nitrogen input (<10 kg N ha^{-1} yr^{-1}) showed much lower mean N_2O fluxes (0.08 μmol N_2O m^{-2} h^{-1}; Bowden et al., 1991; Castro et al., 1993). Results from various forest sites across Europe were summarized within the EU-funded NOFRETETE project (Pilegaard et al., 2006). Their results show that the correlation between atmospheric N-input and nitrogen emissions is highly correlated for NO and not significantly correlated for N_2O. However, a strongly negative correlation between N_2O emissions and the C/N ratio was found. Results from the Höglwald site show that occasional freeze–thaw pulse emissions dominate the annual N_2O fluxes in certain years (Luo et al., 2012). In years with the highest annual N_2O fluxes of approximately 1.22 μmol N_2O m^{-2} h^{-1}, i.e., 1996 and 2006, pulse emissions during freeze–thaw periods contributed 88% in 1996 and 87% in 2006 to the total annual emissions (Luo et al., 2012).

2.2.2.3. CH_4 uptake in soils. Compared to other land-cover types, temperate forest soils showed the highest CH_4-uptake rates (up to −12.50 μmol CH_4 m^{-2} h^{-1}; Smith et al., 2000). However, rates of atmospheric CH_4 oxidation were highly variable in a range of −0.08 to −13.75 μmol CH_4 m^{-2} h^{-1} (e.g., Ambus and Christensen, 1995; Borken and Brumme, 1997; Butterbach-Bahl et al., 1998). Forests receiving high loads of atmospheric nitrogen and acid deposition show low CH_4 oxidation rates (Brumme and Borken, 1999; Butterbach-Bahl et al., 2011; Chan et al., 2005; Liu and Graever, 2009; Luo et al., 2012). The long-term CH_4-flux measurements at the Höglwald site showed only weak negative correlations with soil moisture and weak positive correlations with soil temperature (Luo et al., 2012).

Drawn from the compiled data (again with similar imperfections as with all other land-cover types), it can be calculated that forest soils emit on average 438.9 mg CO_2e m^{-2} h^{-1}. As forest soils exhibit the second highest flux rate after wetlands with their small area coverage, this represents a very important global flux (Fig. 3).

2.3. Bare soil/barren lands

Here, all types of barren land from all climate zones, including tundra, are included in the discussion. The global contribution of CO_2 emissions from tundra and arid ecosystems amounts to about 10% (Raich and Potter, 1995). The sensitivity of these ecosystems to soil respiration is lower than that of other ecosystems, apparent from its low Q_{10} value of 1.39. Soil respiration rates in arid areas under continental climate ranged between −0.24 and 1.82 μmol CO_2 m^{-2} s^{-1} (Zhang et al., 2009), raising explicit open questions about what determined the negative rates and suggesting additional research. Emission rates were lower at a desert shrubland in Inner Mongolia under semi arid continental climate with 0.26 ± 0.04 and 0.58 ± 0.07 μmol CO_2 m^{-2} s^{-1} (Jin et al., 2009). Scrubland ecosystem sites under arid climate in north-western China showed soil respiration rates between 0.57 and 0.86 μmol CO_2 m^{-2} s^{-1} at places with natural vegetation and 1.08 μmol CO_2 m^{-2} s^{-1} at sites with non-native vegetation (Lai et al., 2012).

A desert-shrub ecosystem located at the transition between arid and semi-arid climates showed emission rates between 0.23 and 2.56 μmol CO_2 m^{-2} s^{-1} (Wang et al., 2014). Higher soil respiration rates and elevated Q_{10} values were reported from shrubland and bare land in the Loess Plateau of China under semi-arid conditions with emission rates of 2.03 and 2.66 μmol CO_2 m^{-2} s^{-1}, and Q_{10} values of 1.43 and 2.25, respectively (Shi et al., 2014). Annual GHG emissions were determined from different biological soil crusts in the Tengger desert ecosystem, China. Emission rates ranged between 0.24 and 0.66 μmol CO_2 m^{-2} s^{-1}, and yielded −1.24 μmol CH_4 m^{-2} h^{-1} and −0.08 μmol N_2O m^{-2} h^{-1} (Hu et al., 2014). Under semi-arid temperate desert ecosystems in the eastern Inner Mongolia Plateau in northern China, CH_4 in soils over two years showed a mean uptake rate of 3 μmol CH_4 m^{-2} h^{-1} during the growing season (Hou et al., 2012) (Table 3).

Correia et al. (2014) measured a soil respiration of 2 μmol CO_2 m^{-2} s^{-1} from shrubland sites under Mediterranean climate (Table 3). Similar to the situation explored in Section 2.1 on grassland soil emissions, data for Mediterranean and (sub)tropical environments are underexplored.

Soil respiration rates at sites of different land cover (grass, woody shrub and tree landscapes) under dry conditions in

Table 3
Soil emission rates in μmol CO_2 m^{-2} s^{-1} and μmol X m^{-2} h^{-1} for CH_4 and N_2O from bare soil, (semi)deserts and related land under temperate, mediterranean and (sub)tropical climates. Rates are given without standard deviations. Ranges have been rounded for clarity (see original literature and text for more details).

	Temperate		Mediterranean		(Sub)Tropical	
Species	Rate	Comment	Rate	Comment	Rate	Comment
CO_2	−0.24[a] to 1.82	Continental desert[13]	2.4–3.1	Shrubland[2]	0.035–0.14	Dry bushland[11]
	0.26–0.58	Shrubland[7]	2	Cork oak[1]	0.2–0.96	Wet bushland[11]
	0.57–0.86	Arid scrubland, native vegetation[8]			1.60–9.89	Tree ecosystem[3]
	1.08	Same, non-native vegetation[9]				
	0.23–2.56	Desert shrubland[12]				
	2.03–2.66	Loess shrubland[10]				
	0.24–0.66	Desert soil[6]				
	3.66	Bare soil[9]				
Range CO_2	0.2–3.7 (min. −0.24)		2.4–3.1		0.035–1.6 (max. 9.9)	
CH_4	−1.24	Desert soil[6]				
	−3	Desert soil[5]				
	−0.52	Bare soil[9]				
Range CH_4	−3 to −0.52		No data		No data	
N_2O	−0.08	Desert soil[6]			0.2	Desert soil[4]
	9.57	Bare soil[9]				
Range N_2O	−0.08 to 10		No data		No data	

[a] See text.
[1] Correia et al. (2014) Portugal.
[2] de Dato et al. (2010) morning and midday emission rate Sardinia, Italy.
[3] Gnaana Saraswathi et al. (2008) semi-arid India.
[4] Hall et al. (2008) Sonora desert, Arizona.
[5] Hou et al. (2012) Inner Mongolia, China.
[6] Hu et al. (2014) Tennger desert, China.
[7] Jin et al. (2009) Inner Mongolia China.
[8] Lai et al. (2012) China.
[9] Maljanen et al. (2004) Finland.
[10] Shi et al. (2014) China.
[11] Thomas et al. (2011) Botswana.
[12] Wang et al. (2014) China.
[13] Zhang et al. (2009) China.

Botswana ranged between 0.035–0.14 μmol CO_2 m^{-2} s^{-1}. The values increased to 0.2–0.96 μmol CO_2 m^{-2} s^{-1} when the soils received 50 mm of precipitation (Thomas et al., 2011). Higher soil respiration rates were measured in a tree ecosystem under semi-arid conditions in India, with 1.60 ± 0.36 in winter and 9.89 ± 0.78 μmol CO_2 m^{-2} s^{-1} in summer (Gnaana Saraswathi et al., 2008).

Not surprisingly, bare soil demonstrates the lowest average fluxes from all land-cover types (Fig. 3). Calculations from the compiled data on bare soil in (semi)deserts and related lands results with an average emission of 126.7 mg CO_2e m^{-2} h^{-1}. Such a flux would represent ca. 6.5 % of the global emissions as calculated from the data in this work.

2.4. Cropland

Agriculture influences global warming due to related direct and indirect GHG emissions from C and N dynamics (Hellebrand et al., 2003; Wang et al., 2013b). Arable lands and lands under permanent crops cover about 17 Mio. km^2 or 12.6% of the global land surface (FAO, 2014). Other assessments deliver 15 Mio. km^2 or 12% with an error margin between 12.2 and 17.1 Mio km^2 (Ramankutty et al., 2008). Deforestation and other land-use changes to increase the surface area for crop production further contribute to global warming. In addition, croplands generally stand for intensive agricultural management (e.g., intensive tillage, application of fertilizers and other chemicals), enhancing GHG emissions. Cereals and oil crops (e.g., seed cotton and rapeseed/canola) are the World's most prominent crops (FAO, 2015) (Table 4).

Plant residues are being used as soil cover to decrease erosion, maintain soil humidity and to improve soil quality (mulching). This may influence soil emission rates. Cotton residuals with urea fertilization decreased N_2O emission by 25% from fertilized fields without (2.05 ± 0.14 μmol N_2O m^{-2} h^{-1}) and with cotton residue (1.5 ± 0.11 μmol N_2O m^{-2} h^{-1}), according to Muhammad et al. (2011). The effect on soil emissions depends on the chemical properties of the residue cover. Yet, all types of plant residues (sugarcane trash, maize and sorghum straw, cotton residues and lucerne) increased the cumulative CO_2 emission by about a factor of 3 (Muhammad et al., 2011). Bean fertilization (*Phaseolus vulgaris* L.) under sub-humid temperate climate conditions did not significantly change soil respiration. However, soil respiration rates increased by 137.5% in soils supplemented with vermicompost and by 237.5% in soils with vermicompost and wastewater sludge (WWS). Different composts or fertilizer additions resulted in higher N_2O-soil emissions that spanned one to two orders of magnitude: urea (factor 19), low dose of WWS (factor 19), vermicompost (factor 47) and high dose of WWS (factor 97) according to Fernández-Luqueño et al. (2009).

2.4.1. Temperate climate

Average soil emission data from the growing season vary by one order of magnitude in temperate regions with a spread of 0.45–8.2 μmol m^{-2} s^{-1} CO_2, 0.16–0.31 μmol m^{-2} h^{-1} CH_4 and 0.03–10.7 μmol m^{-2} h^{-1} N_2O; the latter high value refers to a fertilization experiment (Table 5).

Methane uptake in soils decreases with increasing precipitation and increases with addition of N fertilizer; soils with wheat and oilseed rape cultivation under different rotations take up atmospheric CH_4 in a range of 0.25 ± 1.29 to 0.31 ± 0.70 μmol CH_4 m^{-2} h^{-1} (Drewer et al., 2012). Re-forestation of organic soil croplands did not change CH_4 and N_2O emissions, but showed reduced CO_2 emissions (Maljanen et al., 2007).

The rate of soil N_2O emissions was five times higher from annual bioenergy crop fields (wheat and oilseed rape with 0.30 ± 0.79 to 1.15 ± 2.51 μmol N_2O m^{-2} h^{-1}) than perennial crops such as the willow short rotation crop (SRC) and grass, *Miscanthus*, with 0.01 ± 0.21 to 0.13 ± 0.97 μmol N_2O m^{-2} h^{-1} (Drewer et al.,

Table 4
Top 10 cultivated crops in the World for the year 2012 (FAO, 2015).

No.	Crop	Cultivated area (1000 ha)	No.	Crop	Cultivated area (1000 ha)
1	Wheat	215,489	6	Barley	49,526
2	Maize	177,380	7	Sorghum	38,162
3	Rice, paddy	163,199	8	Seed cotton	34,700
4	Soybeans	104,997	9	Rapeseed	34,085
5	Pumpkins for fodder	84,044	10	Millet	31,758

Table 5
Soil emission rates in μmol CO_2 m^{-2} s^{-1} and μmol X m^{-2} h^{-1} for CH_4 and N_2O from cropland under temperate, mediterranean and (sub)tropical climates. Rates are given without standard deviations. The ranges have been rounded for clarity (see original literature and text for more details).

	Temperate		Mediterranean		(Sub)Tropical	
Species	Rate	Crop	Rate	Crop	Rate	Crop
CO_2	5.25	Wheat[21]	1.29	Various[17]	7.01; 4.86	No tillage[13]
	1.33	Rotations[3]	1.12	Olives[1]	13.76; 17.86	Conventional[13]
	1.83	Rotations[4]	1.71	Fallow[1]	1.55	Sugar cane[18]
	6.00	Maize[21]	1.45	Wheat[7]	0.79	Vegetables[10]
	0.45–3.86	Barley and potato, mineral soil[14]	2.21	Tomato[11]		
	2.5–8.18	Barely and potato on peat soil[14]				
	2.0	Corn, black soil[20]				
	7.52	Barley[15]				
	0.76–2.20	Various[17]				
	0.68–1.46	Winter[12]				
	5.77; 2.02	Warm; cold[22]				
Range CO_2	0.45–8.2		1.1–1.7		7.0–17.9	
CH_4	−0.41	Willow[5]	0.09–0.13	Various[17]		
	−0.16	Miscanthus[5]	−0.13	*Vicia sativa*[19]		
	−0.31	Rotations[20]	−0.45	*Vicia sativa*[19]		
	−0.25	Rotations[4]	−0.22	*Vicia sativa*[19]		
Range CH_4	−0.41 to −0.16		−0.45 to 0.13		No data	
N_2O	0.30	Rotations[20]	0.44–1.36	Various[17]	0.27–2.26	Tea fields[8]
	1.15	Rotations[4]	0.02	*Vicia sativa*[19]	1.5–2.1	Experimental[16]
	0.01	Willow[6]	0.05	*Vicia sativa*[19]	0.24	Oat, maize[2]
	0.13	Miscanthus[6]	0.02	*Vicia sativa*[19]		
	4.26	Barley[15]				
	2.50–10.71	Fertilizations[9]				
	1.03–1.26	Various[17]				
Range N_2O	0.01–4.3 (max. 10.7)		0.02–1.36		0.27–2.26	

[1] Almagro et al. (2009) Murcia Spain.
[2] Bayer et al. (2014) southern Brazil.
[3] Drewer et al. (2012) 3-yr wheat-wheat-oilseed (rapeseed) rotations England.
[4] Drewer et al. (2012) rapeseed and wheat rotation England.
[5] Drewer et al. (2012) England.
[6] Drewer et al. (2012) annual bioenergy fields England.
[7] Eshela et al. (2014) rainfed wheat, Israel; rainy season only, dry season: below LLD.
[8] Gogoi and Baruah (2011) India.
[9] Hellebrand et al. (2003) Germany.
[10] Iqbal et al. (2009) China monsoon area.
[11] Kallenbach et al. (2010) Caifornia.
[12] Koch (2015), Leppin (2015), Schach (2015) winter conditions.
[13] La Scala et al. (2006) sugarcane one day/month after tillage, Brazil.
[14] Lohila et al. (2003) on mineral and peaty soils Finland.
[15] Maljanen et al. (2004) Finland.
[16] Muhammad et al. (2011) Australia.
[17] Schaufler et al. (2010) European continental and maritime climate.
[18] Teixeira et al. (2013) São Paulo state, Brazil.
[19] Tellez-Rio et al. (2014) no till, minimum till, conventional till, Spain.
[20] Wei et al. (2014) Northeastern China.
[21] Zhang et al. (2013) *Triticum aestivum* and *Zea mays*, China.
[22] Zurba and Matschullat (2015) median values for SRF in the vegetation period and in winter (December and January).

2012). High N_2O-emission rates (2.5–10.71 μmol N_2O m^{-2} h^{-1}) were reported after 1–2 months of fertilizing bioenergy sites (Hellebrand et al., 2003).

2.4.2. Mediterranean climate

Dry Mediterranean conditions prevail over about 2% of the total global land area. The mean soil respiration rate at rain-fed olive grove sites under such conditions was 1.12 ± 0.12 μmol CO_2 m^{-2} s^{-1} (Almagro et al., 2009). Yet, cultivating bioenergy crops may negatively affect CO_2 savings if other GHG emission rates are high, e.g., fertilizing maize crops may cause a >70% increase in soil N_2O emissions (Sanz-Cobena et al., 2014b). Abandoned agricultural areas still emit GHGs. A mean respiration rate of 1.71 ± 0.09 μmol CO_2 m^{-2} s^{-1} was determined over a former cereal field that lay fallow for about 25 years (Almagro et al., 2009). The authors found a positive correlation between soil respiration and soil temperature only when soil water content was above 15%.

2.4.3. (Sub)tropical climate

Sugar cane is an important crop in Brazil, contributing to more than 37% of the global harvested area (FAO, 2015. The related soil respiration rate was 7.01 shortly after tillage, and measured

13.76 μmol CO_2 m^{-2} s^{-1} in no-tillage (NT) and conventional tillage (CT) sugar cane fields. One month after tillage, emission rates changed to 4.86 μmol CO_2 m^{-2} s^{-1} from NT and 17.87 μmol CO_2 m^{-2} s^{-1} from CT (La Scala et al., 2006). Measured N_2O fluxes from fertilized tea fields in North Eastern India (humid subtropical conditions) showed emissions ranging between 0.27 and 2.26 μmol N_2O m^{-2} h^{-1} (Gogoi and Baruah, 2011).

From the compiled data, it can be calculated that croplands, composing 12.6% of the global land area, emit on average 297.7 mg CO_2e m^{-2} h^{-1}, which is the second lowest flux rate of all land-cover sites (Fig. 3).

2.5. Wetlands, including rice paddy soils

Wetlands deliver important ecosystem services. They cover 2.6% of the Earth's surface and are equivalent to 25% of the total anthropogenic and natural sources of methane (FAO, 2015; Latham et al., 2014). Different parameters such as water depth, temperature and type of plant (floating, submersed, emersed, etc.) affect CH_4 emissions (Schaufler et al., 2010; Whalen, 2005). Of the global CH_4 emissions from wetlands, 75% are estimated to stem from tropical latitudes (EPA, 2010; Whalen, 2005). The average emission rates from natural wetlands were determined as highest in tropical areas (508 μmol CH_4 m^{-2} h^{-1}), lower in temperate areas (391 μmol CH_4 m^{-2} h^{-1}) and lowest in Nordic areas (104 μmol CH_4 m^{-2} h^{-1}; Cao et al., 1998). Average CH_4 emissions of 326 μmol CH_4 m^{-2} h^{-1} emerged from temperate wetlands and 216 μmol CH_4 m^{-2} h^{-1} from subarctic/boreal wetlands in both Scandinavia and North America (Turetsky et al., 2014). Methane emissions also decreased from temperate via boreal to subarctic zones in bogs and rich fens, whereas nutrient-poor fens showed highest emissions in the subarctic zone (Turetsky et al., 2014). Grunwald et al. (2012) found the highest rate of CH_4 emissions from rice paddies in the mediterranean zone (278 μmol CH_4 m^{-2} h^{-1}) and the lowest from temperate wetlands (85 μmol CH_4 m^{-2} h^{-1}) and from wetlands in cold climates (80 μmol CH_4 m^{-2} h^{-1}). Aselmann and Crutzen (1989) provided a global estimate for wetland CH_4-emission rates in bogs (39), shallow lakes (112), fens (208), swamps (219), floodplains (260), marshes (659) and rice fields (807 μmol CH_4 m^{-2} h^{-1}) (Table 6). Methane emissions from soils in the subartic zone can vary largely and depend on presence of permafrost (124.74 μmol CH_4 m^{-2} h^{-1}) or its absence (252.08 μmol CH_4 m^{-2} h^{-1}) (Turetsky et al., 2014).

Tropical wetlands in northern Australia yielded average emission rates of 2.00 μmol CO_2 m^{-2} s^{-1} and 2108 μmol CH_4 m^{-2} h^{-1} (Bass et al., 2014). CH_4 fluxes from natural wetlands, independent of climate zone, were 260.4 μmol CH_4 m^{-2} h^{-1} (Whalen, 2005). Wetlands cultivated with different plants were sinks for CH_4 (-1.84 ± 0.18 μmol CH_4 m^{-2} h^{-1}) and sources for CO_2 (2.55 ± 0.13 μmol CO_2 m^{-2} s^{-1}) and N_2O (0.85 ± 0.33 μmol N_2O m^{-2} h^{-1}) (Schaufler et al., 2010). The highest CH_4 emissions were registered with 95 % WFPS. From wetlands located in an agricultural catchment in Denmark, N_2O emission rates ranged between −1.6 and 4.4 μmol N_2O m^{-2} h^{-1} (Audet et al., 2014). Kluber et al. (2014) determined soil emission rates of 4.47 μmol CO_2 m^{-2} s^{-1} and 4.73 μmol N_2O m^{-2} h^{-1} from natural wetlands in the United States, and 3.29 μmol CO_2 m^{-2} s^{-1} and 14.2 μmol N_2O m^{-2} h^{-1} from one former wetland converted to agriculture in the US Mid-Atlantic region.

About 97% of the global paddy rice is produced in wetlands areas (FAO, 1998). Peak N_2O-fluxes from rice fields in Italy were observed following fertilization at eight locations, revealing emission averages of 42.4 ± 46.1 μmol N_2O m^{-2} h^{-1} which decreased after flooding to around zero (Skiba et al., 2009). Average CH_4 emissions from rice paddies were 130.2–1563 μmol CH_4 m^{-2} h^{-1} (Cao et al., 1998). Bhattacharyya et al. (2013) studied the effect of increasing atmospheric CO_2 (550 ± 30 μmol mol^{-1}) on CH_4 and N_2O emissions at lowland rice fields under tropical climate in eastern India: 26% (CH_4) and 24.6% (N_2O—N) higher emissions were registered. Different tillage systems were compared in rice fields under humid mid-subtropical monsoon climate in China (Li et al., 2013). They reported that the no-tillage system reduced the total seasonal CO_2 emissions by 19–33%. In general, emission rates from paddies with early maturing rice crops ranged between 0.63–9.24 μmol CO_2 m^{-2} s^{-1} and −157.5–7813 μmol CH_4 m^{-2} h^{-1}. GHG-emission rates from rice paddy fields in Eastern China varied between 1.78–1.86 μmol CO_2 m^{-2} s^{-1}, 7.1–7.8 μmol CH_4 m^{-2} h^{-1} and −0.84–0.74 μmol N_2O m^{-2} h^{-1}; emission rates were affected by the fertilizing scheme (Zhang et al., 2014).

Methane emission rates of 95 ± 63 μmol CH_4 m^{-2} h^{-1} were determined in floodplain areas of the Inner Mongolian steppe (Liu et al., 2009a). Higher values were measured from different types of wetlands located in a floodplain in central Ohio, USA, with mean emission rates between 92 and 300 μmol CH_4 m^{-2} h^{-1} (Waletzko and Mitsch, 2013). Summer N_2O-emission rates from waterlogged and from relatively dry tundra marsh sites in maritime Antarctica were −0.07 and 0.95 μmol N_2O m^{-2} h^{-1}, respectively (Zhu et al., 2014). Constructed wetlands (CW) for wastewater treatment also emit GHGs. Søvik et al. (2006) measured such emissions from different sites in Estonia, Finland and Norway. The average emission rates in summer were 4.34 μmol CO_2 m^{-2} s^{-1}, 1007 μmol CH_4 m^{-2} h^{-1}, and 135.4 μmol N_2O m^{-2} h^{-1}, and in winter 1.16 μmol CO_2 m^{-2} s^{-1}, 219 μmol CH_4 m^{-2} h^{-1}, and 6.8 μmol N_2O m^{-2} h^{-1}. Lower values were measured from two different types of CWs in southern Estonia with 1.27–1.41 μmol CO_2 m^{-2} s^{-1}, 0.01–22.1 μmol CH_4 m^{-2} h^{-1} and −0.01–2.1 μmol N_2O m^{-2} h^{-1} (Teiter and Mander, 2005).

A fen site in Finland (peat soil, average air temperature −1.1 °C) delivered CH_4-emission rates of up to 68 μmol CH_4 m^{-2} h^{-1} during the cool season and 347 μmol CH_4 m^{-2} h^{-1} during the warm season. Much lower annual CH_4-emission rates of 1.73 ± 2.10 μmol CH_4 m^{-2} h^{-1} were registered from moorland in the United Kingdom (8.1 °C average annual air temperature) in comparison with the Finnish site (122 ± 28.5 μmol CH_4 m^{-2} h^{-1}). The wetland site in Finland was a net sink for CO_2; its average NEE over two years was −0.06 μmol CO_2 m^{-2} s^{-1} (Skiba et al., 2009).

Peatlands are a type of wetland ecosystem. Their organic-rich soils are considered to be one of the important sources of the GHGs CO_2 and N_2O, mainly when used for agriculture after drainage. High emission rates occur even 20–30 years after agricultural activities have ceased. In contrast to other land-cover types, these peatlands may become a CH_4 sink. Insignificant low emission rates of CH_4 were noticed in short periods from drained peatland sites. Annually, they acted as a sink for atmospheric CH_4. Soil properties affect soil potential to take up atmospheric CH_4. Soils with higher organic matter content have higher CH_4 uptake (1.49 μmol CH_4 m^{-2} h^{-1}) than those with lower organic content (1.0 μmol CH_4 m^{-2} h^{-1}). In addition, soils without vegetation cover have the lowest uptake rate: 0.52 μmol CH_4 m^{-2} h^{-1} (Maljanen et al., 2004). A mangrove ecosystem under dry tropical climate in northern Colombia exhibited emission rates between 0.8–26.8 μmol N_2O m^{-2} h^{-1} and 0–1973 μmol CH_4 m^{-2} h^{-1} (Konnerup et al., 2014). The authors also found a negative correlation between salinity and N_2O emission.

On a global scale, wetlands are still being drained for different purposes. About 50% of global wetland areas have been converted to other land-use forms (Verhoeven and Setter, 2010). Using dried-up peat soils for agriculture enhances CO_2 and N_2O emissions because of fertilization and tillage (Maljanen et al., 2007). Soil under barley crops at drained organic sites in Finland emitted 7.52 μmol CO_2 m^{-2} s^{-1}, while nearby bare soils released 3.66 μmol CO_2 m^{-2} s^{-1}. The rate of N_2O emission was 4.26 μmol N_2O m^{-2} h^{-1} from soil

Table 6
Soil emission rates (μmol CO_2 m^{-2} s^{-1}, μmol CH_4 and N_2O m^{-2} h^{-1}) from natural wetlands, including river stretches and reservoirs under temperate, mediterranean and (sub)tropical climates. Rates are given without standard deviations. Ranges have been rounded for clarity (see original literature and text for more details).

	Temperate		Mediterranean		(Sub)Tropical	
Species	Rate	Type	Rate	Type	Rate	Type
CO_2	1.2–1.9	Small lakes[23]	5.47	Wetland[12]	2.0	Wetland[4]
	2.55	Wetland[27]	1.00	Constructed wetland[3]	1.78–1.86	Paddy[36]
	1.27–1.41	Constr. wetland[30]				
	3.16–3.79	Rewetted fen[11]			1.8–1.9	Marsh, swamp[22]
	1.68; 5.45	Low mountain bog[32]			0.63–9.24	Early rice paddy[17]
	4.08; 8.14	Low mountain fen[32]			1.72	Mangrove[18]
	4.75	Subalpine fen[7]			2.37	Constructed wetland[35]
	4.99	Marsh[34]				
Range CO_2	1.2–8.1		1.0–5.5		1.8–9.2	
CH_4	9.6	Free flowing river[20]	278	Wetland[10]	508	Wetland[6]
	821	Dammed river[21]	437	Restored peatland[13]	2108	Wetland[4]
	338	Open swamp[15]			534	Paddy[6]
	3.6	Swampy gr.[8]			7.1–7.8	Paddy[36]
	201	Edge of pond[9]			1.3–15.6	Wetland[24]
	0.1–6	Swamp[26]			2360; 1860	Marsh; swamp[22]
	3.75–14.1	Wet grassland[5]				
	<68; 347; 122	Peatland in winter; in summer; avg.[29]			27.5; 12.6; 65.2	Rape culture; +manure; +inorg. fertilizer[25]
	1.73	Moorland England[29]				
	27.8	Marsh[34]				
	−1.84; 85; 326; 391	Wetland[27,10,31,6]			105; 381; 514	Wetland[6,28]
	228; 26.7	Low mountain bog[32]				
	2193; 0	Low mountain fen[32]			0–1973	Mangrove[14]
	0.01–22.1	Constructed wetland[30]			−157.5 to 7813	Early rice paddy[17]
	95	Floodplain area[19]				
	92–300	Floodplain areas[33]				
global avg.	39, 112, 208, 219, 260, 659, 807	Bogs, shallow lakes, fens, swamps, floodplains, marshes, paddy fields[1]				
Range CH_4	−1.8 to 28 (max. 810)		278–437		<1–7800	
N_2O	0.29	Natural wetlands[16]	4.73	Wetland[12]	−0.84 to 0.74	Paddy[36]
	−0.01 to 2.1	Constructed wetland[30]			−7.44 to 23.8	Wetland[22]
	−1.6 to 4.4	Wetland[2]			0.38–1.52	Wetland[24]
	0.85	Wetland[27]			0.8–26.8	Mangrove[14]
	0.90	Marsh[34]				
	0.04	Peatland[29]				
Range N_2O	−1.6 to 4.4		4.7		−7 to27	

[1] Aselmann and Crutzen (1989) compilation.
[2] Audet et al. (2014) Denmark.
[3] Barbera et al. (2015) Southern Brazil.
[4] Bass et al. (2014) northern Australia.
[5] Boeckx and van Cleemput (1997) wet grassland, Belgium.
[6] Cao et al. (1998) compilation of temperate and tropical climate zone.
[7] Chimner and Cooper (2003) Colorado, USA.
[8] Fiedler and Sommer (2000) swampy grassland, Germany.
[9] Fiedler and Sommer (2000) colluvial edge of a pond, Germany.
[10] Grunwald et al. (2012).
[11] Günther et al. (2014) Germany, few years after rewetting.
[12] Kluber et al. (2014) Mid-Atlantic region, USA.
[13] Knox et al. (2015) Sacramento, USA.
[14] Konnerup et al. (2014) Colombia.
[15] Kormann et al. (2001) open swamp, Germany.
[16] Leppelt et al. (2014) European wetlands.
[17] Li et al., 2013China.
[18] Liang et al. (2013) Jiulong River estuary, China.
[19] Liu et al., 2009a Inner Mongolia China.
[20] Maeck et al. (2014) Saar river, free flow, Germany.
[21] Maeck et al. (2014) Saar river, dammed, Germany.
[22] Marín-Muñiz et al. (2015) marsh, swamp, Mexico.
[23] Marwinski and Neubert (2015) small ponds, Saxony, Germany.
[24] Musenze et al. (2015) Moreton Bay, Australia.
[25] Nyamadzawu et al. (2014) Zimbabwe.
[26] Priemé (1994) wetland, Denmark.
[27] Schaufler et al. (2010) Europe.
[28] Sjögersten et al. (2015) tropical wetlands with mineral soil (lower) and organic soil (higher).
[29] Skiba et al. (2009) Finland.
[30] Teiter and Mander (2005) Estonia.
[31] Turetsky et al. (2014).
[32] Urbanová et al. (2012) intact and drained wetland in Bohemian forest (Czech Republic).
[33] Waletzko and Mitsch (2013), USA.
[34] Wu et al. (2013) China.
[35] Yan et al. (2012) China.
[36] Zhang et al. (2014) eastern China.

Table 7
Spatial macro dimensions from spot to global scale.

Unit	Sub-unit	Dimension (area)
Global	Supra-continental	50–500 Mio km^2
Continental	Continental	5–50 Mio km^2
	Sub-continental	0.5–5 Mio km^2
Regional	Large regional	50,000–500,000 km^2
	Medium regional	5000–50,000 km^2
	Small regional	500–5000 km^2
Local	Large local	50–500 km^2
	Medium local	5–50 km^2
	Small local	0.5–5 km^2
Site	Large site	0.05–0.5 km^2 (50,000–500,000 m^2)
	Medium site	0.005–0.05 km^2 (5000–50,000 m^2)
	Small site	0.0005–0.005 km^2 (500–5000 m^2)
Spot	Large spot	0.00005–0.0005 km^2 (50–500 m^2)
	Medium spot	0.000005–0.00005 km^2 (5–50 m^2)
	Small spot	0.0000005–0.000005 km^2 (0.5–5 m^2)

Since there seems not to be a unified definition for spatial dimensions in Geosciences, Table 7 suggests a related definition, based on practical experience. It is expanded from Reimann et al. (2009).

under barley cultivation and the highest emission emerged from the bare soils with 147.8 μmol N_2O m^{-2} h^{-1} during the growing season (Maljanen et al., 2004). Soil respiration rates from barley fields ranged between 2.5–8.2 μmol CO_2 m^{-2} s^{-1} on mineral and peaty soils (Lohila et al., 2003).

From the compiled data, the relatively highest average flux rate of wetlands can be calculated at 629.2 mg CO_2e m^{-2} h^{-1}. Given their low total global land area of 2.7% and compared to flux rates for other land-cover types, these emissions become almost negligible for the planetary surface (Figs. 2 and 3).

3. Methodologies to quantify GHG emissions from soils

We introduce the most important methodologies for determining trace gas emissions from soils and discuss their individual strengths and weaknesses. In general, trace gas emissions from soils are being directly measured in both field and laboratory (chamber techniques and micrometeorological methods), obtained through space and airborne measurements, and calculated with empirical and process-oriented models (Fig. 4). Given the relevance of reliable soil respiration data for land-use management, chamber systems receive some more attention in this work, since their output allows for more differentiated localized information.

3.1. Chamber systems

Flux chamber-based analysis is widely used in soil emission studies of CO_2, CH_4, N_2O and NO (Heinemeyer and McNamara, 2011; Kitzler et al., 2006; Oertel et al., 2012; Pumpanen et al., 2004; Šimek et al., 2014). A box or cylinder (diameter XX cm^2 to X m^2 footprint) is placed onto the soil surface so that the section of its base is open to the ground (Fig. 4a). Emitted gases accumulate in its chamber headspace. The change of mixing ratio can be analyzed with various gas sensors, e.g., gas chromatography (CO_2, N_2O, CH_4), IR-spectrometry including NDIR and FID with and without pumps (CO_2, CO, CH_4), chemiluminescence (NO_x), Cavity-Ring-Down spectrometry (CO_2, CO, CH_4, N_2O, H_2S) or photoacoustics (CO_2, CO, CH_4, NO and N_2O). Problems of too high headspace with inhomogeneous gas concentrations inside the chamber can be prevented by decreasing the chamber height (Rochette, 2011) and by lower detection limits (Davidson et al., 2002). Results deliver data for the spot scale, being measured at the small spot dimension (Table 7). However, to cover site-dimension fluxes representatively, chamber systems need to be easily and rapidly moved to measure multiple predetermined spots (Oertel et al., 2015a,b).

Chamber systems can also be used to analyze isotopic ratios of C and O species online in combination with a quantum cascade laser-based spectrometer (Kammer et al., 2011). Such methods can be applied to quantify CH_4 oxidation, since CH_4-oxidizing bacteria use ^{12}C methane (Börjesson et al., 2007). Delta ^{14}C of the emitted CO_2 serves to determine the age of the originating carbon source (Gorczyca, 2013). C-isotopes can also be used to distinguish between plant root and microbial respiration (Pausch et al., 2013).

Chamber systems can be divided into closed and open chambers, with closed chambers being subdivided into closed static and closed dynamic ones (Kutzbach et al., 2007; Rochette et al., 1997). Closed dynamic chambers (see Section 3.1.1) may also be referred to as non-steady state flow-through chambers. There is still no standardized chamber system, which may inhibit direct comparison of data sets from different research groups (Pumpanen et al., 2004).

All chamber systems should be equipped with auxiliary sensors to register soil emission-influencing parameters. Sensors for air temperature, pressure and relative humidity should be installed inside and outside the chamber to log ambient conditions and to register the differences from within the chamber. A photosynthetic active radiation (PAR) sensor needs to be additionally installed outside of the chamber, which is indispensable for net ecosystem exchange (NEE) measurements (Burrows et al., 2005). All chambers have to be installed on a collar (of steel or inexpensive PVC) in order to prevent gas leakage from the chamber to the atmosphere. To minimize the influence of the collar on the soil structure and plant roots, the collar should, if possible, be embedded to a depth of a few centimeters (Heinemeyer et al., 2011). Since collars influence the soil profile, they also influence the flux measurements and need to be installed at least 24 h prior to the first measurement (Bahn et al., 2012). Some chambers may work better without a collar, yet this is not recommended for use on forest soils (Pumpanen et al., 2004). To measure NEE, transparent chambers have to be used (Wang et al., 2013a), while opaque ones serve the determination of ecosystem respiration and of other gases (Sanz-Cobena et al., 2014b). The opaque material also insulates against temperature increases inside the chamber that would lead to pressure changes and influence soil emissions (Xu et al., 2006). A rapid change (few minutes) between transparent and opaque mode is possible with some chamber systems (Oertel et al., 2015b). In such a configuration, NEE and ecosystem respiration can be measured with one system.

If gas production in different soil depth is of interest, gas concentration profiles can be assessed (Chirinda et al., 2014). Samples taken with syringes from specific soil depths may be analyzed in the laboratory by gas chromatography (Petersen et al., 2011). Gas sensors may also be directly installed at specific soil depth for automatic and continuous measurements (Tang et al., 2003).

3.1.1. Closed chambers

Closed static chambers are most common for the analysis of CH_4 and N_2O fluxes (Pihlatie et al., 2013). These chambers also offer an absorption method for CO_2 analysis, where CO_2 is trapped in an alkaline solution. This permits fluxes to be measured over longer times and replicates at several measurement points without the need for additional sensors (Yim et al., 2002). Yet, this method systematically underestimates CO_2 fluxes (Nay et al., 1994) and is rarely used.

In closed dynamic chamber systems, gases accumulating in the chamber are analyzed either externally and pumped back into the chamber (Rochette et al., 1997; Heinemeyer and McNamara, 2011) or are being analyzed inside the chamber with a compact NDIR-sensor that continuously monitors the atmospheric CO_2 concentrations (Oertel et al., 2015a). N_2O and CH_4 emissions can be

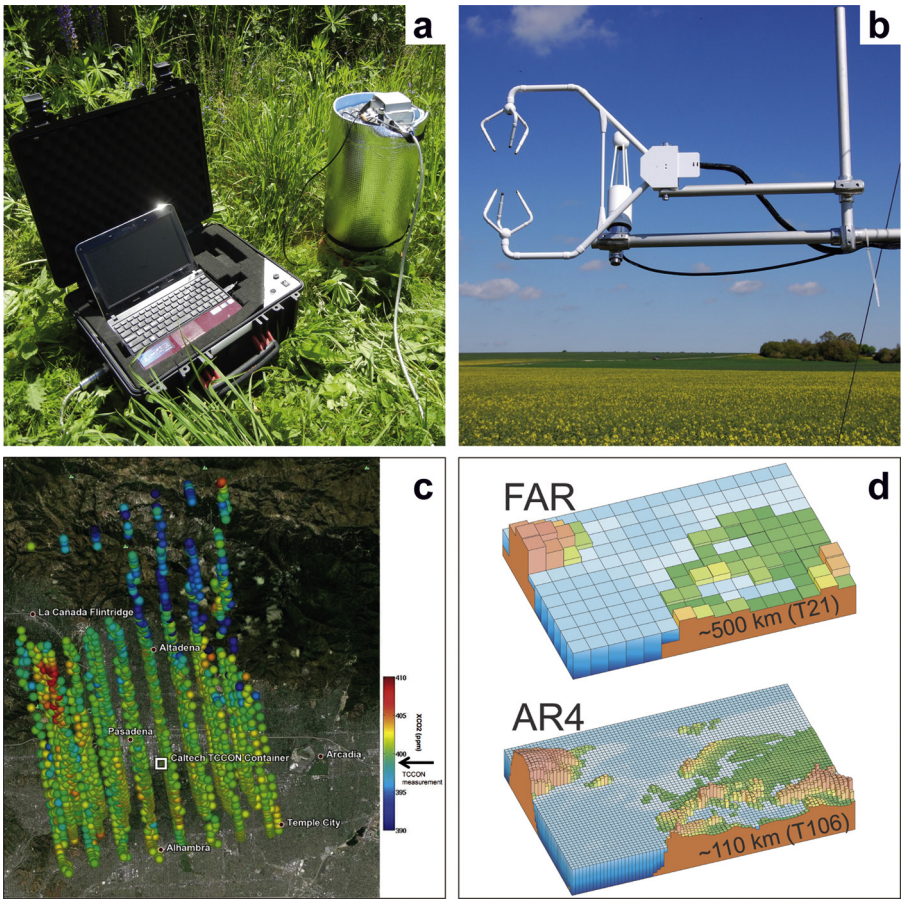

Fig. 4. Overview methodologies. (a) Closed chamber system; (b) eddy covariance; (c) remote sensing; (d) numerical modelling.
Image sources: (b) eddy covariance: http://www120.uni-hohenheim.de/typo3temp/pics/76026f486a.jpg; (c) remote sensing: Orbiting Carbon Observatory-2 (OCO-2) of CO_2 levels over Pasadena CA on September 5, 2014. Each coloured dot depicts a single measurement of CO_2 made during a 5-min satellite flight over the area. Over the heart of Pasadena, a level of 402 ppm_v of CO_2 was recorded. Image.
Source: NASA/JPL-Caltech, http://www.jpl.nasa.gov/spaceimages; (d) numerical modelling: IPCC (2007), modified.

analyzed as well with closed dynamic chambers (Cowan et al., 2015; Yu et al., 2013), but are rarely used compared to static chambers.

Closed static and dynamic chambers are available as both manual and automatic versions. They differ in how ambient conditions inside the chamber are restored. Chambers need to be purged from the soil gases after each measurement. With manual systems the operator needs to lift the chamber off the ground. Automatic chambers may have a moveable lid, comparable to the hinged lid of a traditional beer mug, which enables gas exchange (Edwards and Riggs, 2003; Pape et al., 2009). Alternatively, for some models, the entire chamber is lifted (LI-COR, 2012). A vent needs to be included to prevent pressure gradients during chamber deployment particularly with manual systems (Bain et al., 2005). An additional fan in all chamber types intermixes the inside air to maintain a constant and homogeneous level of increasing emitted gases (Christiansen et al., 2011). Manual chambers are able to cover spatial variability since they are portable and easily repositioned or relocated. No mechanical parts are needed, however, the main drawback is that they have to be permanently operated by hand. To achieve a similar performance with automatic chambers, many more systems are needed. Automatic systems can be used for continuous monitoring and do not have to be assisted, however, they involve higher material costs.

Accumulation times for the gas measurements need to be adapted to the emission rates of the different gases. CO_2 fluxes require the shortest accumulation times (2–4 min; Caprez et al., 2012; Correia et al., 2012; Vesterdal et al., 2012). This requires fast IR-spectrometers that analyze CO_2 fluxes in less than 10 s. During a measurement, the CO_2-mixing ratio may change from several tens to hundreds of ppm_v. Methane measurements take about 60–90 min with sampling intervals of about 20 min, using a gas chromatograph with a manual chamber (Fiedler et al., 2005; Liu et al., 2009a; Sanz-Cobena et al., 2014b). The accumulation time for N_2O measurements lies between 30 and 90 min (Hayakawa et al., 2009; Lamers et al., 2007; Yao et al., 2009). On average, 5–30 min are needed to accumulate NO (Liu et al., 2009b; Oertel et al., 2012; Yan et al., 2013). Cavity ring-down spectroscopy (CRDS) exists for monitoring systems, where CO_2, CH_4, and N_2O are analyzed from one sample, similar to gas chromatography. Yet, CRDS is faster, measuring every 8 s compared to 3–4 min by gas chromatography (GC), achieves higher precision and does not need additional equipment such as gas generators or gas bottles, thus providing better portability (Fleck et al., 2013; Rella et al., 2013). However, high acquisition costs are involved.

3.1.2. Open chambers

A third type of chamber system is the open dynamic chamber. Its two openings draw in ambient air and generate a continuous gas flow (Kutsch et al., 2009). Gas concentrations are analyzed at the air inlet and outlet of the chamber. The gas flux is calculated by the difference of the concentrations at both ends. Consequently, there are no accumulation times needed, since the flux is analyzed continuously. Continuous measurement systems do not need mechanical

parts. Problems due to high headspace mixing ratios do not occur (Balogh et al., 2007). Open chambers are suitable for dry and hot conditions in summer with low gas exchange rates. Closed chamber systems require longer accumulations times under such conditions, leading to temperature and pressure gradients (Balogh et al., 2007). Obviously, open dynamic chambers are technically more sophisticated and more expensive as compared to closed systems. For this reason, economically priced closed dynamic chambers are still the most common systems (Pumpanen et al., 2004).

3.1.3. Data evaluation

Soil flux for all gases can be calculated by linear and non-linear (exponential) regression, using the slope of the concentration change inside the chamber headspace (Christiansen et al., 2011). The linear model is easier to handle and works best for short chamber closure times, ideal for CO_2 (Forbrich et al., 2010). Therefore, it is widely used and least biased for curves with a convex-upward shape (Venterea et al., 2012). These authors found that linear regression is more sensitive to relative flux changes and is useful for studies with changing experimental parameters. The lowest detection limits for flux calculations are gathered with linear regression (Parkin et al., 2012). Yet, soil fluxes are significantly underestimated in contrast to the exponential model (Kutzbach et al., 2007), especially for curves with a convex-downward shape (Venterea et al., 2012). Burrows et al. (2005) explain this with increasing plant stress during measurements. The exponential model is suitable for longer closure times with few data points, e.g., when using a gas chromatograph (Forbrich et al., 2010). We recommend testing the best fit of linear and non-linear models for the measurements of net ecosystem exchange (NEE) during the day, where convex-downward curvatures are common. Evaluation of our own data, calculated with the linear model, showed that NEE during daytime can be more than 50% lower than the flux rate calculated with a suitable non-linear model—even for short measurement periods of five minutes. Nighttime measurements or ecosystem respiration measurements showed only small deviations of less than 10%. Sometimes values calculated with a non-linear model, which should be chosen when calculating fluxes of CH_4 and N_2O, delivered lower values than values calculated with a linear model.

3.2. Micrometeorological methods

The eddy covariance method is a direct micrometeorological approach. It uses vertical turbulences to analyze the turbulent heat and gas exchange between soil surface and atmosphere (Launiainen et al., 2005). A 3-D ultrasonic anemometer and a gas analyzer attached to a tower or mast of at least a 2-m height are needed for this method (Myklebust et al., 2008) (Fig. 4b). The most commonly analyzed gases are CO_2, CH_4 and N_2O (Nicolini et al., 2013; Wang et al., 2013a), yet substances like carbonyl sulphides (Asaf et al., 2013) or volatile organic compounds can be determined too (Ruuskanen et al., 2011). Measurements may run continuously and incorporate areas of up to several square kilometers (Myklebust et al., 2008), the medium local scale (Table 7). Eddy covariance integrates plants and trees, and thus completely covers soil, biosphere and atmosphere to determine NEE. The method does not work properly if very low near-ground turbulent mixing occurs. This leads to an underestimation of fluxes (Papale et al., 2006). This also applies if the system is installed within a forest (Kutsch et al., 2009). It is recommendable to perform measurements on levelled ground, above or within low-density vegetation (Baldocchi, 2003; Baldocchi et al., 2001). Data post-processing is complex and measurement gap filling is important to calculate fluxes (Du et al., 2013).

The open-path Fourier Transform Infrared spectroscopy (FTIR) is another near-ground micrometeorological method, using an instrument that can be mounted on a tower or a pole. A radiation source emits the entire IR-spectrum simultaneously (Griffith et al., 2012). The gases modify the response signals that are received by a telescope along the pathway. Such pathways may have standard lengths of 100–500 m (Griffiths et al., 2009); Kelliher et al. (2002) measured CO_2 and N_2O on a 97-m-long path. Methane, H_2O and other gases as well as C-isotopes can also be analyzed by FTIR (Griffith et al., 2012).

3.3. Laboratory experiments

Laboratory approaches help when the influence of single parameters (e.g., soil temperature or nutrient availability) on soil emissions shall be assessed. Single parameters can be changed, while others are kept constant (Schaufler et al., 2010). Soils from different climate zones can be investigated under controlled temperature and humidity conditions (Schaufler et al., 2010). As an example, Gritsch et al. (2015) analyzed soil monoliths from nine stations across Europe, representing different land-use types. The authors could clearly show dependencies between CO_2 emissions with soil temperature and moisture. Climate chambers that allow full control of temperature and humidity as well as light conditions are used for such experiments. Sieved and homogenized or undisturbed soil (cores) material may be used (small spot scale; Table 7). While undisturbed material does not negatively influence soil structure and microbial life (Petersen et al., 2013; Schaufler et al., 2010; van der Weerden et al., 2010; Yao et al., 2010), the heterogeneity among soil cores demands a larger sample size. Gritsch et al. (2015), for instance, took 33 undisturbed samples per site. Problems lie in destroying roots during sampling and in maintaining constant physical soil core conditions during transport. Influencing parameters can be observed better with homogenized soil material, which is a widespread approach (Aranibar et al., 2004; Feig et al., 2008; Laville et al., 2009; Oertel et al., 2011; Patino-Zuniga et al., 2009). Yet, soil structure is destroyed in the lab and before sieving soil material needs to be air-dried, which inadvertently influences microbial activity.

Lysimeters are an additional option to study soil emissions under controlled conditions in the field. Emissions can be analyzed jointly with the analysis of nutrient leaching (Velty et al., 2007; Zhou et al., 2013). Small field chamber systems can be used both in the laboratory and on lysimeters in the field, while some research groups use chambers, especially designed for laboratory use (Jäger et al., 2011; Schaufler et al., 2010; Yao et al., 2010).

3.4. Spaceborne measurements

Remote sensing from satellites may deliver information on GHG soil emissions in two different ways. One approach is to estimate tropospherical, near-surface CO_2 and CH_4 concentrations based on the measurement of the intensity of the reflected sunlight in small wavelength bands in the visible and short-wavelength IR portion of the spectrum (Fig. 4c). Earlier earth observation missions of the European Space Agency (ESA) like ERS-1 and ENVISAT carried low resolution scanning imaging absorption spectrometers for atmospheric cartography (ERS-GOME; ENVISAT-SCIAMACHY; Frankenberg et al., 2005) with a precision of 1–2% (Schneising et al., 2008). Simultaneously, the Japan Aerospace Exploration Agency (JAXA) operated the GOSAT system with a thermal and near-infrared sensor for carbon observations (TANSO), which is operational until present (Kuze et al., 2009). All previous instruments are limited in their applicability for flux estimates (Yoshida et al., 2011). The OCO-2 (Orbiting Carbon Observatory), a NASA satellite with a precision of 1–2 ppm_v for CO_2, can cover the variability of CO_2 and CH_4 sinks and sources with high spatial and temporal resolution (Boesch et al., 2011). It was successfully

launched on July 02, 2014, and is designed for a two-year lifetime (Greicius et al., 2014; NASA, 2014). The continuation of the global time series of CO_2 and CH_4 concentrations after the lifetime of GOSAT and OCO-2 is being planned with the Carbon Monitoring Satellite (CarbonSat). CarbonSat is a candidate mission for ESA's EarthExplorer 8 (EE8) satellite to be launched at the end of this decade. Unlike the previous missions, it is designed to map natural and anthropogenic GHG sources and sinks at high spatial resolution ($2 \times 2\ km^2$) and spatial coverage in order to localize strong emission sources (Buchwitz et al., 2013).

Mapping the spatial distribution and change of land cover types that represent sources or sinks for CO_2 and CH_4 is the alternative to direct estimations of GHG concentrations from remote sensing systems. Coarse to medium resolution remote sensing data deliver a globally consistent and objective source of information for a spatially explicit mapping of the distribution of potential C stocks in terms of land cover type maps (regional to global scale with local resolution; Table 7). However, there is still considerable uncertainty in the spatial agreement of the area and distribution of the relevant land cover types (e.g., grassland, forests, barren land, cropland, wetland) and hence of the globally stored C (Herold et al., 2008; Pflugmacher et al., 2011). These uncertainties are attributed to a number of limitations that are determined by either the technical specification of the sensor (wavelength, spectral and spatial resolution) or the derived data products (e.g., land cover maps). Differences between land cover maps have important implications on modeling global emissions. Thus, the choice of a map might introduce a significant bias in any regional to global carbon balance model.

3.5. Airborne measurements

Airborne methods use direct sampling approaches to collect gases from transects, e.g., over different types of land use or from near-surface environments to higher tropospheric altitudes. Amelio et al. (2009), for instance, collected air samples on an ascending and descending flight path of an airplane (between 305 and 3600 m a.s.l.). These samples were stored in flasks and analyzed in the laboratory by gas chromatography for CO_2, CH_4 and N_2O. Yet, airborne measurements deliver data over a short time period only and with common spatial distances of about 10–100 km (Desjardins et al., 1997, 2010; Kustas et al., 2006; Pattey et al., 2007; medium local scale; Table 7).

3.6. Modeling soil emissions

Since measurements deliver only punctual data, modeling GHG emissions from soils is important to regionalize such measurements and to calculate global budgets. Next to empirical models (e.g., Freibauer and Kaltschmitt, 2003), process-based models that include general physical and chemical processes are used to process the obtained field data (Pattey et al., 2007). Such approaches may cover local to global scales (Table 7; Fig. 4d).

A widely used process-oriented simulation model is the DeNitrification-DeComposition Modell (DNDC), consisting of four sub-models (Li et al., 1992). It can model daily decomposition, nitrification, ammonia volatilization, CO_2 production (soil microbial and root respiration), N-uptake of plants and plant growth (Li et al., 1994). This model is often used to calculate GHG emissions from soils, especially from agricultural lands (Abdalla et al., 2011; Gu et al., 2014; Li et al., 2012). Meteorological and soil parameters (e.g., soil texture, pH-value, bulk density, organic C) as well as vegetation type and management practices (tillage system, fertilizer application, grain yield) serve as important input data (Abdalla et al., 2009b). GHG fluxes from forest ecosystems can be simulated with the Forest-DNDC model (Abdalla et al., 2013). Falloon and Smith (2012) listed further models for the simulation of C and N trace emissions from soils.

Butterbach-Bahl et al. (2004) compared model output with on-site measurements for agricultural and forest ecosystems. While they conclude that the modelling approach delivers higher quality estimates, results may deviate by one order of magnitude. However, the average error for soil emissions is generally much lower (Butterbach-Bahl et al., 2004; Hastings et al., 2010) and modeling data may often be used with some confidence.

3.7. Method comparison

To compare the different methods with respect to applicability (spatial variability, observable area, continuous monitoring, analyzing processes), the aspects of accuracy and precision (bias, e.g., influence on soil structure) and of costs and workload involved need to be taken into account. There is no single best technique out there. Ideally, investigations would allow for multiple method approaches such as a combination of chamber systems with eddy covariance and remote sensing (Liang et al., 2004; Myklebust et al., 2008).

3.7.1. Applicability

3.7.1.1. Spatial variability, observable area. Chamber measurements are an ideal tool if emissions from distinct sites are to be analyzed. The spatial footprint of an individual chamber measurement generally covers less than 1 m^2 (spot size; Table 7). Different agricultural management practices on soil or plants can be analyzed easily with chamber systems as well. To observe larger areas of up to 10^4 m^2 (Wang et al., 2013a), the number of chambers has to be increased or chambers have to be moved consecutively to cover a larger area (site size, Table 7). When regionalizing chamber flux measurement results, heterogeneity of soils and emission values on small spatial scales have to be kept in mind (e.g., Stoyan et al., 2000). Therefore, the use of representative sampling sites is fundamental. An area of some hectares can be observed with the eddy covariance method (Baldocchi, 2003). The open-path FTIR covers an area of 100–27,000 m^2 (large spot to medium site size; Table 7). The spatial extent is unlimited with modelling and remote sensing approaches. Since modelling or remote sensing data can differ from on-site measurements, validation using ground-based measurements (e.g., chamber measurements) is important, as well as area wide soil property and meteorological data.

3.7.1.2. Continuous monitoring. Chamber systems are suitable for continuous monitoring, except when winter conditions yield higher snow levels. Eddy covariance systems have well known problems at night and during periods of low turbulence (Papale et al., 2006). The open-path FTIR method can be applied during nighttime and in periods without turbulence (Kelliher et al., 2002). Remote sensing and airborne data depend on the orbit of the satellite or the flight frequency and track of the airplane, yet the measuring network is rather sparse over oceans and the tropics. Consequently, more remote sensing data are needed to fill this gap, although it is still challenging to distinguish between sinks and sources (Hungershoefer et al., 2010).

3.7.1.3. Resolution of different processes. To distinguish, e.g., between soil respiration, net ecosystem exchange or photosynthesis, chamber systems have to be used, since this is impossible with eddy covariance (Pumpanen et al., 2012). Nevertheless, entire ecosystem fluxes, e.g., for CO_2, cannot be measured since bigger plants and trees cannot be included inside chamber systems. Here, eddy covariance, remote sensing or modelling is required.

3.7.2. Accuracy and precision

3.7.2.1. Introduction of bias. Chamber systems are invasive, thus influencing soil emissions (use of collars and the chamber itself). Changes of temperature, pressure and humidity inside the chamber may also lead to biased soil emission values (Rochette and Eriksen-Hamel, 2008; Suleau et al., 2009). Short closure duration of the chamber can minimize these artefacts. Own data showed that the temperature inside the non-transparent chamber increased up to 3 K at non-forest sites during summer. Using the transparent mode, temperature differences can be up to 10 K. In forest ecosystems and under colder conditions, temperature differences are below 2 K. Air humidity measurements differed on the agricultural and forest sites by about 10–20% during summer and <10% on cold, humid days. Using a vent can minimize pressure differences, while it increases the sensitivity to wind forcing (Bain et al., 2005).

Field measurements and modeling for the same plots deliver different results. Pumpanen et al. (2004) tested different chambers under constant conditions with a calibration system. Closed chambers tended to underestimate CO_2-fluxes by 10%. Closed static chambers, sampled with syringes, underestimated fluxes by up to 35% and decreased errors with shorter accumulation times. Small overestimations of the fluxes between 2 and 4% were found for open dynamic chamber systems (Pumpanen et al., 2004). Myklebust et al. (2008) compared the NEE measured with a dynamic chamber system and with eddy covariance and found no relevant deviations during the non-growing season and wind speeds above 0.2 m s^{-1}. Wang et al. (2013a) found 4% higher fluxes for NEE on an agricultural site measured with a static chamber system compared to values from eddy covariance. Especially on agricultural sites, comparisons of measurements between eddy covariance and chamber systems are still needed (Wang et al., 2013a).

Compared to N_2O field measurements with static chambers, Abdalla et al. (2009b) found an underestimation with a deviation of up to 360% for the denitrification-decomposition (DNDC) model. Pihlatie et al. (2013) found an underestimation of about 33% for CH_4 fluxes with static closed chambers. Abdalla et al. (2013) compared eddy covariance data with modeled NEE from the forest-DNDC model and found a good agreement for seasonal and annual values. Field measurements better detect peak values of NEE (Abdalla et al., 2013). Butterbach-Bahl et al. (2009) tested both DNDC models for simulating NO soil emissions from agricultural and forest soils and found flux underestimations of up to 60%.

3.7.3. Costs and workload

Chamber systems are easy to handle and can be operated by one person. These devices are easy to transport and due to this, numerous locations can be analyzed within a short period. By contrast, portability of eddy covariance systems (mounted on poles) is limited.

Chamber systems have the lowest material costs compared to the other methods (Wang et al., 2013a). Material costs for manual and automatic chamber systems that perform online analysis of CO_2 range from EUR 6000–8000 (USD 6700–9000). Commercially available solutions cost about EUR 20,000 € (USD 22,500), e.g., from LiCOR. A cavity-ring-down-spectrometer costs about EUR 40,000 (USD 45,000) and eddy-flux systems ~EUR 30,000 (USD 33,700). Open-path FTIR-systems range around EUR 60,000 (USD 67,500).

The workload is high, especially with manual chambers, since they have to be supervised continually. Data processing is highest for remote sensing and eddy covariance and data gap filling for non-turbulent periods is important (Wang et al., 2013b). Compared to all other solutions, chamber systems are easy to install by one person without any further equipment.

4. Synthesis of major findings and conclusions

While the GHG effect has been described a long while ago and Svante Arrhenius published first calculations in 1896 already (Arrhenius, 1896), it was not until the 1970s that the contribution of soils in the global GHG budget was recognized and moved into focus (Hutchinson and Mosier, 1979; Schmidt et al., 1988). It began with individual site (spot) investigations (Table 7). Soon the entire suite of methodological approaches delivered an increasing amount of data and interpretations. It became progressively obvious that the source and sink characteristics of soils with all influencing parameters (as discussed above) could not be neglected in comprehensive global and regional GHG budget considerations (Houghton et al., 1990; IPCC, 2013). This in turn triggered international agreement that individual nations should report to the United Nations about their GHG inventories, including net soil emissions (e.g., Rösemann et al., 2011). Such assessments are mostly based on the IPCC guidelines for national GHG inventories (2006) and their constant advancement. The resulting data deliver the base for global assessments and related mitigation and abatement activities. Yet, we must acknowledge the fact that remote sensing data still do not provide the necessary accuracy and precision and that ground-based data are strongly biased with respect to temperate climate regions and the northern hemisphere at large (Fig. 2). It would be highly desirable to have unified standards and protocols to ensure quality control and data comparability. This includes the necessity for a definition of required meta-data, such as precise information on location, soil and air temperatures, time frame and its relation to annual average meteorological and possibly land-use conditions, instrumentation and its calibration, etc.

Any up- or downscaling of measured data yields a multitude of potential errors and biases (also independent of GHG emissions from soils) that need to be addressed—and overcome (see Reimann et al., 2009, 2010, for an analogous example). Site data acquisition with chamber systems or micrometeorological methods can deliver reliable and highly precise data, with the latter capable to include small local dimensions. However, attempts to upscale their results to a regional level will likely end in rather improbable results, unless a dense station network can be maintained for an extended period of time, representing land cover and land-use types of that region. So far, no such clusters exist for larger spatial scales (e.g., continental) and long-term monitoring studies are very scarce. Nevertheless, high quality site-dependent data are indispensable to allow for proper model validation and to safeguard model output quality.

Model calculations, as the general base for regional and sub-continental inventories (e.g., in the European Union), are increasingly well able to deliver robust results. Nevertheless, it would be premature to fully trust model outputs as representing reality. The output rather serves obtaining an overview and triggering critical questions, which will eventually further improve input database and algorithms. With better satellite data (higher resolution, enhanced representation of small-scale variation), we can expect input data to improve. Rösemann et al. (2011) give an in-depth evaluation of errors and biases in their inventory for Germany.

For global assessments, a combination of site and locality-related data, regional data acquisition/reporting and satellite data is used—again through the filter of appropriate models. All real and possible errors discussed above influence the obtained results. It is no major surprise to see differences of at least a factor of two, and often more, when comparing results from different groups or sources. Even differences in the accurate assessment of land-use and land-cover types (e.g., Ramankutty et al., 2008 versus FAO, 2014, 2015) still lead to noteworthy uncertainties (Fig. 3). Further ambiguity arises, given the rather fast conversion of forestland and other biomes into croplands, especially in (sub)tropical areas with

a rate of 48,000 km^2 per year (1999–2008; Phalan et al., 2013). Therefore, any current assessment can only deliver a momentary outlook onto this dynamic system. Given the as yet non-satisfactory results from modeling and remote sensing, this shortcoming of bias towards the temperate climate zone also needs to be overcome.

There is not one methodology that covers all spatial dimensions. Also, at this point there is no established standard methodology—neither for determining soil emission rates nor for related GHG-budget calculations. Both scientists and policy makers may not always be free from interests that counter the ideal to obtain and deliver unbiased and highly reliable data (Gillenwater et al., 2007; Winiwarter, 2007). This should be kept in mind, when dealing with this topic.

Various parameters enhance soil respiration. Changes in soil organic content and land-use management have a significant influence on respiration rates. Average CO_2-emission rates from wetlands exceed those from forestlands, grasslands, croplands and barren lands (Fig. 3). The highest rates were registered at sites located over peat and drained organic soils, followed by sites with managed grasslands.

Methane, the second most potent GHG, has a 25 times higher global-warming potential (per molecule) than CO_2 over a time horizon of 100 years calculated atmospheric residence time (IPCC, 2007). Wetlands are the major source of methane; available data cover a wide spectrum of wetland types under different climates (Table 6). The limited land cover of wetlands compensates for the high relative emissions, however. Thus, on a gobal scale, wetland emissions appear neither to alter global balances nor to act as a "hidden driver". Emission rates vary widely (0.1–6950 $\mu mol\ CH_4\ m^{-2}\ h^{-1}$), because of the diversity in wetland types and climates. At the same time, it needs to be emphasized that data are still rather scarce and certainly not representative on a larger scale. The highest CH_4 emissions were registered during warm seasons and in tropical environments. When wetlands dried up or were used for other purposes, emissions decreased or the soil even became a sink for atmospheric CH_4; the related uptake rate depended on its organic content. The average grassland CH_4-emission rate was below 3 $\mu mol\ m^{-2}\ h^{-1}$ and for croplands ranged from negative values to a very low rate of <1 $\mu mol\ m^{-2}\ h^{-1}$.

Nitrous oxide yields a 298 times higher global-warming potential per molecule than CO_2 (100 years time horizon; IPCC, 2007). The average N_2O-emission rate from grasslands was low with 10 $\mu mol\ N_2O\ m^{-2}\ h^{-1}$. This is higher, however, than the rate for croplands (<5 $\mu mol\ m^{-2}\ h^{-1}$) except from barren dry organic soils. Grazing and no-tillage practices on grasslands enhanced N_2O-emission rates. If we neglect wetlands for the global balance due to their very limited land cover, grasslands show the highest total CO_2-equivalent emission when compared with other land-cover types and are the most widespread type of land use (about 25% of land area worldwide). Thus, grasslands have the most promising opportunities for GHG mitigation. However, the N_2O-emission rate from wetlands ranged from negative values (sinks for N_2O) to more than 23 $\mu mol\ N_2O\ m^{-2}\ h^{-1}$ during warm seasons.

Through our literature review, limitations in the database became evident, particularly in respect to measured soil respiration in the field using chamber systems. Also, a deficiency in crop diversity information was obvious, with rather limited information about fruit trees (orchards) and vegetable production. The need to explore GHG emissions from fields under other crops appears obvious (Table 4). Most studies that focused on wetlands registered only CH_4, as compared to investigations on croplands and grasslands that yielded a more comprehensive GHG coverage.

Other land-cover types such as urban soils are currently radically underrepresented. Based on first and certainly insufficient data from our group, we assume that soil respiration from partly sealed areas in the temperate climate zone is comparable with wintertime emissions from agricultural fields (cropland). Pavements with plant or moss in the interstices can considerably exceed wintertime emissions, however if the non-sealed proportion of the pavement is extrapolated to the whole measuring spot. Emissions from urban meadows or flowerbeds are in the range of natural grasslands and agricultural fields in the respective season.

While our understanding is obviously increasing – and many new questions that now demand answers were no issue only a couple of years or decades ago – there still remain various sources of errors and uncertainties. These start with the calculated estimates for climate zones, biomes and land-cover areas. Depending on the initial questions, the tools used, and on the chosen scale for any specific analysis, errors up to roughly 10–20% may emerge. A larger relative error relates to GHG-emission data. Again, depending on the methodology used (and there are no established standards so far), results may vary considerably from both ends, when upscaling (e.g., from site measurements) as much as downscaling (e.g., from remote sensing observations). Related errors may easily reach and exceed 50%. Another source of errors relates to the rather uneven distribution of real data that still generates a bias between the northern and the southern hemispheres, their biomes and land-use patterns (Fig. 2). Thus, it may not be surprising that data published for comparable global calculations may deviate considerably from reality. If we use analogies from global data assessment, e.g., the World Soil Average (Koljonen, 1992), there is a strong bias resulting from an overrepresentation of studies with data from the northern hemisphere and temperate climate, as a recent study shows (Caritat et al., 2012). More experimental and monitoring data are needed with an area-representative distribution, even if it was just for a better validation of numerical models. Another aspect to be considered relates to homogenization and equalization of data. To this point, no unified norm or agreed-upon recommendation exists that defines quality standards to be applied (1) to site selection, (2) to methodological set-up of appropriate flux and concentration units, and (3) to data display. With many publications, metadata are missing or insufficient—making the set-up and boundary conditions for the measurements indiscernable and thereby obscuring the data.

Despite all criticism and shortcomings, the achievements and progress made on investigating GHG emissions from soils should not be neglected either. There is an increasing understanding of the very large role of soils as sinks and sources mainly for CO_2, CH_4 and N_2O, and of the various feedback processes involved. This understanding goes hand in hand with an appreciation of the massive power of intervention in the natural feedback processes through land-use (mis)management. While individual studies exist, their results are far from sufficient to be used in models and for policy counselling. The key drivers of soil GHG emissions are known and yet important geo databases such as truly representative maps for total carbon and nitrogen as well as total organic carbon are lacking or available for very restricted geographical areas only (e.g., Reimann et al., 2014). Such globally incomplete databases impede true and verified global representativeness.

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References

Abdalla, M., Jones, M., Smith, P., Williams, M., 2009a. Nitrous oxide fluxes and denitrification sensitivity to temperature in Irish pasture soils. Soil Use Manage. 25, 376–388.

Abdalla, M., Wattenbach, M., Smith, P., Ambus, P., Jones, M., Williams, M., 2009b. Application of the DNDC model to predict emissions of N_2O from Irish agriculture. Geoderma 151, 327–337.

Abdalla, M., Kumar, S., Jones, M., Burke, J., Williams, M., 2011. Testing DNDC model for simulating soil respiration and assessing the effects of climate change on the CO_2 gas flux from Irish agriculture. Glob. Planet. Change 78, 106–115.

Abdalla, M., Saunders, M., Hastings, A., Williams, M., Smith, P., Osborne, B., Lanigan, G., Jones, M.B., 2013. Simulating the impacts of land use in Northwest Europe on net ecosystem exchange (NEE): the role of arable ecosystems, grasslands and forest plantations in climate change mitigation. Sci. Total Environ. 465, 325–336.

Achtnich, C., Bak, F., Conrad, R., 1995. Competition for electron donors among nitrate reducers, ferric iron reducers, sulfate reducers, and methanogens in anoxic paddy soil. Biol. Fertil. Soil. 19, 65–72.

Almagro, M., López, J., Querejeta, J.I., Martínez-Mena, M., 2009. Temperature dependence of soil CO_2 efflux is strongly modulated by seasonal patterns of moisture availability in a Mediterranean ecosystem. Soil Biol. Biochem. 41, 594–605.

Ambus, P., Christensen, D., 1995. Spatial and seasonal nitrous oxide and methane fluxes in Danish forest-, grassland-, and agroecosystems. J. Environ. Qual. 25, 993–1001.

Amelio, M.T.S.D., Gatti, L.V., Miller, J.B., Tans, P., 2009. Regional N_2O fluxes in Amazonia derived from aircraft vertical profiles. Atmos. Chem. Phys. 9, 8785–8797.

Aranibar, J.N., Otter, L., Macko, S.A., Feral, C.J.W., Epstein, H.E., Dowty, P.R., Eckardt, F., Shugart, H.H., Swap, R.J., 2004. Nitrogen cycling in the soil-plant system along a precipitation gradient in the Kalahari sands. Glob. Change Biol. 10, 359–373.

Arnold, P.W., 1954. Losses of nitrous oxide from soil. J. Soil Sci. 5 (1), 116–128.

Arrhenius, S., 1896. On the influence of carbonic acid in the air upon the temperature of the ground. Philos. Mag. J. Sci. 41, 239–276.

Asaf, D., Rotenberg, E., Tatarinov, F., Dicken, U., Montzka, S.A., Yakir, D., 2013. Ecosystem photosynthesis inferred from measurements of carbonyl sulphide flux. Nat. Geosci. 6, 186–190.

Aselmann, I., Crutzen, P.J., 1989. Global distribution of natural freshwater wetlands and rice paddies, their net primary productivity, seasonality and possible methane emissions. J. Atmos. Chem. 8, 307–358.

Audet, J., Hoffmann, C.C., Andersen, P.M., Baattrup-Pedersen, A., Johansen, J.R., Larsen, S.E., Kjaergaard, C., Elsgaard, L., 2014. Nitrous oxide fluxes in undisturbed riparian wetlands located in agricultural catchments: emission, uptake and controlling factors. Soil Biol. Biochem. 68, 291–299.

Baggs, E.M., Stevenson, M., Pihlatie, M., Regar, A., Cook, H., Cadisch, G., 2003. Nitrous oxide emissions following application of residues and fertiliser under zero and conventional tillage. Plant Soil 254, 361–370.

Bahn, M., Reichstein, M., Davidson, E.A., Grünzweig, J., Jung, M., Carbone, M.S., Epron, D., Misson, L., Nouvellon, Y., Roupsard, O., Savage, K., Trumbore, S.E., Gimeno, C., Curiel Yuste, J., Tang, J., Vargas, R., Janssens, I.A., 2010. Soil respiration at mean annual temperature predicts annual total across vegetation types and biomes. Biogeoscience 7 (7), 2147–2157.

Bahn, M., Kutsch, W.L., Heinemeyer, A., Janssens, I.A., 2012. Appendix: Towards a Standardized Protocol for the Measurement of Soil CO_2 Efflux. Soil Carbon Dynamics. An Integrated Methodology. Cambridge Univ. Press, pp. 272–281.

Bain, W.G., Hutyra, L., Patterson, D.C., Bright, A.V., Daube, B.C., Munger, J.W., Wofsy, S.C., 2005. Wind-induced error in the measurement of soil respiration using closed dynamic chambers. Agric. For. Meteorol. 131, 225–232.

Baldocchi, D.D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Glob. Change Biol. 9, 479–492.

Baldocchi, D., Falge, E., Gu, L., Olson, R., Hollinger, D., Running, S., Anthoni, P., Bernhofer, C., Davis, K., Evans, R., Fuentes, J., Goldstein, A., Katul, G., Law, B., Lee, X., Malhi, Y., Meyers, T., Munger, W., Oechel, W., Paw, U.K.T., Pilegaard, K., Schmid, H.P., Valentini, R., Verma, S., Vesala, T., Wilson, K., Wofsy, S., 2001. FLUXNET: a new tool to study the temporal and spatial variability of ecosystem-scale carbon dioxide, water vapor, and energy flux densities. Bull. Am. Meteorol. Soc. 82, 2415–2434.

Balogh, J., Nagy, Z., Fóti, S., Pintér, K., Czóbel, S., Péli, E.R., Acosta, M., Marek, M.V., Csintalan, Z., Tuba, Z., 2007. Comparison of CO_2 and H_2O fluxes over grassland vegetation measured by the eddy-covariance technique and by open system chamber. Photosynthetica 45, 288–292.

Barbera, A.C., Borin, M., Cirelli, G.L., Toscano, A., Maucieri, C., 2015. Comparison of carbon balance in Mediterranean pilot constructed wetlands vegetated with different C4 plant species. Environ. Sci. Pollut. Res. 22 (4), 2372–2383.

Bass, A.M., O'Grady, D., Leblanc, M., Tweed, S., Nelson, P.N., Bird, M.I., 2014. Carbon dioxide and methane emissions from a wet-dry tropical floodplain in northern Australia. Wetlands 34 (3), 619–627.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.

Bayer, C., Gomes, J., Accordi Zanatta, J., Costa, F., Vieira, B., De Cássia Piccolo, M., Dieckow, J., Six, J., 2014. Soil nitrous oxide emissions as affected by long-term tillage, cropping systems and nitrogen fertilization in southern Brazil. Soil Tillage Res. 146B, 213–222.

Berglund, Ö., Berglund, K., Klemedtsson, L., 2010. A lysimeter study on the effect of temperature on CO_2 emission from cultivated peat soils. Geoderma 154, 211–218.

Bhattacharyya, P., Roy, K.S., Neogi, S., Dash, P.K., Nayak, A.K., Mohanty, S., Baig, M.J., Sarkar, R.K., Rao, K.S., 2013. Impact of elevated CO_2 and temperature on soil C and N dynamics in relation to CH_4 and N_2O emissions from tropical flooded rice (*Oryza sativa* L.). Sci. Total Environ. 461–462, 601–611.

Birch, H.F., 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10, 9–31.

Boeckx, P., van Cleemput, O., 1997. Methane emission from a freshwater wetland in Belgium. Soil Sci. Soc. Am. J. 61 (4), 1250–1256.

Börjesson, G., Samuelsson, J., Chanton, J., 2007. Methane oxidation in Swedish landfills quantified with the stable carbon isotope technique in combination with an optical method for emitted methane. Environ. Sci. Technol. 41, 6684–6690.

Boesch, H., Baker, D., Connor, B., Crisp, D., Miller, C., 2011. Global characterization of CO_2 column retrievals from shortwave-infrared satellite observations of the orbiting carbon observatory-2 mission. Remote Sens. 3, 270–304.

Bond-Lamberty, B., Thompson, A., 2010. A global database of soil respiration data. Biogeoscience 7, 1915–1926.

Borken, W., Brumme, R., 1997. Liming practice in temperate forest ecosystems and the effects on CO_2, N_2O and CH_4 fluxes. Soil Use Manage. 13, 251–257.

Borken, W., Matzner, E., 2009. Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. Glob. Change Biol. 15, 808–824.

Borken, W., Xu, X.J., Brumme, R., Lamersdorf, N., 1999. A climate change scenario for carbon dioxide and dissolved organic carbon fluxes from a temperate forest soil: drought and rewetting effects. Soil Sci. Soc. Am. J. 63, 1848–1855.

Borken, W., Xu, X.J., Davidson, E.A., Beese, A., 2002. Site and temporal variation of soil respiration in European beech, Norway spruce, and Scots pine forests. Glob. Change Biol. 8, 1205–1216.

Boussingault, J.B., Levy, B., 1853. Mémoire sur la composition de l'aire confine dans la terre vegetal. Ann. Chem. Phys., 5–50.

Bowden, R.D., Melillo, J.M., Steudler, P.A., Aber, J.D., 1991. Effects of nitrogen additions on annual nitrous oxide fluxes from temperate forest soils in the northeastern United States. J. Geophys. Res. 97, 9321–9328.

Bowden, R.D., Davidson, E., Savage, K., Arabia, C., Steudler, P., 2004. Chronic nitrogen additions reduce total soil respiration and microbial respiration in temperate forest soils at the Harvard Forest. Forest Ecol. Manage. 196 (1), 43–56.

Brümmer, C., Brüggemann, N., Butterbach-Bahl, K., Falk, U., Szarzynski, J., Vielhauer, K., Wassmann, R., Papen, H., 2008. Soil-atmosphere exchange of N_2O and NO in near-natural savanna and agricultural land in Burkina Faso (W. Africa). Ecosystem 11, 582–600.

Breecker, D.O., McFadden, L.D., Sharp, Z.D., Martinez, M., Litvak, M.E., 2012. Deep autotrophic soil respiration in shrubland and woodland ecosystems in central New Mexico. Ecosystem 15, 83–96.

Brooks, P.D., Schmidt, S.K., Williams, M.W., 1997. Winter production of CO_2 and N_2O from alpine tundra: environmental controls and relationship to inter-system C and N fluxes. Oecologia 110, 403–413.

Brumme, R., Borken, W., 1999. Site variation in methane oxidation as affected by atmospheric deposition and type of temperate forest ecosystem. Glob. Biogeochem. Cycles 13, 493–501.

Buchwitz, M., Reuter, M., Bovensmann, H., Pillai, D., Heymann, J., Schneising, O., Rozanov, V., Krings, T., Burrows, J.P., Boesch, H., Gerbig, C., Meijer, Y., Löscher, A., 2013. Carbon monitoring satellite (CarbonSat): assessment of atmospheric CO_2 and CH_4 retrieval errors by error parameterization. Atmos. Meas. Tech. 6, 3477–3500, http://dx.doi.org/10.5194/amt-6-3477-2013.

Burrows, E.H., Bubier, J.L., Mosedale, A., Cobb, G.W., Crill, P.M., 2005. Net ecosystem exchange of carbon dioxide in a temperate poor fen: a comparison of automated and manual chamber techniques. Biogeochemistry 76, 21–45.

Butterbach-Bahl, K., Gasche, R., Huber, C., Kreutzer, K., Papen, H., 1998. Impact of N-input by wet deposition on N-trace fluxes and CH_4-oxidation in spruce forest ecosystems of the temperate zone in Europe. Atmos. Environ. 32, 559–564.

Butterbach-Bahl, K., Kesik, M., Miehle, P., Papern, H., Li, C., 2004. Quantifying the regional source strength of N-trace gases across agricultural and forest ecosystems with process-based models. Plant Soil 260, 311–329.

Butterbach-Bahl, K., Kahl, M., Mykhayliv, L., Werner, C., Kiese, R., Li, C., 2009. A European-wide inventory of soil NO emissions using the biogeochemical models DNDC/Forest-DNDC. Atmos. Environ. 43 (7), 1392–1402.

Butterbach-Bahl, K., Kiese, R., Liu, C.Y., 2011. Measurement of biosphere-atmosphere exchange of CH_4 in terrestrial ecosystems. Meth. Enzymol. 495, 271–287.

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philos. Trans. R. Soc. B: Biol. Sci. 368, 20130122.

Cao, M., Gregson, K., Marshal, S., 1998. Global methane emission from wetlands and its sensitivity to climate change. Atmos. Environ. 32, 3293–3299.

Caprez, R., Niklaus, P.A., Körner, C., 2012. Forest soil respiration reflects plant productivity across a temperature gradient in the Alps. Oecologia 170, 1143–1154.
Caritat, P., de Reimann, C., NGSA Project Team, GEMAS Project Team, 2012. Comparing results from two continental geochemical surveys to world soil composition and deriving Predicted Empirical global Soil (PEGS-2) reference values. Earth Planet. Sci. Lett. 319–320, 269–287.
Castaldi, S., Fierro, A., 2005. Soil-atmosphere methane exchange in undisturbed and burned Mediterranean shrubland of southern Italy. Ecosystem 8, 182–190.
Castaldi, S., Ermice, A., Strumia, S., 2006. Fluxes of N_2O and CH_4 from soils of savannas and seasonally-dry ecosystems. J. Biogeogr. 33 (3), 401–415.
Castro, M.S., Steudler, P.A., Melillo, J.M., Aber, J.D., Millham, S., 1993. Exchange of N_2O and CH_4 between the atmosphere and soils in spruce-fir forest in the Northeastern United States. Biogeochemistry 18, 119–135.
Chan, A.S.K., Steudler, P.K., Bowden, R.D., Gulledge, J., Cavanaugh, C.M., 2005. Consequences of nitrogen fertilization on soil methane consumption in a productive temperate deciduous forest. Biol. Fertil. Soil. 41, 182–189.
Chapuis-Lardy, L., Wrage, N., Metay, A., Chotte, J.L., Bernoux, M., 2007. Soils, a sink for N_2O? A review. Glob. Change Biol. 13, 1–17.
Chimner, R.A., Cooper, D.J., 2003. Influence of water table levels on CO_2 emissions in a Colorado sub-alpine fen: an in situ microcosm study. Soil Biol. Biochem. 35 (3), 345–351.
Chirinda, N., Plauborg, F., Heckrath, G., Elsgaard, L., Thomsen, I.K., Olesen, J.E., 2014. Carbon dioxide in arable soil profiles: a comparison of automated and manual measuring systems. Commun. Soil Sci. Plant Anal. 45 (9), 1278–1291.
Choudhary, M.A., Akramkhanov, A., Saggar, S., 2002. Nitrous oxide emissions from a New Zealand cropped soil: tillage effects, spatial and seasonal variability. Agric. Ecosyst. Environ. 93, 33–43.
Christensen, S., Christensen, B.T., 1991. Organic matter available for denitrification in different soil fractions: effect of freeze/thaw cycles and straw disposal. J. Soil Sci. 42, 637–647.
Christiansen, J.R., Korhonen, J.F.J., Juszczak, R., Giebels, M., Pihlatie, M., 2011. Assessing the effects of chamber placement, manual sampling and headspace mixing on CH_4 fluxes in a laboratory experiment. Plant Soil 343, 171–185.
Correia, A.C., Minunno, F., Caldeira, M.C., Banza, J., Mateus, J., Carneiro, M., Wingate, L., Shvaleva, A., Ramos, A., Jongen, M., Bugalho, M.N., Nogueira, C., Lecomte, X., Pereira, J.S., 2012. Soil water availability strongly modulates soil CO_2 efflux in different Mediterranean ecosystems: model calibration using the Bayesian approach. Agric. Ecosyst. Environ. 161, 88–100.
Correia, A.C., Costa e Silva, F., Correia, A.V., Hussain, M.Z., Rodrigues, A.D., David, J.S., Pereira, J.S., 2014. Carbon sink strength of a Mediterranean cork oak understorey: how do semi-deciduous and evergreen shrubs face summer drought? J. Veg. Sci. 25, 411–426.
Cowan, N.J., Norman, P., Famulari, D., Levy, D., Reay, D.S., Skiba, U.M., 2015. Spatial variability and hotspots of soil N_2O fluxes from intensively grazed grassland. Biogeoscience 12, 1585–1596.
Cox, P.M., Betts, R.A., Jones, C.D., Spall, S.A., Totterdell, I.J., 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. Nature 408, 184–187.
Cuhel, J., Simek, M., Laughlin, R.J., Bru, D., Chèneby, D., Watson, C.J., Philippot, L., 2010. Insights into the effect of soil pH on N_2O and N_2 emissions and denitrifier community size and activity. Appl. Environ. Microbiol. 76, 1870–1878.
Dalal, R.C., Allen, D.E., 2008. Greenhouse gas fluxes from natural ecosystems. Turner review no. 18, Australian. J. Bot. 56, 369–407.
Davidson, E.A., Savage, K., Verchot, L.V., Navarro, R., 2002. Minimizing artifacts and biases in chamber-based measurements of soil respiration. Agric. For. Meteorol. 113 (1–4), 21–37.
de Dato, G., De Angelis, P., Sirca, C., Beier, C., 2010. Impact of drought and increasing temperatures on soil CO_2 emissions in a Mediterranean shrubland (gariga). Plant Soil 327, 153–166.
Degryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. Glob. Change Biol. 10, 1120–1132.
Desjardins, R.L., MacPherson, J.I., Mahrt, L., Schuepp, P., Pattey, E., Neumann, H., Baldocchi, D., Wofsy, S., Fitzjarrald, D., McCaughey, H., Joiner, D.W., 1997. Scaling up flux measurements for the boreal forest using aircraft-tower combinations. J. Geophys. Res. 102, 29125.
Desjardins, R.L., Pattey, E., Smith, W.N., Worth, D., Grant, B., Srinivasan, R., MacPherson, J.I., Mauder, M., 2010. Multiscale estimates of N_2O emissions from agricultural lands. Agric. For. Meteorol. 150 (6), 817–824.
Dilustro, J.J., Collins, B., Duncan, L., Crawford, C., 2005. Moisture and soil texture effects on soil CO_2 efflux components in southeastern mixed pine forests. For. Ecol. Manage. 204, 87–97.
Ding, W., Cai, Z., Tsuruta, H., Li, X., 2003. Key factors affecting spatial variation of methane emissions from freshwater marshes. Chemosphere 51, 167–173.
Dixon, R.K., Brown, S., Houghton, R.A., Solomon, A.M., Trexler, M.C., Wisniewski, J., 1994. Carbon pools and flux of global forest ecosystems. Science 263, 185–190.
Dorodnikov, M., Blagodatskaya, E., Blagodatsky, S., Marhan, S., Fangmeier, A., Kuzyakov, Y., 2009. Stimulation of microbial extracellular enzyme activities by elevated CO_2 depends on soil aggregate size. Glob. Change Biol. 15, 1603–1614.
Drewer, J., Finch, J.W., Lloyd, C.R., Baggs, E.M., Skiba, U., 2012. How do soil emissions of N_2O, CH_4 and CO_2 from perennial bioenergy crops differ from arable annual crops? GCB Bioenergy 4, 408–419.
Du, Q., Liu, H., Feng, J., Wang, L., 2013. Effects of different gap filling methods and land surface energy balance closure on annual net ecosystem exchange in a semiarid area of China. Sci. China Earth Sci. 57 (6), 1340–1351.
Dutaur, L., Verchot, L.V., 2007. A global inventory of the soil CH_4 sink. Glob. Biogeochem. Cycles 21 (n/a), http://dx.doi.org/10.1029/2006GB002734.
EPA, 2010. Methane and nitrous oxide emissions from natural sources. Report EPA 430-R-10-001 Office of Atmospheric Programs (6207J) Washington, DC., to be found under nepis.epa.gov, then search for document name; last access December 08, 2015.
Edwards, N.T., Riggs, J.S., 2003. Automated monitoring of soil respiration. A moving chamber design. Soil Sci. Soc. Am. J. 67, 1266–1271.
Eickenscheidt, N., Brumme, R., 2012. NO_x and N_2O fluxes in a nitrogen-enriched European spruce forest soil under experimental long-term reduction of nitrogen depositions. Atmos. Environ. 60, 51–58.
Eshela, G., Lifschitz, D., Bonfil, D.J., Sternberg, M., 2014. Carbon exchange in rainfed wheat fields: effects of long-term tillage and fertilization under arid conditions. Agric. Ecosyst. Environ. 195, 112–119.
FAO, 1998. Wetland characterization and classification for sustainable agricultural development. Food and Agriculture Organization of the United Nations, Harare. http://www.fao.org/docrep/003/x6611e/x6611e00.HTM (last access December 08, 2015).
FAO, 2014. Global Land Cover-SHARE of year 2014—Beta-Release 1.0. http://www.glcn.org/databases/lc_glcshare_en.jsp (last access April 18, 2016).
FAO, 2014. FAOSTAT Food and Agriculture Organization of the United Nations, Rome, http://faostat3.fao.org/home/E (last access April 18, 2016).
Falloon, P., Smith, P., 2012. Modelling soil carbon dynamics. In: Soil Carbon Dynamics. An Integrated Methodology. Cambridge University Press, pp. 221–244.
Fang, C., Moncrieff, J.B., 2001. The dependence of soil CO_2 efflux on temperature. Soil Biol. Biochem. 33, 155–165.
Feig, G.T., Mamtimin, B., Meixner, F.X., 2008. Use of laboratory and remote sensing techniques to estimate vegetation patch scale emissions of nitric oxide from an arid Kalahari savannah. Biogeosci. Discus. 5, 4621–4680.
Fernández-Luqueño, F., Reyes-Varela, V., Martínez-Suárez, C., Reynoso-Keller, R.E., Méndez-Bautista, J., Ruiz-Romero, E., López-Valdez, F., Luna-Guido, M.L., Dendooven, L., 2009. Emission of CO_2 and N_2O from soil cultivated with common bean (*Phaseolus vulgaris* L.) fertilized with different N sources. Sci. Total Environ. 407, 4289–4296.
Fiedler, S., Sommer, M., 2000. Methane emissions, groundwater levels, and redox potentials of common wetland soils in a temperate-humid climate. Glob. Biogeochem. Cycle 14 (4), 1081–1093.
Fiedler, S., Höll, B.S., Jungkunst, H.F., 2005. Methane budget of a Black Forest spruce ecosystem considering soil pattern. Biogeochemistry 76, 1–20.
Fleck, D., He, Y., Alexander, C., Jacobsen, G., Cunningham, K.L., 2013. Simultaneous soil flux measurements of five gases —N_2O, CH_4, CO_2, NH_3, and H_2O with the Picarro G2508. Santa Clara. Picarro application note, http://www.picarro.com/application_notes/an034_simultaneous_soil_flux_measurements_of_five_gases_n2o_ch4_co2_nh3_and_h2o_wi (last access December 08, 2015).
Forbrich, I., Kutzbach, L., Hormann, A., Wilmking, M., 2010. A comparison of linear and exponential regression for estimating diffusive CH_4 fluxes by closed-chambers in peatlands. Soil Biol. Biochem. 42, 507–515.
Fornara, D.A., Tilman, D., 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. J. Ecol. 96, 314–322.
Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Changes in Atmospheric Constituents and in Radiative Forcing. In Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Fowler, D., Pilegaard, K., Sutton, M.A., Ambus, P., Raivonen, M., Duyzer, J., Simpson, D., Fagerli, H., Fuzzi, S., Schjoerring, J.K., Granier, C., Neftel, A., Isaksen, I.S.A., Laj, P., Maione, M., Monks, P.S., Burkhardt, J., Daemmgen, U., Neirynck, J., Personne, E., Wichink-Kruit, R., Butterbach-Bahl, K., Flechard, C., Tuovinen, J.P., Coyle, M., Gerosa, G., Loubet, B., Altimir, N., Gruenhage, L., Ammann, C., Cieslik, S., Paoletti, E., Mikkelsen, T.N., Ro-Poulsen, H., Cellier, P., Cape, J.N., Horváth, L., Loreto, F., Niinemets Ü, Palmer, P.I., Rinne, J., Misztal, P., Nemitz, E., Nilsson, D., Pryor, S., Gallagher, M.W., Vesala, T., Skiba, U., Brüggemann, N., Zechmeister-Boltenstern, S., Williams, J., O'Dowd, C., Facchini, M.C., Leeuw, G., de, Flossman, A., Chaumerliac, N., Erisman, J.W., 2009. Atmospheric composition change: ecosystems-Atmosphere interactions. Atmos. Environ. 43 (33), 5193–5267.
Frankenberg, C., Platt, U., Wagner, T., 2005. Iterative maximum a posteriori (IMAP)-DOAS for retrieval of strongly absorbing trace gases: model studies for CH_4 and CO_2 retrieval from near infrared spectra of SCIAMACHY onboard ENVISAT. Atmos. Chem. Phys. 5, 9–22.
Freibauer, A., Kaltschmitt, M., 2003. Controls and models for estimating direct nitrous oxide emissions from temperate and sub-boreal agricultural mineral soils in Europe. Biogeochemistry 63, 93–115.
Fumoto, T., Kobayashi, K., Li, C., Yagi, K., Hasegawa, T., 2008. Revising a process-based biogeochemistry model (DNDC) to simulate methane emission from rice paddy fields under various residue management and fertilizer regimes. Glob. Change Biol. 14, 382–402.
G7, 2015. Leaders declaration G7 summit. 8. June 2015. Schloss Elmau. http://www.bundesregierung.de/Content/EN/_Anlagen/G7/2015-06-08-g7-abschluss-eng_en.pdf?__blob=publicationFile&v=1 (last access April 18, 2016).

Galbally, I.E., Freney, J.R., Muirhead, W.A., Simpson, J.R., Trevitt, A.C.F., Chalk, P.M., 1987. Emission of nitrogen oxides (NO_x) from a flooded soil fertilized with urea: relation to other nitrogen loss processes. J. Atmos. Chem. 5, 343–365.

Gao, B., Ju, X., Su, F., Meng, Q., Oenema, O., Christie, P., Chen, X., Zhang, F., 2014. Nitrous oxide and methane emissions from optimized and alternative cereal cropping systems on the North China Plain: a two-year field study. Sci. Total Environ. 472, 112–124.

Gillenwater, M., Sussman, F., Cohen, J., 2007. Practical policy applications of uncertainty analysis for national greenhouse gas inventories. Water Air Soil Pollut. 7, 451–474.

Gnaana Saraswathi, S., Lalrammawia, C., Paliwal, K., 2008. Seasonal variability in soil-surface CO_2 efflux in selected young tree plantations in semi-arid eco-climate of Madurai. Curr. Sci. 95, 94–99.

Gogoi, B., Baruah, K.K., 2011. Nitrous oxide emission from tea (*Camellia sinensis* (L.) O. kuntze)-planted soils of North East India and soil parameters associated with the emission. Curr. Sci. 101, 531–536.

Goldberg, S.D., Gebauer, G., 2009. N_2O and NO fluxes between a Norway spruce forest soil and atmosphere as affected by prolonged summer drought. Soil Biol. Biochem. 41, 1986–1995.

Gong, J., Ge, Z., An, R., Duan, Q., You, X., Huang, Y., 2012. Soil respiration in poplar plantations in northern China at different forest ages. Plant Soil 360, 109–122.

Gorczyca, Z., 2013. Concentration of radiocarbon in soil-respired CO_2 flux: data-model comparison for three different ecosystems in southern Poland. Radiocarbon 55 (2–3), 1521–1532, http://dx.doi.org/10.2458/azu_js_rc.55.16321 (last access April 18, 2016).

Greicius, T., Perez, M., Espinoza, L., Orr, K., 2014. NASA set for a big year in Earth Science. http://www.jpl.nasa.gov/news/news.php?release=2014-019 (last access April 18, 2016).

Griffith, D.W.T., Deutscher, N.M., Caldow, C., Kettlewell, G., Riggenbach, M., Hammer, S., 2012. A Fourier transform infrared trace gas and isotope analyser for atmospheric applications. Atmos. Meas. Tech. 5, 2481–2498.

Griffiths, P.R., Shao, L., Leytem, A.B., 2009. Completely automated open-path FT-IR spectrometry. Anal. Bioanal. Chem. 393, 45–50.

Gritsch, C., Zimmermann, M., Zechmeister-Boltenstern, S., 2015. Interdependencies between temperature and moisture sensitivities of CO_2 emissions in European land ecosystems. Biogeosci. 12, 5981–5993.

Groffman, P.M., Hardy, J.P., Driscoll, C.T., Fahey, T.J., 2006. Snow depth, soil freezing, and fluxes of carbon dioxide, nitrous oxide and methane in a northern hardwood forest. Glob. Change Biol. 12, 1748–1760.

Groffman, P.M., Butterbach-Bahl, K., Fulweiler, R.W., Gold, A.J., Morse, J.L., Stander, E.K., Tague, C., Tonitto, C., Vidon, P., 2009. Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. Biogeochemistry 93, 49–77.

Grunwald, D., Fender, A.C., Erasmi, S., Jungkunst, H.F., 2012. Towards improved bottom-up inventories of methane from the European land surface. Atmos. Environ. 51, 203–211.

Gu, J., Nicoullaud, B., Rochette, P., Grossel, A., Hénault, C., Cellier, P., Richard, G., 2013. A regional experiment suggests that soil texture is a major control of N_2O emissions from tile-drained winter wheat fields during the fertilization period. Soil Biol. Biochem. 60, 134–141.

Gu, J., Loustau, D., Hénault, C., Rochette, P., Cellier, P., Nicoullaud, B., Grossel, A., Richard, G., 2014. Modeling nitrous oxide emissions from tile-drained winter wheat fields in Central France. Nutr. Cyc. Agroecosyst. 98, 27–40.

Günther, A., Huth, V., Jurasinski, G., Glatzel, S., 2014. The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. GCB Bioenergy 7 (5), 1092–1106, http://dx.doi.org/10.1111/gcbb.12214.

Gundersen, P., Christiansen, J.R., Alberti, G., Brüggemann, N., Castaldi, S., Gasche, R., Kitzler, B., Klemedtsson, L., Lobo-do-Vale, R., Moldan, F., Rütting, T., Schleppi, P., Weslien, P., Zechmeister-Boltenstern, S., 2012a. The response of methane and nitrous oxide fluxes to forest change in Europe. Biogeoscience 9, 3999–4012.

Gundersen, P., Christiansen, J.R., Frederiksen, P., Vesterdal, L., 2012b. Influence of hydromorphic soil conditions on greenhouse gas emissions and soil carbon stocks in a Danish temperate forest. For. Ecol. Manage. 284, 185–195.

Hall, S.J., Huber, D., Grimm, N.B., 2008. Soil N_2O and NO emissions from an arid, urban ecosystem. J. Geophys. Res.: Biogeosci. 113, G1, http://dx.doi.org/10.1029/2007JG000523 (last access April 18, 2016).

Hanson, P.J., Edwards, N.T., Garten, C.T., Andrews, J.A., 2000. Separating root and soil microbial contributions to soil respiration: a review of methods and observations. Biogeochemistry 48, 115–146.

Hastings, A.F., Wattenbach, M., Eugster, W., Li, C., Buchmann, N., Smith, P., 2010. Uncertainty propagation in soil greenhouse gas emission models: An experiment using the DNDC model and at the Oensingen cropland site. Agric. Ecosyst. Environ. 136 (1–2), 97–110.

Hayakawa, A., Akiyama, H., Sudo, S., Yagi, K., 2009. N_2O and NO emissions from an Andisol field as influenced by pelleted poultry manure. Soil Biol. Biochem. 41, 521–529.

Heinemeyer, A., McNamara, N.P., 2011. Comparing the closed static versus the closed dynamic chamber flux methodology: implications for soil respiration studies. Plant Soil 346, 145–151.

Heinemeyer, A., Di Bene, C., Lloyd, A.R., Tortorella, D., Baxter, R., Huntley, B., Gelsomino, A., Ineson, P., 2011. Soil respiration: implications of the plant-soil continuum and respiration chamber collar-insertion depth on measurement and modelling of soil CO_2 efflux rates in three ecosystems. Eur. J. Soil Sci. 62, 82–94.

Hellebrand, H.J., Kern, J., Scholz, V., 2003. Long-term studies on greenhouse gas fluxes during cultivation of energy crops on sandy soils. Atmos. Environ. 37, 1635–1644.

Herold, M., Mayaux, P., Woodcock, C.E., Baccini, A., Schmullius, C., 2008. Some challenges in global land cover mapping, an assessment of agreement and accuracy in existing 1 km datasets. Remote Sens. Environ. 112 (5), 2538–2556, Earth observations for terrestrial biodiversity and ecosystems, Spec. Iss.

Holst, J., Liu, C., Yao, Z., Brüggemann, N., Zheng, X., Giese, M., Butterbach-Bahl, K., 2008. Fluxes of nitrous oxide, methane and carbon dioxide during freezing–thawing cycles in an Inner Mongolian steppe. Plant Soil 308, 105–117.

Holzapfel-Pschorn, A., Conrad, R., Seiler, W., 1985. Production, oxidation and emission of methane in rice paddies. FEMS Microbiol. Lett. 31, 343–351.

Hou, L.Y., Wang, Z.P., Wang, J.M., Wang, B., Zhou, S.B., Li, L.H., 2012. Growing season in situ uptake of atmospheric methane by desert soils in a semiarid region of northern China. Geoderma 189–190, 415–422, http://dx.doi.org/10.1016/j.geoderma.2012.05.012 (last access April 18, 2016).

Houghton, J.T., Jenkins, G.J., Ephraums, J.J. (Eds.), 1990. Report Prepared for Intergovernmental Panel on Climate Change by Working Group I. Cambridge University Press, Cambridge, Great Britain, New York, NY, USA and Melbourne, Australia, p. 410.

Hu, Y.G., Feng, Y.L., Zhang, Z.S., Huang, L., Zhang, P., Xu, B.X., 2014. Greenhouse gases fluxes of biological soil crusts and soil ecosystem in the artificial sand-fixing vegetation region in Shapotou area. Chin. J. Appl. Ecol. 25, 61–68.

Hu, Q., Wu, Q., Yao, B., Xu, X., 2015. Ecosystem respiration and its components from a Carex meadow of Poyang Lake during the drawdown period. Atmos. Environ. 100, 124–132.

Hungershoefer, K., Breon, F.M., Peylin, P., Chevallier, F., Rayner, P., Klonecki, A., Houweling, S., Marshall, J., 2010. Evaluation of various observing systems for the global monitoring of CO_2 surface fluxes. Atmos. Chem. Phys. 10, 10503–10520.

Hutchinson, G.L., Mosier, A.R., 1979. Nitrous oxide emissions from an irrigated cornfield. Sci. New Ser. 205 (September (4411)), 1125–1127.

IPCC, 2007. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2013. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, p. 1535.

Iqbal, J., Lin, S., Hu, R., Feng, M., 2009. Temporal variability of soil-atmosphre CO_2 and CH_4 fluxes from different land uses in mid-subtropical China. Atmos. Environ. 43 (37), 5865–5875, http://dx.doi.org/10.1016/j.atmosenv.2009.08.025 (last access April 18, 2016).

Izaurralde, R.C., Lemke, R.L., Goddard, T.W., McConkey, B., Zhang, Z., 2004. Nitrous oxide emissions from agricultural toposequences in Alberta and Saskatchewan. Soil Sci. Soc. Am. J. 68, 1285.

Jäger, N., Stange, C.F., Ludwig, B., Flessa, H., 2011. Emission rates of N_2O and CO_2 from soils with different organic matter content from three long-term fertilization experiments—a laboratory study. Biol. Fertil. Soil. 47, 483–494.

Jacinthe, P.A., Lal, R., 2006. Spatial variability of soil properties and trace gas fluxes in reclaimed mine land of southeastern Ohio. Geoderma 136, 598–608.

Jiang, J., Guo, S., Zhang, Y., Liu, Q., Wang, R., Wang, Z., Li, N., Li, R., 2015. Changes in temperature sensitivity of soil respiration in the phases of a three-year crop rotation system. Soil Tillage Res. 150, 139–146.

Jin, Z., Dong, Y.S., Qi, Y.C., Domroes, M., 2009. Precipitation pulses and soil CO_2 emission in desert shrubland of *Artemisia ordosica* on the Ordos Plateau of Inner Mongolia, China. Pedosphere 19, 799–807.

Jungkunst, H.F., Fiedler, S., 2007. Latitudinal differentiated water table control of carbon dioxide, methane and nitrous oxide fluxes from hydromorphic soils: feedbacks to climate change. Glob. Change Biol. 13, 2668–2683.

Kallenbach, C.M., Rolston, D.E., Horwath, W.R., 2010. Cover cropping affects soil N_2O and CO_2 emissions differently depending on type of irrigation. Agric. Ecosyst. Environ. 137 (3–4), 251–260, http://dx.doi.org/10.1016/j.agee.2010.02.010 (last access April 18, 2016).

Kammer, A., Tuzson, B., Emmenegger, L., Knohl, A., Mohn, J., Hagedorn, F., 2011. Application of a quantum cascade laser-based spectrometer in a closed chamber system for real-time $\delta^{13}C$ and $\delta^{18}O$ measurements of soil-respired CO_2. Agric. For. Meteorol. 151, 39–48.

Kampfl, G., Kristóf, K., Algaidi, A.A., Bayoumi Hamuda, H.E.A.F., Heltai, G., 2007. Study of NO_x and CO_2 production of cultivated soil in closed microcosm experimental system. Microchem. J. 85, 31–38.

Kasimir-Klemedtsson, A., Klemdtsson, L., Berglund, K., Martikainen, P.J., Silvola, J., Oenema, O., 1997. Greenhouse gas emissions from farmed organic soils: a review. Soil Use Manage. 13, 245–250.

Keller, M., Reiners, W.A., 1994. Soil-atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica. Glob. Biogeochem. Cycles 8, 399–409, http://dx.doi.org/10.1029/94GB01660 (last access April 18, 2016).

Kelliher, F.M., Reisinger, A.R., Martin, R.J., Harvey, M.J., Price, S.J., Sherlock, R.R., 2002. Measuring nitrous oxide emission rate from grazed pasture using Fourier-transform infrared spectroscopy in the nocturnal boundary layer. Agric. For. Meteorol. 111, 29–38.

Kim, Y.S., 2013. Soil-atmosphere exchange of CO_2, CH_4 and N_2O in northern temperate forests: effects of elevated CO_2 concentration, N deposition and forest fire. Eurasian J. For. Res. 16 (1), 1–43.
Kitzler, B., Holtermann, C., Zechmeister-Boltenstern, S., Skiba, U., Butterbach-Bahl, K., 2006. Nitrogen oxides emission from two beech forests subjected to different nitrogen loads. Biogeoscience 3, 293–310.
Klopatek, J.M., 2002. Belowground carbon pools and processes in different age stands of Douglas-fir. Tree Physiol. 22, 197–204.
Kluber, L.A., Miller, J.O., Ducey, T.F., Hunt, P.G., Lang, M., Ro, K.S., 2014. Multistate assessment of wetland restoration on CO_2 and N_2O emissions and soil bacterial communities. Appl. Soil Ecol. 76, 87–94.
Knox, S.H., Sturtevant, C., Matthes, J.H., Koteen, L., Verfaillie, J., Baldocchi, D., 2015. Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta. Glob. Change Biol. 21 (2), 750–765, http://dx.doi.org/10.1111/gcb.12745 (last access April 18, 2016).
Koch, K., 2015. Winter CO_2 Respiration of Agricultural Soils in Saxony. Unpubl. B. Sc. Hon. Thesis. TU Bergakademie, Freiberg.
Kögel-Knabner, I., Matzner, E., Knickner, H., Kandeler, E., Guggenberger, G., 1998. Structure and stability. Mitt. Dtsch. Bodenkdl. Ges. 87, 51–56.
Kögel-Knabner, I., Amelung, W., Cao, Z., Fiedler, S., Frenzel, P., Jahn, R., Kalbitz, K., Kölbl, A., Schloter, M., 2010. Biogeochemistry of paddy soils. Geoderma 157, 1–14.
Koljonen, T., 1992. The geochemical atlas of Finland. Part 2: Till. Geol. Surv Finland, Espoo, Finland, 218 p.
Konnerup, D., Betancourt-Portela, J.M., Villamil, C., Parra, J.P., 2014. Nitrous oxide and methane emissions from the restored mangrove ecosystem of the Ciénaga Grande de Santa Marta, Colombia. Estuar. Coast. Shelf Sci. 140, 43–51.
Kormann, R., Müller, H., Werle, P., 2001. Eddy flux measurements of methane over the den Murnauer Moos, 11°11′E, 47°39′N, using a fast tunable diode laser spectrometer. Atmos. Environ. 35 (14), 2533–2544.
Kustas, W.P., Anderson, M.C., French, A.N., Vickers, D., 2006. Using a remote sensing field experiment to investigate flux-footprint relations and flux sampling distributions for tower and aircraft-based observations. Adv. Water Res. 29 (2), 355–368, Exerimental Hydrology: A Bright Future.
Kutsch, W., Bahn, M., Heinemeyer, A., 2009. Soil Carbon Dynamics: An Integrated Methodology, xi. Cambridge University Press, Cambridge, UK, New York, 286 p.
Kutzbach, L., Schneider, J., Sachs, T., Giebels, M., Nykänen, H., Shurpali, N.J., Martikainen, P.J., Alm, J., Wilmking, M., 2007. CO_2 flux determination by closed-chamber methods can be seriously biased by inappropriate application of linear regression. Biogeoscience 4, 1005–1025.
Kuze, A., Suto, H., Nakajima, M., Hamazaki, T., 2009. Thermal and near infrared sensor for carbon observation Fourier-transform spectrometer on the greenhouse gases observing Satellite for greenhouse gases monitoring. Appl. Opt. 48, 6716–6733, http://dx.doi.org/10.1364/AO.48.006716 (last access April 18, 2016).
La Scala Jr., N., Bolonhezi, D., Pereira, G.T., 2006. Short-term soil CO_2 emission after conventional and reduced tillage of a no-till sugar cane area in southern Brazil. Soil Tillage Res. 91, 244–248.
Lai, L., Zhao, X., Jiang, L., Wang, Y., Luo, L., Zheng, Y., Chen, X., Rimmington, G.M., 2012. Soil respiration in different agricultural and natural ecosystems in an arid region. PLoS One 7 (10), e48011, http://dx.doi.org/10.1371/journal.pone.0048011 (last access April 18, 2016).
Lal, R., Follett, R.F. (Eds.), 2001. Soil Carbon Sequestration and the Greenhouse Effect. Soil Sci. Soc. Am., Spec. Publ. 57, Madison, ISBN: 978-0-89118-850-6.
Lamers, M., Ingwersen, J., Streck, T., 2007. Nitrous oxide emissions from mineral and organic soils of a Norway spruce stand in South-West Germany. Atmos. Environ. 41, 1681–1688.
Latham, J., Cumani, R., Rosati, I., Bloise, M., 2014. Global Land Cover SHARE (GLC-SHARE) database. Beta-Release Version 1.0. http://www.glcn.org/downs/prj/glcshare/GLC_SHARE_beta_v1.0_2014.pdf (last access April 18, 2016).
Launiainen, S., Rinne, J., Pumpanen, J., Kulmala, L., Kolari, P., Keronen, P., Siivola, E., Pohja, T., Hari, P., Vesala, T., 2005. Eddy covariance measurements of CO_2 and sensible and latent heat fluxes during a full year in a boreal pine forest trunk-space. Boreal Environ. Res. 10, 569–588.
Laville, P., Flura, D., Gabrielle, B., Loubet, B., Fanucci, O., Rolland, M.N., Cellier, P., 2009. Characterisation of soil emissions of nitric oxide at field and laboratory scale using high-resolution method. Atmos. Environ. 43 (16), 2648–2658.
Law, B.E., Harmon, M.E., 2011. Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. Carbon Manage. 2 (1), 73–84.
Law, B.E., Ryan, M.G., Anthoni, P.M., 1999. Seasonal and annual respiration of a ponderosa pine ecosystem. Glob. Change Biol. 5, 169–182.
Leppelt, T., Dechow, R., Gebbert, S., Freibauer, A., Lohila, A., Augustin, J., Drösler, M., Fiedler, S., Glatzel, S., Höper, H., Järveoja, J., Lærke, P.E., Maljanen, M., Mander, Ü., Mäkiranta, P., Minkkinen, K., Ojanen, P., Regina, K., Strömgren, M., 2014. Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. Biogeoscience 11, 6595–6612.
Leppin, J.W., 2015. Net Ecosystem Exchange (CO_2) in Winter. Unpubl. B. Sc. Hon. Thesis. TU Bergakademie, Freiberg.
Li, C., Frolking, S., Frolking, T.A., 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. model structure and sensitivity. J. Geophys. Res. 97, 9759–9776.
Li, C., Frolking, S., Harriss, R., 1994. Modeling carbon biogeochemistry in agricultural soils. Glob. Biogeochem. Cycles 8, 237–254.
Li, C., Frolking, S., Butterbach-Bahl, K., 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Clim. Change 72, 321–338.
Li, H., Qiu, J.J., Wang, L., Xu, M.Y., Liu, Z., Wang, W., 2012. Estimates of N_2O emissions and mitigation potential from a spring maize field based on DNDC model. J. Integr. Agric. 11, 2067–2078.
Li, C., Zhang, Z., Guo, L., Cai, M., Cao, C., 2013. Emissions of CH_4 and CO_2 from double rice cropping systems under varying tillage and seeding methods. Atmos. Environ. 80, 438–444.
Liang, N., Nakadai, T., Hirano, T., Qu, L., Koike, T., Fujinuma, Y., Inoue, G., 2004. In situ comparison of four approaches to estimating soil CO_2 efflux in a northern larch (*Larix kaempferi* Sarg.) forest. Agric. For. Meteorol. 123, 97–117, http://dx.doi.org/10.1016/j.agrformet.2003.10.002 (last access April 18, 2016).
Liang, J., Chang-yi, L., Yong, Y., Gong-Fu, Y., 2013. Soil respiration in a subtropical mangrove wetland in the Jiulong River estuary, China. Pedosphere 23 (5), 678–685, http://dx.doi.org/10.1016/S1002-0160(13)60060-0 (last access April 18, 2016).
LI-COR, 2012. LI-810A Automated Soil CO_2 flux System. http://www.licor.com/env/products/soil_flux/ (last access April 18, 2016).
Liu, C., Holst, J., Yao, Z., Brüggemann, N., Butterbach-Bahl, K., Han, S., Han, X., Tas, B., Susenbeth, A., Zheng, X., 2009a. Growing season methane budget of an Inner Mongolian steppe. Atmos. Environ. 43, 3086–3095.
Liu, C., Holst, J., Yao, Z., Brüggemann, N., Butterbach-Bahl, K., Han, S., Han, X., Zheng, X., 2009b. Sheepfolds as hotspots of nitric oxide (NO) emission in an Inner Mongolian steppe. Agric. Ecosyst. Environ. 134, 136–142.
Liu, L., Graever, T.L., 2009. A review of nitrogen enrichment effects on three biogenic GHGs: the CO_2 sink may be largely offset by stimulated N_2O and CH_4 emission. Ecol. Lett. 12, 1103–1117.
Lohila, A., Aurela, M., Regina, K., Laurila, T., 2003. Soil and total ecosystem respiration in agricultural fields: effect of soil and crop type. Plant Soil 251, 303–317.
Ludwig, J., Meixner, F.X., Vogel, B., Förstner, J., 2001. Soil-air exchange of nitric oxide: an overview of processes, environmental factors, and modeling studies. Biogeochemistry 52, 225–257.
Lundegardh, H., 1927. Carbon dioxide evolution of soil and crop growth. Soil Sci. 23, 417–453.
Luo, G.J., Brüggemann, N., Wolf, B., Gasche, R., Grote, R., Butterbach-Bahl, K., 2012. Decadal variability of soil CO_2, NO, N_2O, and CH_4 fluxes at the Höglwald forest, Germany. Biogeoscience 9, 1741–1763.
Luyssaert, S., Inglima, I., Jung, M., Richardson, A.D., Reichstein, M., Papale, D., Piao, S.L., Schulze, E.D., Wingate, L., Matteucci, G., Aragoa, L., Aubinet, M., Beer, C., Bernhofer, C., Black, K.G., Bonal, D., Bonnefond, J.M., Chambers, J., Ciais, P., Cook, B., Davis, K.J., Dolman, A.J., Gielen, B., Goulden, M., Grace, J., Granier, A., Grelle, A., Griffis, T., Grünwald, T., Guidolotti, G., Hanson, P.J., Harding, R., Hollinger, D.Y., Hutyra, L.R., Kolari, P., Kruijt, B., Kutsch, W., Lagergren, F., Laurila, T., Law, B.E., Le Marie, G., Lindroth, A., Loustau, D., Malhi, Y., Mateus, J., Migliavacca, M., Misson, L., Montagnani, L., Moncrieff, J., Moors, E., Munger, J.W., Nikinmaa, E., Ollinger, S.V., Pita, G., Rebmann, C., Roupsard, O., Saigussa, N., Sanz, M.J., Seufert, G., Sierra, C., Smith, M.L., Tang, J., Valentini, R., Vesala, T., Janssebs, I.A., 2007. CO_2 balance of boreal, temperate, and tropical forests derived from a global database. Glob. Change Biol. 13, 2509–2537, http://dx.doi.org/10.1111/j.1365-2486.2007.01439x (last access April 18, 2016).
MacKenzie, M.D., DeLuca, T.H., Sala, A., 2004. Forest structure and organic horizon analysis along a fire chronosequence in the low elevation forests of western Montana. For. Ecol. Manage. 203, 331–343.
Maeck, A., Hofmann, H., Lorke, A., 2014. Pumping methane out of aquatic sediments–ebullition forcing mechanisms in an impounded river. Biogeoscience 11, 2925–2938.
Maljanen, M., Komulainen, V.M., Hytönen, J., Martikainen, P.J., Laine, J., 2004. Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics. Soil Biol. Biochem. 36, 1801–1808.
Maljanen, M., Hytönen, J., Mäkiranta, P., Alm, J., Minkkinen, K., Laine, J., Martikainen, P.J., 2007. Greenhouse gas emissions from cultivated and abandoned organic croplands in Finland. Boreal Environ. Res. 12, 133–140.
Marín-Muñiz, J.L., Hernández, M.E., Moreno-Casasola, P., 2015. Greenhouse gas emissions from coastal freshwater wetlands in Veracruz Mexico: effect of plant community and seasonal dynamics. Atmos. Environ. 107, 107–117.
Marwinski, I., Neubert, T., 2015. Greenhouse Gas Balances of Small Inland Waters and Wetlands. Unpubl. B. Sc. Thesis. TU Bergakademie, Freiberg.
McSwiney, C.P., Robertson, G.P., 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (*Zea mays* L.) cropping system. Glob. Change Biol. 11, 1712–1719.
McTaggart, I.P., Akiyama, A., Tsuruta, H., Ball, B.C.F., 2002. Influence of soil physical properties, fertiliser type and moisture tension on N_2O and NO emissions from nearly saturated Japanese upland soils. Nutr. Cycl. Agroecosyst. 63, 207–217.
Merbold, L., Steinlin, C., Hagedorn, F., 2013. Winter greenhouse gas fluxes (CO_2 CH_4 and N_2O) from a subalpine grassland. Biogeoscience 10, 3185–3203.
Merbold, L., Eugster, W., Stieger, J., Zahniser, M., Nelson, D., Buchmann, N., 2014. Greenhouse gas budget (CO_2 CH_4 and N_2O) of intensively managed grassland following restoration. Glob. Change Biol. 20, 1913–1928.
Micks, P., Aber, J.D., Boone, R.D., Davidson, E.A., 2004. Short-term soil respiration and nitrogen immobilization response to nitrogen applications in control and nitrogen-enriched temperate forests. For. Ecol. Manage. 196, 57–70.
Monson, R.K., Lipson, D.L., Burns, S.P., Turnipseed, A.A., Delany, A.C., Williams, M.W., Schmidt, S.K., 2006. Winter forest soils respiration controlled by climate

and microbial community composition. Nature 439, 711–714, http://dx.doi.org/10.1038/nature04555 (last access April 18, 2016).
Montzka, S.A., Dlugokencky, E.J., Butler, J.H., 2011. Non-CO_2 greenhouse gases and climate change. Nature 473, 43–50, http://dx.doi.org/10.1038/nature10322 (last access April 18, 2016).
Mordhorst, A., Peth, S., Horn, R., 2014. Influence of mechanical loading on static and dynamic CO_2 efflux on differently textured and managed luvisols. Geoderma 219–220, 1–13.
Mørkved, P.T., Dörsch, P., Henriksen, T.M., Bakken, L.R., 2006. N_2O emissions and product ratios of nitrification and denitrification as affected by freezing and thawing. Soil Biol. Biochem. 38, 3411–3420.
Muhammad, W., Vaughan, S., Dalal, R., Menzies, N., 2011. Crop residues and fertilizer nitrogen influence residue decomposition and nitrous oxide emission from a Vertisol. Biol. Fertil. Soil 47, 15–23.
Musenze, R.S., Werner, U., Grinham, A., Udy, J., Yuan, Z., 2015. Methane and nitrous oxide emissions from a subtropical coastal embayment (Moreton Bay, Australia). J. Environ. Sci. (China) 29, 82–96.
Myklebust, M.C., Hipps, L.E., Ryel, R.J., 2008. Comparison of eddy covariance, chamber, and gradient methods of measuring soil CO_2 efflux in an annual semi-arid grass, Bromus tectorum. Agric. For. Meteorol. 148, 1894–1907.
NASA, 2014. Orbiting Carbon Observatory (OCO-2). http://www.nasa.gov/mission_pages/oco2/index.html (last access April 18, 2016).
Nay, S.M., Mattson, K.G., Bormann, B.T., 1994. Biases of chamber methods for measuring soil CO_2 efflux demonstrated with a laboratory apparatus. Ecology 75, 2460–2463.
Ng, B.J.L., Hutyra, L.R., Nguyen, H., Cobb, A.R., Kai, F.M., Harvey, C., Gandois, L., 2014. Carbon fluxes from an urban tropical grassland. Environ. Pollut. 14, S0269–S7491, http://dx.doi.org/10.1016/j.envpol.2014.06.009 (last access April 18, 2016).
Nicolini, G., Castaldi, S., Fratini, G., Valentini, R., 2013. A literature overview of micrometeorological CH_4 and N_2O flux measurements in terrestrial ecosystems. Atmos. Environ. 81, 311–319.
Nieder, R., Benbi, D.K., 2008. Carbon and Nitrogen in the Terrestrial Environment. Springer Science, 430 pp.
Niu, S., Wu, M., Han, Y.I., Xia, J., Zhang, Z.H., Yang, H., Wan, S., 2010. Nitrogen effects on net ecosystem carbon exchange in a temperate steppe. Glob. Change Biol. 16, 144–155.
Nugroho, R.A., Röling, W.F.M., Laverman, A.M., Verhoef, H.A., 2007. Low nitrification rates in acid Scots pine forest soils are due to pH-related factors. Microbiol. Ecol. 53, 89–97.
Nyamadzawu, G., Wuta, M., Nyamangara, J., Smith, J.L., Rees, R.M., 2014. Nitrous oxide and methane emissions from cultivated seasonal wetland (sambo) soils with inorganic, organic and integrated nutrient management. Nutr. Cycl. Agroecosyst. 100 (2), 161–175.
Oertel, C., Kartuzova, M., Matschullat, J., Ratnikov, A., 2011. CO_2-fluxes from disturbed soil samples—a pot experiment with different fertilisers and plant cover. In: Department of Extension and Information IUNG-PIB (Ed.), Protection of Agricultural Soils Against Joint Stress of Natural and Anthropogenic Factors: International Conference 13th-16th June 2011, Pulawy, Poland, 47–48.
Oertel, C., Herklotz, K., Matschullat, J., Zimmermann, F., 2012. Nitric oxide emissions from soils: a case study with temperate soils from Saxony, Germany. Environ. Earth Sci. 66, 2343–2351.
Oertel, C., Matschullat, J., Andreae, H., Drauschke, T., Schröder, C., Winter, C., 2015a. Soil respiration at forest sites in Saxony (Central Europe). Environ. Earth Sci. 74, 2405–2412.
Oertel, C., Börner, E., Kempe, H., Matschullat, J. 2015b. Kammersystem für die Analyse von Gasflüssen von Ökosystemen. DE 10 2014 000 816 A1 2015.07.23.
Omonode, R.A., Smith, D.R., Gál, A., Vyn, T.J., 2011. Soil nitrous oxide emissions in corn following three decades of tillage and rotation treatments. Soil Sci. Soc. Am. J. 75 (1), 152–163.
Papale, D., Reichstein, M., Aubinet, M., Canfora, E., Bernhofer, C., Kutsch, W., Longdoz, B., Rambal, S., Valentini, R., Vesala, T., Yakir, D., 2006. Towards a standardized processing of net ecosystem exchange measured with eddy covariance technique: algorithms and uncertainty estimation. Biogeoscience 3, 571–583.
Pape, L., Ammann, C., Nyfeler-Brunner, A., Spirig, C., Hens, K., Meixner, F.X., 2009. An automated dynamic chamber system for surface exchange measurement of non-reactive and reactive trace gases of grassland ecosystems. Biogeoscience 6, 405–429.
Parkin, T.B., Venterea, R.T., Hargreaves, S.K., 2012. Calculating the detection limits of chamber-based soil greenhouse gas flux measurements. J. Environ. Qual. 41, 705–715.
Patino-Zuniga, L., Ceja-Navarro, J.A., Govaerts, B., Luna-Guido, M., Sayre, K.D., Dendooven, L., 2009. The effect of different tillage and residue management practices on soil characteristics, inorganic N dynamics and emissions of N_2O, CO_2 and CH_4 in the central highlands of Mexico: a laboratory study. Plant Soil 314, 231–241.
Pattey, E., Edwards, G.C., Desjardins, R.L., Pennock, D.J., Smith, W., Grant, B., MacPherson, J.I., 2007. Tools for quantifying N_2O emissions from agroecosystems. Agric. For. Meteorol. 142 (2–4), 103–119.
Pausch, J., Tian, J., Riederer, M., Kuzyakov, Y., 2013. Estimation of rhizodeposition at field scale: upscaling of a ^{14}C labeling study. Plant Soil 364, 273–285.
Peng, Q., Dong, Y., Qi, Y., Xiao, S., He, Y., Ma, T., 2011. Effects of nitrogen fertilization on soil respiration in temperate grassland in Inner Mongolia, China. Environ. Earth Sci. 62, 1163–1171.
Petersen, S.O., Mutegi, J.K., Hansen, E.M., Munkholm, L.J., 2011. Tillage effects on N_2O emissions as influenced by a winter cover crop. Soil Biol. Biochem. 43, 1509–1517.
Petersen, S.O., Ambus, P., Elsgaard, L., Schjønning, P., Olesen, J.E., 2013. Long-term effects of cropping system on N_2O emission potential. Soil Biol. Biochem. 57, 706–712, http://dx.doi.org/10.1016/j.soilbio.2012.08.032 (last access April 18, 2016).
Pflugmacher, D., Krankina, O.N., Cohen, W.B., Friedl, M.A., Sulla-Menashe, D., Kennedy, R.E., Nelson, P., Loboda, T.V., Kuemmerle, T., Dyukarev, E., Elsakov, V., Kharuk, V.I., 2011. Comparison and assessment of coarse resolution land cover maps for Northern Eurasia. Remote Sens. Environ. 115 (12), 3539–3553.
Phalan, B., Bertzky, M., Butchart, S.H.M., Donald, P.F., Scharlemann, J.P.W., Stattersfield, A.J., Balmford, A., 2013. Crop expansion and conservation priorities in tropical countries. PLoS One 8 (1), e51759, http://dx.doi.org/10.1371/journal.pone.0051759 (last access April 18, 2016).
Phillips, S.C., Varner, R.K., Frolking, S., Munger, J.W., Bubier, J.L., Wofsy, S.C., Crill, P.M., 2010. Interannual, seasonal, and diel variation in soil respiration relative to ecosystem respiration at wetland to upland slope at Harvard Forest. J. Geophys. Res.: Biogeosci. 115, http://dx.doi.org/10.1029/2008JG000858 (last access April 18, 2016).
Pihlatie, M.K., Christiansen, J.R., Aaltonen, H., Korhonen, J.F.J., Nordbo, A., Rasilo, T., Benanti, G., Giebels, M., Helmy, M., Sheehy, J., Jones, S., Juszczak, R., Klefoth, R., Lobo-do-Vale, R., Rosa, A.P., Schreiber, P., Serça, D., Vicca, S., Wolf, B., Pumpanen, J., 2013. Comparison of static chambers to measure CH_4 emissions from soils. Agric. For. Meteorol. 171–172, 124–136.
Pilegaard, K., Skiba, U., Ambus, P., Beier, C., Brüggemann, N., Butterbach-Bahl, K., Dick, J., Dorsey, J., Duyzer, J., Gallagher, M., Gasche, R., Horvath, L., Kitzler, B., Leip, A., Pihlatie, M.K., Rosenkranz, P., Seufert, G., Vesala, T., Westrate, H., Zechmeister-Boltenstern, S., 2006. Factors controlling regional differences in forest soil emission of nitrogen oxides (NO and N_2O). Biogeoscience 3, 651–661.
Poeplau, C., Don, A., 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma 192, 189–201.
Poth, M., Anderson, I.C., Sinatora Miranda, H., Carlos Miranda, A., Riggan, P.J., 1995. The magnitude and persistence of soil NO, N_2O CH_4, and CO_2 fluxes from burned tropical savanna in Brazil. Glob. Biogeochem. Cycles 9 (4), 503–513.
Priemé, A., 1994. Production and emission of methane in a brackish and a freshwater wetland. Soil Biol. Biochem. 26 (1), 7–18.
Pumpanen, J., Kolari, P., Ilvesniemi, H., Minkkinen, K., Vesala, T., Niinistö, S., Lohila, A., Larmola, T., Morero, M., Pihlatie, M., Janssens, I., Yuste, J.C., Grünzweig, J.M., Reth, S., Subke, J.A., Savage, K., Kutsch, W., Østreng, G., Ziegler, W., Anthoni, P., Lindroth, A., Hari, P., 2004. Comparison of different chamber techniques for measuring soil CO_2 efflux. Agric. For. Meteorol. 123, 159–176.
Pumpanen, J., Longdoz, B., Kutsch, W.L., 2012. Field measurements of soil respiration: principles and constraints, potentials and limitations of different methods. In: Kutsch, W.L., Bahn, M., Heinemeyer, A. (Eds.), Soil Carbon Dynamics. An Integrated Methodology. Cambridge Univ. Press.
Raich, J.W., Potter, C.S., 1995. Global patterns of carbon dioxide emissions from soils. Glob. Biogeochem. Cycles 9 (1), 23–36, http://dx.doi.org/10.1029/94GB02723 (last access April 18, 2016).
Raich, J.W., Schlesinger, W.H., 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. Tellus 44B, 81–99.
Raich, J.W., Tufekcioglu, A., 2000. Vegetation and soil respiration: correlations and controls. Biogeochemistry 48, 71–90.
Ramankutty, N., Evan, A.T., Monfreda, C., Foley, J.A., 2008. Farming the planet: 1. geographic distribution of global agricultural lands in the year 2000. Glob. Biogeochem. Cycle 22, http://dx.doi.org/10.1029/2007GB002952 (last access April 18, 2016).
Rees, R.M., Wuta, M., Furley, P.A., Li, C., 2006. Nitrous oxide fluxes from savanna (miombo) woodlands in Zimbabwe. J. Biogeogr. 33 (3), 424–437.
Reicosky, D.C., Gesch, R.W., Wagner, S.W., Gilbert, R.A., Wente, C.D., Morris, D.R., 2008. Tillage and wind effects on soil CO_2 concentrations in muck soils. Soil Tillage Res. 99, 221–231.
Reimann, C., Matschullat, J., Birke, M., Salminen, R., 2009. Arsenic distribution in the environment: the effects of scale. Appl. Geochem. 24 (7), 1147–1167.
Reimann, C., Matschullat, J., Birke, M., Salminen, R., 2010. Antimony in the environment: lessons from geochemical mapping. Appl. Geochem. 25 (2), 175–198.
Reimann, C., Birke, M., Demetriades, A., Filzmoser, P., O'Connor, P. (Eds.), 2014. Chemistry of Europe's Agricultural Soils. Part A and B. Geol. Jb. 102, 880 p.
Rella, C.W., Chen, H., Andrews, A.E., Filges, A., Gerbig, C., Hatakka, J., Karion, A., Miles, N.L., Richardson, S.J., Steinbacher, M., Sweeney, C., Wastine, B., Zellweger, C., 2013. High accuracy measurements of dry mole fractions of carbon dioxide and methane in humid air. Atmos. Meas. Tech. 6, 837–860.
Remde, A., Conrad, R., 1991. Role of nitrification and denitrification for NO metabolism in soil. Biogeochemistry 12, 189–205.
Robertson, G.P., 1989. Nitrification and denitrification in humid tropical ecosystems: potential controls on nitrogen retention. In: Mineral Nutrients in Tropical Forest and Savanna Ecosystems. Blackwell Scientific, Cambridge, Massachusetts, USA, pp. 55–69.
Rochette, P., Eriksen-Hamel, N.S., 2008. Chamber measurements of soil nitrous oxide flux: are absolute values reliable? Soil Sci. Soc. Am. J. 72, 331–342.
Rochette, P., Ellert, B., Gregorich, E.G., Desjardins, R.L., Pattey, E., Lessard, R., Johnson, B.G., 1997. Description of a dynamic closed chamber for measuring soil respiration and its comparison with other techniques. Can. J. Soil. Sci. 77, 195–203.

Rochette, P., 2011. Towards a standard non-steady-state chamber methodology for measuring soil N_2O emissions. Anim. Feed Sci. Tech. 166-167, 141–146.
Rösemann, C., Haenel, H.D., Poddey, E., Dämmgen, U., Döhler, H., Eurich-Menden, B., Laubach, P., Dieterle, M., Osterburg, B., 2011. Calculations of gaseous and particulate emissions from German agriculture 1990–2009. vTI Agric. For. Res. Spec., 389.
Russell, E.J., Appleyard, A., 1915. The atmosphere of the soil: gas composition and the causes of variation. J. Agric. Sci. 7, 43.
Ruuskanen, T.M., Müller, M., Schnitzhofer, R., Karl, T., Graus, M., Bamberger, I., Hörtnagl, L., Brilli, F., Wohlfahrt, G., Hansel, A., 2011. Eddy covariance VOC emission and deposition fluxes above grassland using PTR-TOF. Atmos. Chem. Phys. 11, 611–625.
Ryals, R., Silver, W.L., 2013. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. Ecol. Appl. 23 (1), 46–59.
Saggar, S., Tate, K.R., Giltrap, D.L., Singh, J., 2008. Soil-atmosphere exchange of nitrous oxide and methane in New Zealand terrestrial ecosystems and their mitigation options: a review. Plant Soil 309, 25–42.
Sahrawat, K.L., 2004. Terminal electron acceptors for controlling methane emissions from submerged rice soils. Commun. Soil Sci. Plant Anal. 35, 1401–1413.
Sainju, U.M., Jabro, J.D., Stevens, W.B., 2008. Soil carbon dioxide emission and carbon content as affected by irrigation, tillage, cropping system, and nitrogen fertilization. J. Environ. Qual. 37, 98–106.
Saiz, G., Byrne, K.A., Butterbach-Bahl, K., Kiese, R., Blujdea, V., Farrell, E.P., 2006. Stand age-related effects on soil respiration in a first rotation Sitka spruce chronosequence in central Ireland. Glob. Change Biol. 12, 1007–1020.
Sanhueza, E., Hao, W.M., Scharffe, D., Donoso, L., Crutzen, P.J., 1990. N_2O and NO emissions from soils of the northern part of the Guayana Shield, Venezuela. J. Geophys. Res.: Atmos. (1984–2012) 95, D13, 22481–22488.
Sanz-Cobena, A., Abalos, D., Meijide, A., Sanchez-Martin, L., Vallejo, A., 2014a. Soil moisture determines the effectiveness of two urease inhibitors to decrease N_2O emission. Mitig. Adapt. Strateg. Glob. Change, http://dx.doi.org/10.1007/s11027-014-9548-5 (last access April 18, 2016).
Sanz-Cobena, A., García-Marco, S., Quemada, M., Gabriel, J.L., Almendros, P., Vallejo, A., 2014b. Do cover crops enhance N_2O, CO_2 or CH_4 emissions from soil in Mediterranean arable systems? Sci. Total Environ. 466–467, 164–174.
Savage, K.E., Davidson, E.A., 2001. Interannual variation of soil respiration in two New England forests. Glob. Biogeochem. Cycles 15, 337–350.
Schach, R., 2015. Soil Respiration (CO_2) of an Agricultural Site in Winter (Spohla, Saxony). Unpubl. B. Sc. Hon. Thesis. TU Bergakademie, Freiberg.
Schaufler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton, M.A., Zechmeister-Boltenstern, S., 2010. Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. Eur. J. Soil Sci. 61, 683–696.
Schindlbacher, A., Zechmeister-Boltenstern, S., Butterbach-Bahl, K., 2004. Effects of soil moisture and temperature on NO, NO_2, and N_2O emissions from European forest soils. J. Geophys. Res. D: Atmos. 109 (D17), http://dx.doi.org/10.1029/2004JD004590 (last access April 18, 2016).
Schlesinger, W.H., Andrews, J.A., 2000. Soil respiration and the global carbon cycle. Biogeochemistry 48, 7–20.
Schmidt, J., Seiler, W., Conrad, R., 1988. Emission of nitrous oxide from temperate forest soils into the atmosphere. J. Atmos. Chem. 6 (1–2), 95–115.
Schneising, O., Buchwitz, M., Burrows, J.P., Bovensmann, H., Reuter, M., Notholt, J., Macatangay, R., Warneke, T., 2008. Three years of greenhouse gas column-averaged dry air mole fractions retrieved from satellite—part 1: carbon dioxide. Atmos. Chem. Phys. 8, 3827–3853.
Schulze, E.D., Luyssaert, S., Ciais, P., Freibauer, A., Janssens, I.A., Soussana, J.F., Smith, P., Grace, J., Levin, I., Thiruchittampalam, B., Heimann, M., Dolman, A.J., Valentini, R., Bousquet, P., Peylin, P., Peters, W., Rödenbeck, C., Etiope, G., Vuichard, N., Wattenbach, M., Nabuurs, G.J., Poussi, Z., Nieschulze, J., Gash, J.H., CarboEurope Team, 2009. Importance of methane and nitrous oxide for Europës terrestrial greenhouse-gas balance. Nat. Geosci. 2, 842–850.
Schulze, E.D., Ciais, P., Luyssaert, S., Schrumpf, M., Janssens, I.A., Thiruchittampalam, B., Thelokes, J., Saurat, M., Bringezu, S., Lelieveld, J., Lohila, A., Rebmann, C., Jung, M., Bastviken, D., Abril, G., Grassi, G., Leip, A., Freibauer, A., Kutsch, W., Don, A., Nieschulze, J., Börner, A., Gash, J.H., Dolman, A.J., 2010. The European carbon balance. Part 4: integration of carbon and trace-gas fluxes. Glob. Change Biol. 16, 1451–1469.
Schwenke, G.D., Herridge, D.F., Scheer, C., Rowlings, D.W., Haigh, B.M., McMullen, K.G., 2015. Soil N_2O emissions under N_2-fixing legumes and N-fertilised canola: a reappraisal of emissions factor calculations. Agric. Ecosyst. Environ. 202, 232–242.
Shi, W.Y., Yan, M.J., Zhang, J.G., Guan, J.H., Du, S., 2014. Soil CO_2 emissions from five different types of land use on the semiarid Loess Plateau of China, with emphasis on the contribution of winter soil respiration. Atmos. Environ. 88, 74–82.
Shoji, S., Delgado, J., Mosier, A., Miura, Y., 2001. Use of controlled release fertilizers and nitrification inhibitors to increase nitrogen use efficiency and to conserve air and water quality. Commun. Soil Sci. Plant Anal. 32, 1051–1070.
Šimek, M., Hynšt, J., Šimek, P., 2014. Emissions of CH_4 CO_2, and N_2O from soil at a cattle overwintering area as affected by available C and N. Appl. Soil Ecol. 75, 52–62.
Simpson, I.J., Edwards, G.C., Thurtell, G.W., den Hartog, H., Neumann, H.H., Staebler, R.M., 1997. Micrometeorological measurements of methane and nitrous oxide exchange above a boreal aspen forest. J. Geophys. Res. 102 (D24), 29331–29341.
Six, J., Feller, C., Denef, K., Ogle, S.M., Moraes Sa, J.C., Albrecht, de, 2002. Soil organic matter, biota and aggregation in temperate and tropical soils—effects of no-tillage. Agronomy 22, 755–775.
Six, J., Ogle, S.M., Jay Breidt, F., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. Glob. Change Biol. 10, 155–160.
Sjögersten, S., Black, C.R., Evers, S., Hoyos-Santillan, J., Wright, E.L., Turner, B.L., 2015. Tropical wetlands: a missing link in the global carbon cycle? Glob. Biogeochem. Cycle 28, 1371–1386.
Skiba, U., Drewer, J., Tang, Y.S., van Dijk, N., Helfter, C., Nemitz, E., Famulari, D., Cape, J.N., Jones, S.K., Twigg, M., Pihlatie, M., Vesala, T., Larsen, K.S., Carter, M.S., Ambus, P., Ibrom, A., Beier, C., Hensen, A., Frumau, A., Erisman, J.W., Brüggemann, N., Gasche, R., Butterbach-Bahl, K., Neftel, A., Spirig, C., Horvath, L., Freibauer, A., Cellier, P., Laville, P., Loubet, B., Magliulo, E., Bertolini, T., Seufert, G., Andersson, M., Manca, G., Laurila, T., Aurela, M., Lohila, A., Zechmeister-Boltenstern, S., Kitzler, B., Schaufler, G., Siemens, J., Kindler, R., Flechard, C., Sutton, M.A., 2009. Biosphere–atmosphere exchange of reactive nitrogen and greenhouse gases at the NitroEurope core flux measurement sites: measurement strategy and first data sets. Agric. Ecosyst. Environ. 133, 139–149.
Smart, D.R., Ritchie, K., Stark, J.M., Bugbee, B., 1997. Evidence that elevated CO_2 levels can indirectly increase rhizosphere denitrifier activity? Appl. Environ. Microbiol. 63, 4621–4624.
Smith, K.A., Dobbie, K.E., Ball, B.C., Bakken, L.R., Situala, B.K., Hansen, S., Brumme, R., Borken, W., Christensen, S., Prieme, A., Fowler, D., Macdonald, J.A., Skiba, U., Klemedtsson, L., Kasimir-Klemedtssson, A., Degorska, A., Orlanski, P., 2000. Oxidation of atmospheric methane in Northern European soils, comparison with other ecosystems, and uncertainties in the global terrestrial sink. Glob. Change Biol. 6, 793–803.
Smith, K.A., Ball, T., Conen, F., Dobbie, K.E., Massheder, J., Rey, A., 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54, 779–791.
Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric. Ecosyst. Environ. 133 (3–4), 247–266.
Søvik, A.K., Augustin, J., Heikkinen, K., Huttunen, J.T., Necki, J.M., Karjalainen, S.M., Kløve, B., Liikanen, A., Mander, Ü., Puustinen, M., Teiter, S., Wachniew, P., 2006. Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. J. Environ. Qual. 35 (6), 2360–2373.
Sponseller, R.A., 2007. Precipitation pulses and soil CO_2 flux in a Sonoran Desert ecosystem. Glob. Change Biol. 13, 426–436.
Stehfest, E., Bouwman, L., 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosyst. 74 (3), 207–228.
Stoyan, H., De-Polli, H., Böhm, S., Robertson, G.P., Paul, E.A., 2000. Spatial heterogeneity of soil respiration and related properties at the plant scale. Plant Soil 222, 203–214.
Suleau, M., Debacq, A., Dehaes, V., Aubinet, M., 2009. Wind velocity perturbation of soil respiration measurements using closed dynamic chambers. Eur. J. Soil Sci. 60, 515–524.
Tang, J., Baldocchi, D.D., Qi, Y., Xu, L., 2003. Assessing soil CO_2 efflux using continuous measurements of CO_2 profiles in soils with small solid-state sensors. Agric. For. Meteorol. 118, 207–220, http://dx.doi.org/10.1016/S0168-1923(03)00112-6 (last access April 18, 2016).
Teiter, S., Mander, Ü., 2005. Emission of N_2O, N_2 CH_4, and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones. Ecol. Eng. 25, 528–541.
Teixeira, D.D.B., Bicalho, E.S., Cerri, C.E.P., Panosso, A.R., Pereira, G.T., La Scala, N., 2013. Quantification of uncertainties associated with space-time estimates of short-term soil CO_2 emissions in a sugar cane area. Agric. Ecosyst. Environ. 167 (1), 33–37, http://dx.doi.org/10.1016/j.agee.2013.01.004 (last access April 18, 2016).
Tellez-Rio, A., García-Marco, S., Navas, M., López-Solanilla, E., Rees, R.M., Tenorio, J.L., Vallejo, A., 2014. Nitrous oxide and methane emissions from a vetch-cropping season are changed by long-term tillage practices in a Mediterranean agroecosystem. Biol. Fertil. Soils 51 (1), 77–88.
Tenuta, M., Beauchamp, E.G., 2003. Nitrous oxide production from granular nitrogen fertilizers applied to a silt loam soil. Can. J. Soil Sci. 83, 521–532.
Thomas, A.D., Hoon, S.R., Dougill, A.J., 2011. Soil respiration at five sites along the Kalahari Transect: effects of temperature, precipitation pulses and biological soil crust cover. Geoderma 167–168, 284–294.
Turetsky, M.R., Kotowska, A., Bubier, J., Dise, N.B., Crill, P., Hornibrook, E.R.C., Minkkinen, K., Moore, T.R., Myers-Smith, I.H., Nykänen, H., Olefeldt, D., Rinne, J., Saarnio, S., Shurpali, N., Tuittila, E.S., Waddington, J.M., White, J.R., Wickland, K.P., Wilmking, M., 2014. A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. Glob. Change Biol. 20, 2183–2197, http://dx.doi.org/10.1111/gcb.12580 (last access April 18, 2016).
UBA, 2013. Treibhausgasneutrales Deutschland im Jahr 2050. Hintergrundbericht, 32 pp. http://www.uba.de/publikationen/treibhausgasneutrales-deutschland-im-jahr-2050 (last access April 18, 2016).
Urbanová, Z., Picek, T., Hájek, T., Bufková, I., Tuittila, E.S., 2012. Vegetation and carbon gas dynamics under a changed hydrological regime in central European peatlands. Plant Ecol. Divers. 5 (1), 89–103, http://dx.doi.org/10.1080/17550874.2012.688069 (last access April 18, 2016).

Urbanski, S., Barford, C., Wofsy, S., Kucharik, C., Pyle, E., Budney, J., McKain, J., Fitzjarrald, D., Czikowsky, M., Munger, J.W., 2007. Factors controlling CO_2 exchange on timescales from hourly to decadal at Harvard Forest. J. Geophys. Res. 112 (G02020), http://dx.doi.org/10.1029/2006JG000293 (last access April 18, 2016).

Ussiri, D., Lal, R., 2013. Soil Emission of Nitrous Oxide and Its Mitigation. Springer, Dordrecht, 378 pp.

van der Weerden, T.J., Kelliher, F.M., de Klein, C.A.M., 2010. Influence of pore size distribution and soil water content on nitrous oxide emissions. Soil Res. 50 (2), 125–135.

Velty, S., Augustin, J., Behrendt, A., Zeitz, J., 2007. Greenhouse gas fluxes during rewetting of peatlands by use of effluents—a lysimeter study. Arch. Agron. Soil Sci. 53, 629–643.

Venterea, R.T., Burger, M., Spokas, K.A., 2005. Nitrogen oxide and methane emissions under varying tillage and fertilizer management. J. Environ. Qual. 34, 1467–1477.

Venterea, R.T., Halvorson, A.D., Kitchen, N., Liebig, M.A., Cavigelli, M.A., Del Grosso, Stephen J., Motavalli, P.P., Nelson, K.A., Spokas, K.A., Singh, B.P., Stewart, C.E., Ranaivoson, A., Strock, J., Collins, H., 2012. Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. Front. Ecol. Environ. 10 (10), 562–570.

Verchot, L.V., Davidson, E.A., Cattânio, H., Ackerman, I.L., 1999a. Land-use change and biogeochemical controls of methane fluxes in soils of eastern Amazonia. Ecosystems 3 (1), 41–56.

Verchot, L.V., Davidson, E.A., Cattânio, H., Ackerman, I.L., Erickson, H.E., Keller, M., 1999b. Land use change and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia. Glob. Biogeochem. Cycles 13 (1), 31–46.

Verhoeven, J.T.A., Setter, T.L., 2010. Agricultural use of wetlands: opportunities and limitations. Ann. Bot. 105, 155–163.

Vermeulen, S.J., Campbell, B.M., Ingram, J.S.I., 2012. Climate change and food systems. Ann. Rev. Environ. Resour. 37, 195–222.

Vesterdal, L., Elberling, B., Christiansen, J.R., Callesen, I., Schmidt, I.K., 2012. Soil respiration and rates of soil carbon turnover differ among six common European tree species. For. Ecol. Manage. 264, 185–196.

Virkajärvi, P., Maljanen, M., Saarijärvi, K., Haapala, J., Martikainen, P., 2010. N_2O emissions from boreal grass and grass–clover pasture soils. Agric. Ecosyst. Environ. 137, 59–67.

Waletzko, E.J., Mitsch, W.J., 2013. Methane emissions from wetlands: an in situ side-by-side comparison of two static accumulation chamber designs. Ecol. Eng. 72, 95–102, http://dx.doi.org/10.1016/j.ecoleng.2013.09.008 (last access April 18, 2016).

Wang, K., Liu, C., Zheng, X., Pihlatie, M., Li, B., Haapanala, S., Vesala, T., Liu, H., Wang, Y., Liu, G., Hu, F., 2013a. Comparison between eddy covariance and automatic chamber techniques for measuring net ecosystem exchange of carbon dioxide in cotton and wheat fields. Biogeoscience 10, 6865–6877.

Wang, Y.Y., Hu, C.S., Ming, H., Zhang, Y.M., Li, X.X., Dong, W.X., Oenema, O., 2013b. Concentration profiles of CH_4 CO_2 and N_2O in soils of a wheat–maize rotation ecosystem in North China Plain, measured weekly over a whole year. Agric. Ecosyst. Environ. 164, 260–272.

Wang, B., Zha, T.S., Jia, X., Wu, B., Zhang, Y.Q., Qin, S.G., 2014. Soil moisture modifies the response of soil respiration to temperature in a desert shrub ecosystem. Biogeoscience 11, 259–268.

Wei, S., Zhang, X., McLaughlin, N.B., Liang, A., Jia, S., Chen, X., Chen, X., 2014. Effect of soil temperature and soil moisture on CO_2 flux from eroded landscape positions on black soil in Northeast China. Soil Tillage Res. 144 (1), 119–125, http://dx.doi.org/10.1016/j.still.2014.07.012 (last access April 18, 2016).

Weiß, L., 2011. CO_2-Flüsse rekultivierter Böden am Beispiel von Braunkohletagebaugebieten in Sachsen. Unpubl. M. Sc. Thesis. TU Bergakademie, Freiberg.

Weslien, P., Kasimir Klemedtsson, Å., Börjesson, G., Klemedtsson, L., 2009. Strong pH influence on N_2O and CH_4 fluxes from forested organic soils. Eur. J. Soil Sci. 60, 311–320.

Whalen, S.C., 2005. Biogeochemistry of methane exchange between natural wetlands and the atmosphere. Environ. Eng. Sci. 22, 73–94.

Willson, R., Brown, K., 2008. Carbon neutrality at the local level: achievable goal or fantasy? J. Am. Plan. Assoc. 74 (4), 497–504.

Winiwarter, W., 2007. National greenhouse gas inventories: understanding uncertainties versus potential for improving reliability. Water Air Soil Pollut. 7, 443–450.

Wolf, B., Zheng, X., Brüggemann, N., Chen, W., Dannenmann, M., Han, X., Sutton, M.A., Wu, H., Yao, Z., Butterbach-Bahl, K., 2010. Grazing-induced reduction of natural nitrous oxide release from continental steppe. Nature 464, 881–884, http://dx.doi.org/10.1038/nature08931 (last access April 18, 2016).

Wu, H., Lu, X., Wu, D., Song, L., Yan, X., Liu, J., 2013. Ant mounds alter spatial ad temporal patterns of CO_2, CH_4 and N_2O emissions from a marsh soil. Soil Biol. Biochem. 57, 884–891, http://dx.doi.org/10.1016/j.soilbio.2012.10.034 (last access April 18, 2016).

Xu, L., Furtaw, M.D., Madsen, R.A., Garcia, R.L., Anderson, D.J., McDermitt, D.K., 2006. On maintaining pressure equilibrium between a soil CO_2 flux chamber and the ambient air. J. Geophys. Res. 111 (D08), S10.

Yan, C., Zhang, H., Li, B., Wang, D., Zhao, Y., Zheng, Z., 2012. Effects of influent C/N ratios on CO_2 and CH_4 emissions from vertical subsurface flow constructed wetlands treating synthetic municipal wastewater. J. Hazard. Mater. 203–204, 188–194, http://dx.doi.org/10.1016/j.jhazmat.2011.12.002 (last access April 18, 2016).

Yan, G., Zheng, X., Cui, F., Yao, Z., Zhou, Z., Deng, J., Xu, Y., 2013. Two-year simultaneous records of N_2O and NO fluxes from a farmed cropland in the northern China plain with a reduced nitrogen addition rate by one-third. Agric. Ecosyst. Environ. 178, 39–50.

Yao, Z., Zheng, X., Xie, B., Mei, B., Wang, R., Butterbach-Bahl, K., Zhu, J., Yin, R., 2009. Tillage and crop residue management significantly affects N-trace gas emissions during the non-rice season of a subtropical rice-wheat rotation. Soil Biol. Biochem. 41, 2131–2140.

Yao, Z., Wolf, B., Chen, W., Butterbach-Bahl, K., Brüggemann, N., Wiesmeier, M., Dannenmann, M., Blank, B., Zheng, X., 2010. Spatial variability of N_2O, CH_4 and CO_2 fluxes within the Xilin River catchment of Inner Mongolia, China: a soil core study. Plant Soil 331, 341–359, http://dx.doi.org/10.1007/s11104-009-0257-x (last access April 18, 2016).

Yim, M.H., Joo, S.J., Nakane, K., 2002. Comparison of field methods for measuring soil respiration: a static alkali absorption method and two dynamic closed chamber methods. For. Ecol. Manage. 170, 189–197.

Yoshida, Y., Ota, Y., Eguchi, N., Kikuchi, N., Nobuta, K., Tran, H., Morino, I., Yokota, T., 2011. Retrieval algorithm for CO_2 and CH_4 column abundances from short-wavelength infrared spectral observations by the Greenhouse gases observing satellite. Atmos. Meas. Tech. 4, 717–734.

Yu, L., Wang, H., Wang, G., Song, W., Huang, Y., Li, S., Liang, N., Tang, Y., He, J., 2013. A comparison of methane emission measurements using eddy covariance and manual and automated chamber-based techniques in Tibetan Plateau alpine wetland. Environ. Pollut. 181, 81–90.

Zhang, L., Chen, Y., Li, W., Zhao, R., 2009. Abiotic regulators of soil respiration in desert ecosystems. Environ. Geol. 57, 1855–1864.

Zhang, Q., Lei, H.M., Yang, D.W., 2013. Seasonal variations in soil respiration, heterotrophic respiration and autotrophic respiration of a wheat and maize rotation cropland in the North China Plain. Agric. For. Meteorol. 180, 34–43.

Zhang, X., Yin, S., Li, Y., Zhuang, H., Li, C., Liu, C., 2014. Comparison of greenhouse gas emissions from rice paddy fields under different nitrogen fertilization loads in Chongming Island, Eastern China. Sci. Total Environ. 472, 381–388.

Zhou, M., Zhu, B., Butterbach-Bahl, K., Zheng, X., Wang, T., Wang, Y., 2013. Nitrous oxide emissions and nitrate leaching from a rain-fed wheat-maize rotation in the Sichuan Basin, China. Plant Soil 362, 149–159.

Zhu, R., Ma, D., Xu, H., 2014. Summertime N_2O, CH_4 and CO_2 exchanges from a tundra marsh and an upland tundra in maritime Antarctica. Atmos. Environ. 83, 269–281.

Zurba, K., Matschullat, J., 2015. Short rotation forestry (SRF) versus rapeseed plantations: insights from soil respiration and combustion heat per area. Energy Procedia 76, 398–405.