Carbon dioxide emissions from a septic tank soakaway in a northern maritime climate

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Abstract

Here, we present the first attempt to quantify long-term and diurnal variations of CO_2 fluxes from a soakaway of an on-site wastewater treatment system serving a single house located in a northern maritime climate (Ireland). An automated soil gas flux chamber system was deployed semi-continuously over a period of 17 months, recording hourly flux measurements from the soakaway (F_{soak}) and a control site ($F_{control}$). Soil gas fluxes expressed seasonal and diurnal variations: F_{soak} and $F_{control}$ ranged from 0.43 to 100.26 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$ and 0.45 to 19.92 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$ with median fluxes of 6.86 and 5.05 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$, respectively. While temperature, soil water content, and atmospheric pressure were identified as the most significant environmental factors correlated to the release of CO_2 from the control site, fluxes from the soakaway showed weaker correlations in regard to environmental factors. Assuming homogeneous spatial flux distributions, the soakaway emitted 15.0 $kg\,yr^{-1}$ more CO_2 into the atmosphere in total compared to a similarly sized control site.

Keywords:
anthropogenic greenhouse gas emissions, Ireland, on-site wastewater treatment system, sanitation, soil flux chamber

1. Introduction

On-site septic tank systems are a common choice for treating domestic wastewater in areas not connected to a centralized or decentralized wastewater treatment system. In the European Economic Area (EEA) a total of 23 % of households are estimated to use on-site wastewater treatment and disposal (EEA, 2013). In particular, regions with a relatively low population density and a significant share of dispersed settlements tend to rely more on on-site septic systems as means for domestic wastewater treatment and disposal; e. g. in Ireland, where 38 % of the population lives in rural areas, nearly 30 % of households treat their wastewater on site (CSO, 2011). In the Nordic Countries about 34 % of the population is connected to an on-site treatment or collection facility (Norin and Tideström, 2003) while in the United States about one fifth of the population relies on septic systems (USEPA, 2016).

Globally, there has been a shift from regarding sanitation infrastructure merely as a service for the provision of basic needs towards a more comprehensive implementation and promotion of long-term environmentally sustainable decentralized and on-site treatment systems (Massoud et al., 2009; Rosenqvist et al., 2016), particularly in regions currently underserved with basic services provision (Libralato et al., 2012; Parkinson and Tayler, 2003). Currently, with 64 % of the urban population in low- and middle-income coun-

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tries using on-site systems (Hawkins et al., 2014) and an estimated total of 2.4 billion people still lacking access to basic sanitation services globally (UNICEF/WHO, 2015) the total number of installed septic system is likely to rise in the future.

A conventional domestic septic system consists of two components; a septic tank (ST) and a soil dispersal system (SDS). The ST facilitates the initial collection and storage of the raw sewage from one or several households and allows the retention of settleable solids as sludge at the bottom of the tank and flocculent waste as a floating scum layer. While the settled solids are partially anaerobically digested within the tank, the effluent is ideally discharged into the vadose zone via an engineered SDS (e. g. soakaways, percolation trenches/leach fields, mound soil systems, drip line systems).

In Ireland, septic tank systems installed before 1991 had their effluent released into the soil mainly via soakaways (pits back-filled with stone or rubble for effluent disposal). In 1991, the National Standards Authority of Ireland recommended the construction of septic tank systems with larger percolation areas where the effluent would be distributed through percolation trenches – also known as leach fields, drainfields, or infiltration areas. However, of the current more than 450,000 installed on-site systems in Ireland, it is estimated that still approximately 65 % remain constructed before the implementation of these revised guidelines (CSO, 2011).

Microbial processes in the ST mainly follow anaerobic pathways for the degradation of organic matter via hydrolysis, acidogenesis, and methanogenesis resulting in the production of CH_4 and CO_2 gas. In the SDS a clogging zone forms at the infiltrative surface for the liquid ST effluent over time. Initial clogging occurs due to an accumulation of suspended solids, organic matter, and chemical precipitation resulting in potentially saturated conditions and ponding of effluent at the infiltrative surface (Beach and McCray, 2003). The increasing impedance to flow, in turn, allows for the formation of a mature biological clogging zone – also known as a microbial biomat – providing significant treatment and attenuation of contaminants before the wastewater reaches the underlying groundwater aquifer (Gill et al., 2007). Bacteria forming the biomat utilize an efficient defense mechanism, producing extracellular polymeric substances (EPS) to create anaerobic microenvironments to protect their bacterial cells. EPS have been characterized as containing significant concentrations of humic substances and polysaccharides and can cause soil clogging leading to lower infiltration rates (McKinley and Siegrist, 2010). Site specific parameters such as system design, mineral subsoil composition, subsoil permeability, hydraulic and organic loading rates, and further environmental factors such as soil temperature and rainfall patters influence the extend and microbial composition of the clogging zone (Beach and McCray, 2003; Gill et al., 2009; Winstanley and Fowler, 2013). A continuously fed biomat acts as a potential source of CO_2, CH_4, and N_2O emissions from transformation and degradation of organic and inorganic contaminants in the soil.

Considering the large number of on-site septic systems in use internationally, potentially constituting a significant source of GHG emissions, there has been a surprising lack of direct field measurements of these fluxes to the atmosphere. Most of the existing septic system emission models rely on load-based calculations or estimated emission factors. The IPCC provides guidelines on national GHG inventories following an organic load-based approach to estimate septic system emissions (IPCC, 2006). These guidelines only consider CH_4 emissions from anaerobic degradation in STs, disregarding the potential emissions from microbial degradation processes in the SDS. Direct CO_2 emissions from septic systems are omitted in the GHG inventories as they are of biogenic origin.

Numerous recent studies on septic systems mainly focused on the attenuation of chemical and biological pollutants and the risk for contamination of groundwater (Gill et al., 2007; Godfrey et al., 2007; Katz et al., 2010; Keegan et al., 2014), wells (Hynds et al., 2014; Lu et al., 2008), or surface waters (Dubber et al., 2016; Ockenden et al., 2014; Rosario et al., 2014; Withers et al., 2012) from septic systems. However, there is a limited number of studies with a scope on quantifying gas emissions from septic systems. Kinnicutt et al. (1910) reported gas emissions of $39\,\mathrm{L\,m^{-3}}$ treated sewage, of which 75.2 % were CH_4 and 5.9 % were CO_2, from a closed municipal septic tank in Worcester (MA, USA) fed with sewage from domestic and industrial sources. More recent studies by Leverenz et al. (2010) and Diaz-Valbuena et al. (2011) identified, for the first time, emissions of all three major GHGs (CH_4, CO_2, and N_2O) from eight septic systems and two SDSs in CA, USA. The studies noted that the septic tank itself would be the primary source of CH_4 emissions while most of the CO_2 is emitted from the SDS with negligible overall N_2O emissions. Emissions from direct flux measurements over the SDSs were deemed negligible. In the latest study, Truhlar et al. (2016) quantified CH_4, CO_2, and N_2O emissions from SDSs, sand filters, and vents for a period of three months at eight septic systems in NY, USA. While the majority of GHG emissions escaped through the roof vent, interpreted as proxy for direct emissions from the ST surface itself, the SDS was found to be a negligible source of CO_2 but potentially releases N_2O .

The existing studies on soil gas fluxes from septic systems have a limited temporal (September to December 2009 in Diaz-Valbuena et al. (2011), June to August 2014 in Truhlar et al. (2016)) and spatial span, thus, not being able to fully capture the expected seasonal variability of the emission rates.

Here, we are presenting the first attempt of measuring the CO_2 soil flux from a soakaway and a control area semi-continuously using an automated in-situ flux chamber measurement technique with hourly measurements over a 17-month period. The objective of this study was (i) to quantify the CO_2 emissions of a soakaway receiving domestic septic tank effluent under a normal load in order to detect potential seasonal and diurnal variations, and (ii) to identify potential environmental factors that drive the release of CO_2 over such a system.

2. Materials and methods

2.1. Study site

CO_2 soil flux measurements were conducted at an on-site wastewater treatment system receiving effluent from a single house in Co. Westmeath, Ireland (N53°24′ W7°30′). The system, consisting of a single-chamber septic tank and subsequent soakaway, was constructed more than 20 years ago. The septic tank has a total capacity of $2.6\,\mathrm{m^3}$ with a theoretical hydraulic retention time of 7 d and is fed by a single gravity flow effluent pipe from the household with a fluctuating number of occupants averaging 2. The septic tank was last desludged 3 years before the start of this study. The subsequent soakaway distributes an average $360\,\mathrm{l\,d^{-1}}$ of effluent over a total area of approximately $6\,\mathrm{m^2}$.

Co. Westmeath lies in central Ireland and its climate is classified as maritime Cfb (warm temperate, fully humid, warm summer) after Köppen-Geiger (Kottek et al., 2006). The mean annual temperature is 10 °C with mean seasonal minimum and maximum between 4 °C and 15 °C in winter and summer, respectively; average annual rainfall is 1200 mm without significant seasonal variation and 1250 h total annual sunshine (Walsh, 2012). The research site is a homogeneously vegetated grassland previously used for cattle grazing lying in an area of high groundwater vulnerability and is characterized by high permeability, gravelly SAND subsoil according to British Standard classification (BSI, 1999), consisting of till derived mainly from limestone.

The soakaway had previously been the subject of an intensive study which aimed to delineate the effluent plume and determine pollutant attenuation (Gill et al., 2015). It had been instrumented with soil moisture samplers (suction cup lysimeters) throughout its depth. The effluent from the septic tank was found to move rapidly through the subsoil in a predominantly vertical direction following the overall gradient of the site. Falling-head percolation tests yielded a field saturated conductivity K_{fs} of $1.39\,\mathrm{m\,d^{-1}}$. There was little evidence of the effluent plume extending laterally (apart from along the topographic gradient), consistent with the free draining subsoil characteristics. Despite the high permeability of the site, most of the organic and phosphorus concentration in the effluent was reduced within the biologically active zone of the soakaway which develops at the infiltrative surface over time. Equally, the unsaturated conditions in the soil gave rise to high levels of nitrification which occurred rapidly within the receiving subsoil, with 98 % of the inorganic-N present as NO_3-N indicating a potential source of N_2O emissions (Gill et al., 2015).

2.2. Soil gas flux measurement

Long-term soil gas flux measurements of CO_2 were carried out using an automated cylindrical chamber system (LI-8100A Automated Soil Gas Flux System, LI-COR Biosciences, Inc.) consisting of a non-dispersive infrared gas analyzer, a multiplexer, and two automated opaque long-term chambers (LI8100-104, LI-COR Biosciences, Inc.).

For each measurement cycle, the chambers were automatically moved over and lowered onto a PVC soil collar ($317.8\,\mathrm{cm}^2$) permanently inserted approximately 5 cm into the soil to prevent lateral diffusion of CO_2 and thus, creating a closed gas loop between chamber and analyzer in order to monitor the change of CO_2 concentration inside the chamber (Hutchinson and Livingston, 2001; LI-COR, 2012). Atmospheric pressure (p_{air}) and temperature (T_{air}) were recorded continuously by the soil gas flux system during the measurements.

The system was deployed semi-continuously between February 2015 and June 2016 with hourly measurements using a 30 s dead band and a 3 min measurement period, i. e. the soil surrounded by the collars was exposed to environmental conditions for 94 % of the time. The chambers were operated in sequence with one chamber placed over the soakaway approximately 2 m downslope from the septic tank outlet in an area identified by the previous monitoring with suction cup lysimeters to be within the effluent plume and the other chamber located several metres upslope of the septic tank as control. The locations of the collars were not changed between deployments. The initial 24 h of measurement after installing the soil collars and the initial 6 h of measurement after installing the chambers for each deployment period were discarded to eliminate effects of potential soil disturbance during installation (Bahn et al., 2009).

2.3. Flux estimation using non-linear regression

Soil CO_2 fluxes F [$\mu\mathrm{mol\,CO_2\,m^{-2}\,s^{-1}}$] were calculated using SoilFluxPro 4.0 (LI-COR Biosciences, Inc.), which implements a mass balance approach as

$$F = \frac{V_{\mathrm{cham}}\, p_0\, (1-\chi_{\mathrm{w}})}{\mathrm{R} s T_0} \cdot \frac{\partial \chi_{\mathrm{c}}(t)}{\partial t} \qquad (1)$$

using the volume of the chamber V_{cham} [$\mathrm{m^3}$], the atmospheric pressure at the beginning of the measurement p_0 [Pa], the chamber air water vapor mole fraction χ_{w} [$\mathrm{mol\,mol^{-1}}$], the universal gas constant R [$\mathrm{Pa\,m^3\,K^{-1}\,mol^{-1}}$], the soil collar surface area s [$\mathrm{m^2}$], the absolute temperature at the beginning of the measurement T_0 [K], and the initial change of chamber water vapor corrected CO_2 mole fraction $\partial\chi_{\mathrm{c}}/\partial t$ [$\mu\mathrm{mol\,mol^{-1}\,s^{-1}}$] (LI-COR, 2012). $\chi_{\mathrm{c}}(t)$ was calculated using an empirical exponential regression model which is fit to the measured CO_2 concentration data

$$\chi_{\mathrm{c}}(t) = (\chi_0 - \chi_{\mathrm{x}}) \exp\left[-a\,(t-t_0)\right] + \chi_{\mathrm{x}} \qquad (2)$$

with the initial water vapor corrected CO_2 mole fraction χ_0 [$\mathrm{mol\,mol^{-1}}$], and fit parameters χ_{x} [$\mathrm{mol\,mol^{-1}}$] and a [$\mathrm{s^{-1}}$] (LI-COR, 2012). The fit is used to derive the initial change of chamber water vapor corrected CO_2 mole fraction as the slope of the fit at the time of chamber closure. A non-linear model was chosen in order to reduce the influence of chamber feedback due to increasing resistance to naturally occurring diffusion as the main driver for transport of CO_2 from the soil to the atmosphere following the increase of CO_2 mole fraction inside the chamber during measurement (Davidson et al., 2002; Healy et al., 1996; Kutzbach et al., 2007). For each measurement, the R^2-value of the regressions was used as a quality control parameter and measurements with $R^2 < 0.9$ were rejected (less than 0.5% of total measurements).

To compare obtained flux values with previous studies, F_{c} was converted to per capita mass emission rates E_{cap} [$g\,CO_2\,cap^{-1}\,d^{-1}$]

$$E_{cap} = \frac{A_{soak} M_{CO_2}}{n} \cdot F_c \quad (3)$$

assuming a spatially uniform flux distribution and using the number of occupants n in the household, the surface area of the soakaway A_{soak}, and the molar mass M_{CO_2} of CO_2 [$44.01\,g\,mol^{-1}$] as normalization factors.

2.4. Environmental parameters

During each measurement, air temperature, atmospheric pressure, soil temperature at the surface layer adjacent to the chamber, and volumetric water content of the soil (Soil Moisture Probe EC-5, Decagon Devices, Inc.) at the surface layer (installation depth: 5 cm below soil surface) adjacent to the chamber were recorded. A tipping bucket rain gauge (Casella CEL, Inc.) was deployed from July 2015 until June 2016. Other meteorological parameters (wind speed and direction, relative humidity, and atmospheric pressure) were obtained from a weather station operated by the Irish Meteorological Service which was located 18 km away from the study site in Mullingar (N53°31′ W7°21′) (Met Éireann, 2016). The R package INSOL (Corripio, 2014) was used to compute daylight hours for the given latitude and dates during data processing.

2.5. Statistical analysis

All statistical analysis was performed with R version 3.2.3 (R Core Team, 2014). Welch's unequal variances t-tests were used to address the significance of whether observed CO_2 fluxes from the soakaway (F_{soak}) were different from the fluxes observed at the control site ($F_{control}$). Relationships between CO_2 fluxes and environmental variables were analyzed with multiple linear regression models; the relative importance of the predictors was calculated using the LMG method with bootstrap confidence intervals ($b = 1000$) and an averaging of the sequential sum-of-squares obtained from all possible orderings of the predictors, using the R package RELAIMPO (Grömping, 2006). All variables x were examined for evidence of normality and homogeneity and log-transformed [$\lg(x + 1)$] if needed to attain normality.

3. Results and Discussions

3.1. Temporal variation

CO_2 fluxes were measured over the soakaway (F_{soak}) and control site ($F_{control}$) semi-continuously, including 3709 hourly paired measurements. F_{soak} and $F_{control}$ ranged from 0.43 to 100.26 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$ and 0.45 to 19.92 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$ with medians of 6.86 and 5.05 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$, respectively. The normalized total yearly mean emissions over the soakaway were 57.1 kg CO_2 compared to 42.1 kg CO_2 for the undisturbed control, yielding net emissions of 15.0 kg $CO_2\,yr^{-1}$.

Overall, the calculated flux values showed a bi-modal distribution for both F_{soak} and $F_{control}$. Soakaway fluxes up to 20-times higher than the overall median were observed during periods of extremely high fluxes in October 2015 (Figure 1A). Over the control site the bi-modal distribution was controlled by soil temperature with an identified threshold of 10 °C (Figure 2). In order to ensure normal distribution of flux values for statistical analysis, data were therefore divided into four subgroups according to temporal (October and non–October fluxes) and temperature-related ($T_{soil} < 10$ °C and $T_{soil} \geq 10$ °C) selection rules. Despite seasonal variations in soil gas fluxes resulting in lower median fluxes from the soakaway compared to the control in April 2015, July 2015, and May 2016 (Figure 1A), Welch's two-sample t-tests for unequal variances confirmed that, overall, the mean soakaway fluxes were significantly higher than fluxes observed over the control site for each subgroup (Table 2).

Day-night soil gas flux variations (Figure 1B) were observed with higher total median fluxes during day time (soakaway: 7.62 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$; control: 5.77 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$) compared to night time (soakaway: 6.02 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$; control: 4.38 $\mu mol\,CO_2\,m^{-2}\,s^{-1}$). Welch's two-sample t-tests identified significantly higher mean day time fluxes in three of four subgroups (control fluxes at soil temperatures above and below 10 °C, and non–October soakaway fluxes), while during the October measurements over the soakaway mean day time fluxes were lower than mean night time fluxes (Table 3). Hence, monitoring soakaway fluxes during day time only could lead to potential overestimation of total emissions.

Comparing the diurnal pattern of soakaway and control soil gas fluxes, the median fluxes from the soakaway expressed, on average, an earlier increase and decrease starting from around 9 a. m. and 3 p. m. respectively (Figure 1C). The earlier increase in hourly median flux from the soakaway is presumed to be due to the morning use of water from the household resulting in partially treated wastewater from the ST to be discharge into the soakaway.

3.2. Impact of meteorological factors on carbon dioxide flux

Multilinear regression models were applied to the log-scaled flux data using the environmental parameters air temperature T_{air}, atmospheric pressure p_{air}, relative humidity ϕ_{rel}, water vapor corrected CO_2 mole fraction χ_{c}, soil temperature T_{soil}, soil moisture θ_{soil}, wind speed v_{wind}, and wind direction d_{wind} as continuous and rainfall P as binary input variables. A total of 132 measurements (i. e. 3.5 %) was discarded due to incorrect soil moisture data. October flux data were analyzed separately in order to attain a normal distribution for the data set.

The analysis shows that 33 and 81 % of the variation in soakaway and control fluxes, respectively, can be explained by environmental parameters (Figure 3). The relative importance of the predictors expressed similar patterns for both sites. While air and soil temperature had the highest relative importance, humidity, CO_2 mole fraction, wind speed, wind direction, and rain were insignificant. Soil moisture and atmospheric pressure were significant predictors only for the control fluxes.

In October, only 24 % of the variation in fluxes from the soakaway could be explained by environmental predictors and the distribution of their relative importance was distinctly different from the non–October measurements; soil moisture, CO_2 mole fraction, and soil temperature express the highest relative importance with relatively large 95 % bootstrap confidence intervals (Figure 3). Further regression models were tested for the flux data collected in October, but the environmental parameters recorded in this study were not able to explain the uniquely high fluxes.

To better understand the potential drivers of soil gas fluxes from the soakaway and control site, the most important predictors (responsive variance $> 5\,\%$) were further analyzed for potential correlation using a linear regression model and Pearson correlation coefficient r (Figure 4). Positive correlations were found between fluxes and both air and soil temperature as well as between F_{control} and p_{air} and a negative correlation was identified between F_{control} and θ_{soil}. However, no correlation was found for F_{soak}.

14 and 25 % of variation in F_{soak} can be explained with T_{soil} and T_{air}, respectively. For F_{control} this relation is reverse, with 64 and 42 % of the variation explained by T_{soil} and T_{air}, respectively. The regressions, for both soakaway and control fluxes related to air temperature had similar slopes with 0.34 ± 0.01 and $0.36 \pm 0.01\,\mathrm{\mu mol\,CO_2\,m^{-2}\,s^{-1}\,°C^{-1}}$, respectively (slope ± SE), resulting in similar flux increases with increasing air temperature. However, both F_{soak} and F_{control} expressed a higher sensitivity towards increasing soil temperature with 0.51 and $0.80\,\mathrm{\mu mol\,CO_2\,m^{-2}\,s^{-1}\,°C^{-1}}$, respectively. 10 and 42 % of F_{control} variation can be explained with p_{air} and θ_{soil}, respectively.

The seasonal and diurnal variations of fluxes appeared to mimic the change in air temperature and related soil temperature, as indicated by the high correlation between flux and temperature values. $\mathrm{F_{soak}}$ expressed a stronger correlation with T_{air} than T_{soil}.

Apart from heavy rainfall events, the soil at the control site is normally unsaturated. However, due to a continuous recharge with partially treated effluent from the ST, the vadose zone surrounding the soakaway is mostly close to or at saturated water content. In future studies, the exact hydraulic and organic loading rates of wastewater discharged into the SDS and vadose zone soil water content should be monitored in order to being able to derive more accurate correlations with non-environmental factors governing the production and release of GHGs from septic systems.

3.3. Comparison with previous studies

CO_2 flux rates were converted to per capita mass emission rates E_{cap} (Equation 3) and compared with previous studies (Table 3). The IPCC and USEPA consider CO_2 emissions from wastewater treatment as biogenic and, thus, do not included them into their guidelines for GHG estimations from on-site systems. Diaz-Valbuena et al. (2011) observed significant CO_2 emissions from the septic tank and venting system, and found negligible emissions from the leach field. Mean emissions from this study including all measurement showed good agreement with emission values estimated by Truhlar et al. (2016) who attempted to monitor CO_2 fluxes from SDSs, and as well as from the sand filters, and the vents. However, the mean flux value from our study, $155\,\mathrm{g\,CO_2\,cap^{-1}\,d^{-1}}$, is highly biased by the extremely high fluxes found in October 2015. Excluding the October data results in mean CO_2 emissions rates of $78\,\mathrm{g\,CO_2\,cap^{-1}\,d^{-1}}$, compared to the $602\,\mathrm{g\,CO_2\,cap^{-1}\,d^{-1}}$ from the October data alone.

3.4. Implications of the study

The semi-continuous measurement of CO_2 flux from a septic tank soakaway (F_{soak}) and control site (F_{control}) over a period of 17 months expressed seasonal and diurnal variations: F_{soak} and F_{control} ranged from 0.43 to $100.26\,\mathrm{\mu mol\,CO_2\,m^{-2}\,s^{-1}}$ and 0.45 to $19.92\,\mathrm{\mu mol\,CO_2\,m^{-2}\,s^{-1}}$ with median fluxes of 6.86 and $5.05\,\mathrm{\mu mol\,CO_2\,m^{-2}\,s^{-1}}$, respectively. This means, assuming a spatially homogeneous flux distribution, the soakaway emitted a total of $15.0\,\mathrm{kg\,yr^{-1}}$ more CO_2 into the atmosphere (soakaway emissions: $42.1\,\mathrm{kg\,CO_2\,yr^{-1}}$) compared to a similarly sized control site (control emissions: $57.1\,\mathrm{kg\,CO_2\,yr^{-1}}$).

However, this study was limited with respect to determining the spatial variation of CO_2 flux from the SDS due to the number of soil gas flux chambers available. Thus, it is not possible to conclude whether the fluxes measured over the soakaway were representative of the average total CO_2 fluxes resulting from the main plume of percolating effluent. Especially, the 20-fold higher median emissions observed in October (compared to all other monitoring months) could not be explained with the measured environmental parameters alone and give rise to the questions whether a shifting effluent plume in the vadose zone could result in spatially and temporarily highly variable soil gas flux emissions. It is possible that the area of maximum emissions is moving with the effluent plume and that the October measurements captured this phenomenon.

Future studies should (i) further investigate the spatial distribution of fluxes using a multiple chamber set-up to account for the potentially high spatial variability of GHG fluxes over a soakaway or other SDSs in order to be able to more accurately predict total mass emission rates per unit area from such systems, and (ii) monitor the soil moisture conditions of the system in question more closely and at different depths in order to establish a better understanding of the actual spatial and temporal variation of the effluent plume in the soil.

Among the recorded environmental parameters, atmospheric and soil temperature were the best predictors of CO_2 fluxes from the soakaway. For a more comprehensive understanding of the temporal variations of soil gas fluxes from septic systems, future studies should also record quantitative and qualitative parameters and the spatial distribution of the septic tank effluent released to the SDS.

To understand total septic system emissions, long-term integrated studies for simultaneous monitoring of major GHG emissions (CO_2, CH_4, and N_2O) are needed, including direct measurements on the ST surface and a variation of measuring points over the SDS to accurately capture the spatial and temporal heterogeneity of gas fluxes.

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Author contributions

L.G. conceived the study. C.S., J.K., and L.G. planned the experiments. C.S. and J.K. performed the experiments and analyzed data. C.S., J.K., and L.G. wrote the manuscript. All authors interpreted the results and reviewed the manuscript.

References

Bahn, M., Kutsch, W.L., Heinemeyer, A., 2009. Soil Carbon Dynamics – an Integrated Methodology. Cambridge: Cambridge Univ. Press.. chapter Synthesis: emerging issues and challenges for an integrated understanding of soil carbon fluxes. pp. 257–271.

Beach, D.N.H., McCray, J.E., 2003. Numerical modeling of unsaturated flow in wastewater soil absorption systems. Ground Water Monit. Remediat. 23, 64–72. doi:10.1111/j.1745-6592.2003.tb00672.x.

BSI, 1999. British Standards Institution BS 5930:1999 Code of practies for site investigations.

Corripio, J.G., 2014. insol: Solar Radiation. R package version 1.1.1. URL: https://CRAN.R-project.org/package=insol. last access: 20 October 2016.

CSO, 2011. This is Ireland – Highlights from Census 2011, Part 1. Central Statistics Office.

Davidson, E.A., Savage, K., Verchot, L.V., Navarro, R., 2002. Minimizing artifacts and biases in chamber-based measurements of soil respiration. Agr. Forest. Meteorol. 113, 21–37. doi:10.1016/s0168-1923(02)00100-4.

Diaz-Valbuena, L.R., Leverenz, H.L., Cappa, C.D., Tchobanoglous, G., Horwath, W.R., Darby, J.L., 2011. Methane, carbon dioxide, and nitrous oxide emissions from septic tank systems. Environ. Sci. Technol. 45, 2741–2747. doi:10.1021/es1036095.

Dubber, D., Brophy, L., Behan, P., Gill, L.W., 2016. Using faecal sterols and fluorescent whitening compounds to detect human faecal contamination from septic tanks in Irish catchments, in: 13th International Conference on Modelling, Monitoring and Management of Water Pollution (Wessex Institute), Venice, Italy.

EEA, 2013. Durostat database. URL: http://ec.europa.eu/eurostat/data/database. last access: 20 October 2016.

Gill, L.W., Dubber, D., O'Flaherty, V., Keegan, M., Kilroy, K., Curneen, S., Mistear, B., Johnston, P., Pilla, F., McCarty, T., Qazi, N., Smyth, D., 2015. Assessment of disposal options for treated wastewater from single houses in low-permeability subsoils. Technical Report 161. Environmental Protection Agency Ireland.

Gill, L.W., O'Luanaigh, N., Johnston, P.M., Misstear, B.D.R., O'Súlleabháin, C., 2009. Nutrient loading on subsoils from on-site wastewater effluent, comparing septic tank and secondary treatment systems. Water Res. 43, 2739–2749. doi:10.1016/j.watres.2009.03.024.

Gill, L.W., O'Súlleabháin, C., Misstear, B.D.R., Johnston, P.J., 2007. The treatment performance of different subsoils in Ireland receiving on-site wastewater effluent. J. Environ. Qual. 36, 1843–1855. doi:10.2134/jeq2007.0064.

Godfrey, E., Woessner, W.W., Benotti, M.J., 2007. Pharmaceuticals in on-site sewage effluent and ground water, Western Montana. Ground Water 45, 263–271. doi:10.1111/j.1745-6584.2006.00288.x.

Grömping, U., 2006. Relative importance for linear regression in R: The package relaimpo. J. Stat. Softw. 17. doi:10.18637/jss.v017.i01.

Hawkins, P., Blackett, I., Heymans, C., 2014. The Missing Link in Sanitation Service Delivery: A Review of Fecal Sludge Management in 12 cities. WSP Research Brief 88258. WSP, Washington, DC, USA.

Healy, R.W., Striegl, R.G., Russell, T.F., Hutchinson, G.L., Livingston, G.P., 1996. Numerical evaluation of static-chamber measurements of soil—atmosphere gas exchange: Identification of physical processes. Soil Sci. Soc. Am. J. 60, 740–747. doi:10.2136/sssaj1996.03615995006000030009x.

Hutchinson, G.L., Livingston, G.P., 2001. Vents and seals in non-steady-state chambers used for measuring gas exchange between soil and the atmosphere. Eur J Soil Science 52, 675–682. doi:10.1046/j.1365-2389.2001.00415.x.

Hynds, P., Misstear, B.D., Gill, L.W., Murphy, H.M., 2014. Groundwater source contamination mechanisms: Physicochemical profile clustering, risk factor analysis and multivariate modelling. J. Contam. Hydrol. 159, 47–56. doi:10.1016/j.jconhyd.2014.02.001.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.

Katz, B.G., Griffin, D.W., McMahon, P.B., Harden, H.S., Wade, E., Hicks, R.W., Chanton, J.P., 2010. Fate of effluent-borne contaminants beneath septic tank drainfields overlying a karst aquifer. J. Environ. Qual. 39, 1181–1195. doi:10.2134/jeq2009.0244.

Keegan, M., Kilroy, K., Nolan, D., Dubber, D., Johnston, P.M., Misstear, B.D.R., O'Flaherty, V., Barrett, M., Gill, L.W., 2014. Assessment of the impact of traditional septic tank soakaway systems on water quality in Ireland. Water Sci. Technol. 70, 634–641. doi:10.2166/wst.2014.227.

Kinnicutt, L.P., Winslow, C.E.A., Winthrop Pratt, R., 1910. Sewage Disposal. 1st ed., John Wiley & Sons,

New York, NY, USA.
Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World map of the Köppen-Geiger climate classification updated. Meteorol. Z. 15, 259–263. doi:10.1127/0941-2948/2006/0130.
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. Biogeosciences 4, 1005–1025. doi:10.5194/bg-4-1005-2007.
Leverenz, L.L., Tchobanoglous, G., Darby, J.L., 2010. Evaluation of Greenhouse Gas Emissions from Septic Systems. Technical Report. University of California, Davis.
LI-COR, 2012. LI-8100A Automated Soil CO_2 Flux System. LI-COR Biosciences, Inc.
Libralato, G., Ghirardini, A.V., Avezzù, F., 2012. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. J. Environ. Manage. 94, 61–68. doi:10.1016/j.jenvman.2011.07.010.
Lu, Y., Tang, C., Chen, J., Sakura, Y., 2008. Impact of septic tank systems on local groundwater quality and water supply in the Pearl River Delta, China: case study. Hydrol. Process. 22, 443–450. doi:10.1002/hyp.6617.
Massoud, M.A., Tarhini, A., Nasr, J.A., 2009. Decentralized approaches to wastewater treatment and management: Applicability in developing countries. J. Environ. Manage. 90, 652–659. doi:10.1016/j.jenvman.2008.07.001.
McKinley, J.W., Siegrist, R.L., 2010. Accumulation of organic matter components in soil under conditions imposed by wastewater infiltration. Soil Sci. Soc. Am. J. 74, 1690–1700. doi:10.2136/sssaj2009.0395.
Met Éireann, 2016. The Irish Meterological Service Online. URL: http://www.met.ie. last access: 20 October 2016.
Norin, E., Tideström, H., 2003. Septic tank sludge handling in the Nordic countries – consequences of a revised EC sludge directive on the future treatment and land use. TemaNORD 2003:560, Nordic Council of Ministers, Copenhagen, DK.
Ockenden, M.C., Quinton, J.N., Favaretto, N., Deasy, C., Surridge, B., 2014. Reduced nutrient pollution in a rural stream following septic tank upgrade and installation of runoff retention measures. Environ. Sci.: Process Impacts 16, 1637–1645. doi:10.1039/c3em00681f.
Parkinson, J., Tayler, K., 2003. Decentralized wastewater management in peri-urban areas in low-income countries. Environ. Urban. 15, 75–90. doi:10.1177/095624780301500119.
R Core Team, 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: http://www.R-project.org/. last access: 20 October 2016.
Rosario, K.L.D., Mitra, S., Humphrey, C.P., O'Driscoll, M.A., 2014. Detection of pharmaceuticals and other personal care products in groundwater beneath and adjacent to onsite wastewater treatment systems in a coastal plain shallow aquifer. Sci. Total Environ. 487, 216–223. doi:10.1016/j.scitotenv.2014.03.135.
Rosenqvist, T., Mitchell, C., Willetts, J., 2016. A short history of how we think and talk about sanitation services and why it matters. J. Water Sanit. Hyg. Dev. 6, 298–312. doi:10.2166/washdev.2016.118.
Truhlar, A.M., Rahm, B.G., Brooks, R.A., Nadeau, S.A., Makarsky, E.T., Todd Walter, M., 2016. Greenhouse gas emissions from septic systems in new york state. J. Environ. Qual. 45, 1153–1160. doi:10.2134/jeq2015.09.0478.
UNICEF/WHO, 2015. Progress on sanitation and drinking water – 2015 update and MDG assessment. UNICEF.
USEPA, 2016. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2014.
Walsh, S., 2012. A Summary of Climate Averages for Ireland 1981 - 2010. Technical Report. Met Eireann Climatological Note No. 14.
Winstanley, H.F., Fowler, A.C., 2013. Biomat development in soil treatment units for on-site wastewater treatment. Bull. Math. Biol. 75, 1985–2001. doi:10.1007/s11538-013-9881-y.
Withers, P.J.A., May, L., Jarvie, H.P., Jordan, P., Doody, D., Foy, R.H., Bechmann, M., Cooksley, S., Dils, R., Deal, N., 2012. Nutrient emissions to water from septic tank systems in rural catchments: Uncertainties and implications for policy. Environ. Sci.: Policy 24, 71–82. doi:10.1016/j.envsci.2012.07.023.

List of Figures

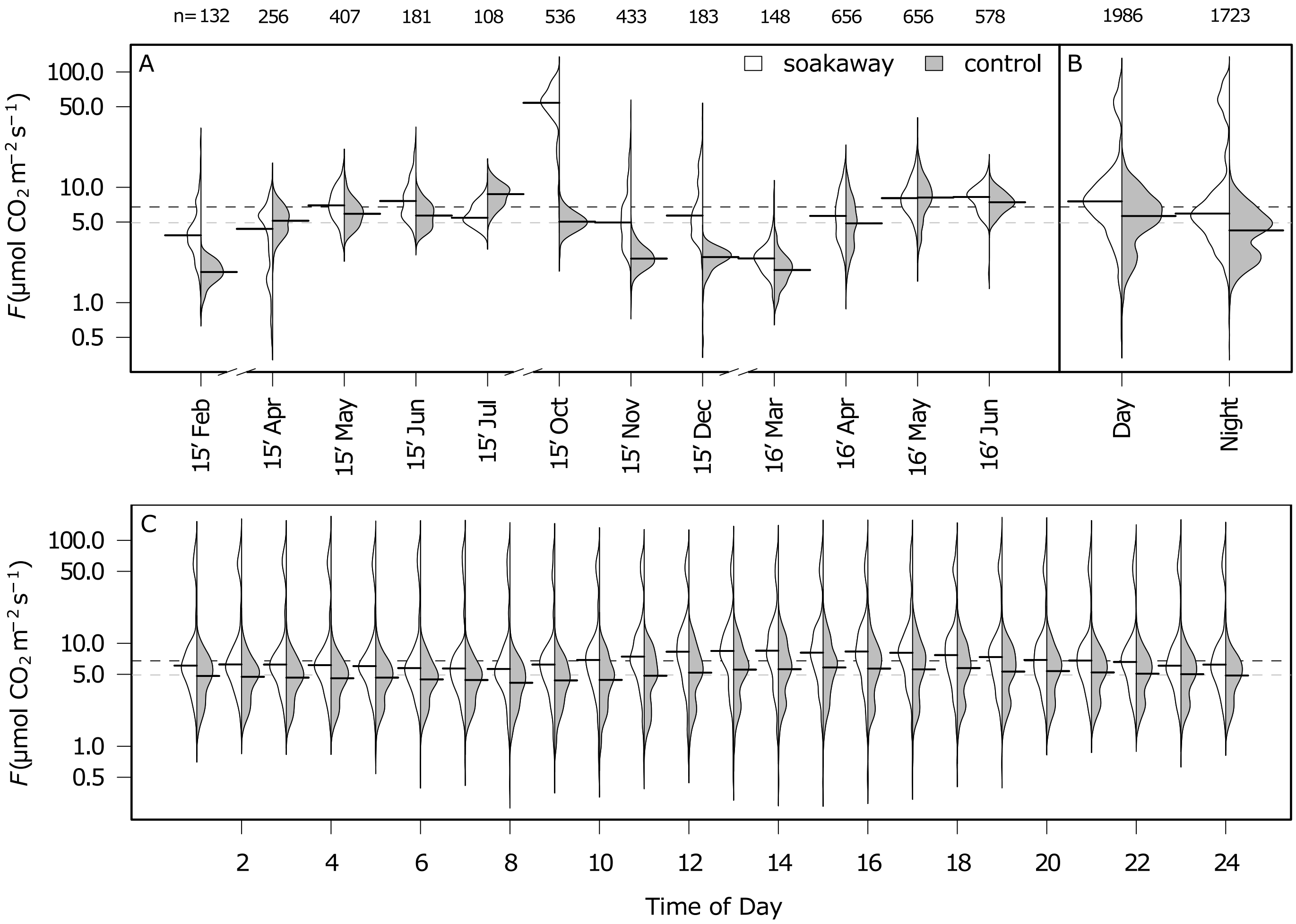

Figure 1: The monthly (A), diurnal (B), and hourly (C) variation of log-scaled CO_2 fluxes from soakaway (white) and control (grey) as density curve bean plots with median values (black lines) for each group. Dashed lines represent median soil gas flux values from the soakaway (black) and control (grey).

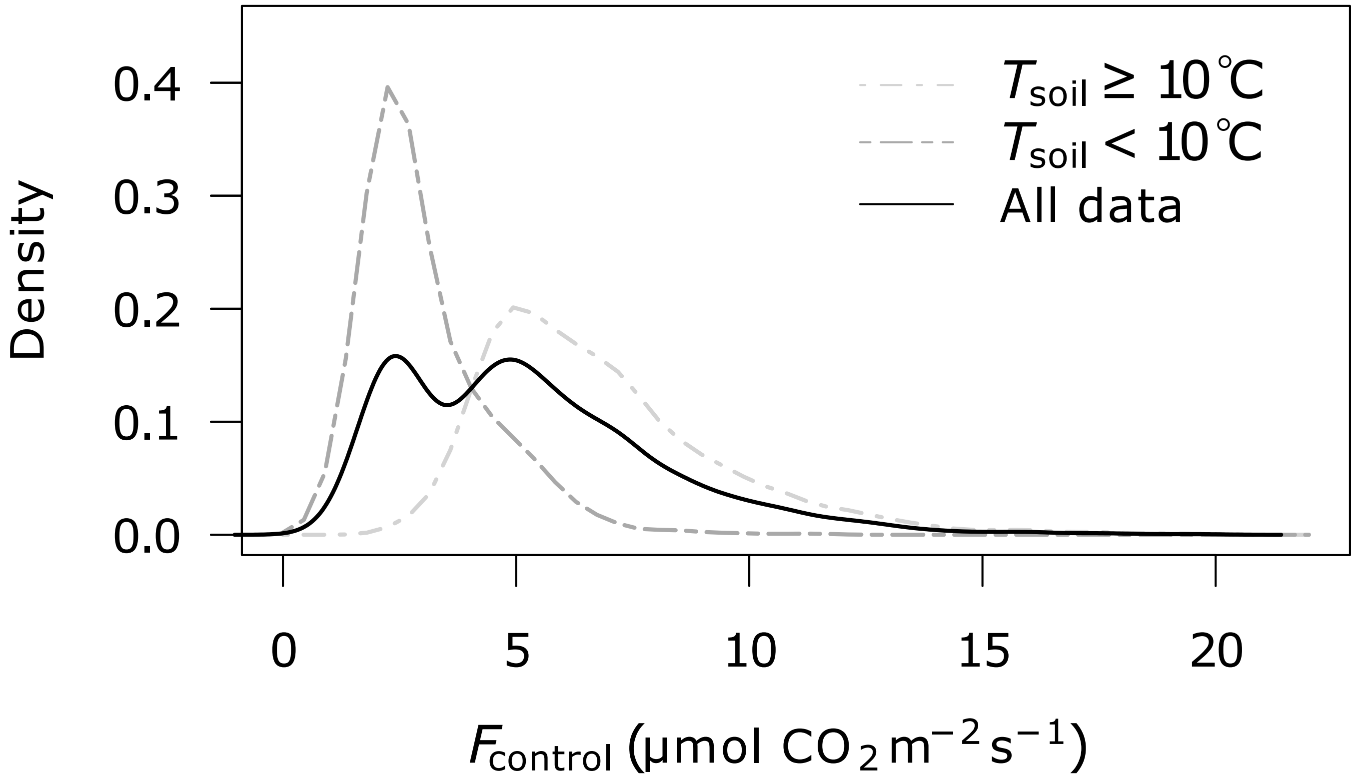

Figure 2: Density curves of control soil gas flux F_{control} grouped by temperature regime.

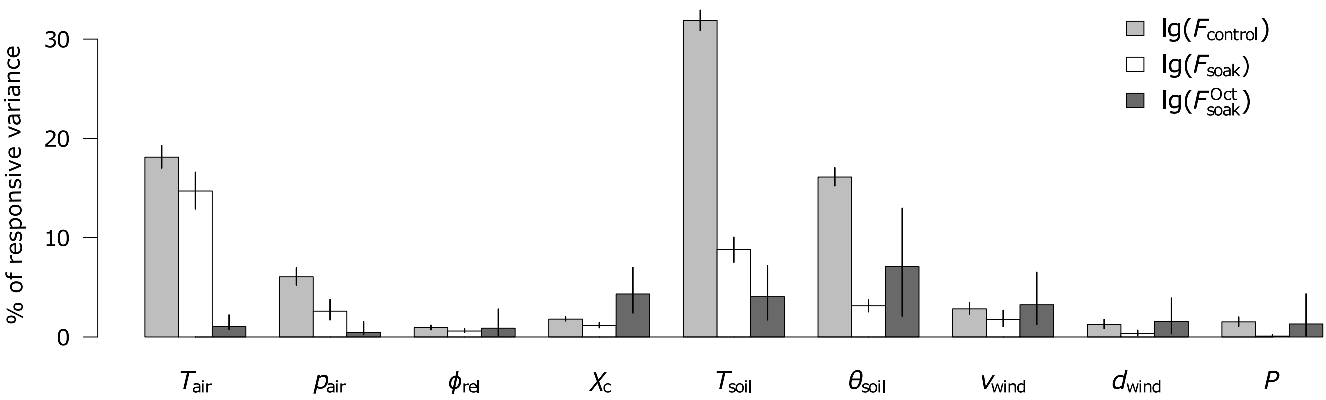

Figure 3: Relative importance of air temperature T_{air}, atmospheric pressure p_{air}, relative humidity ϕ_{rel}, water vapor corrected $\mathrm{CO_2}$ mole fraction χ_{c}, soil temperature T_{soil}, soil volumetric water content θ_{soil}, wind speed v_{wind}, wind direction d_{wind}, and rain P for predicting $\lg F_{\mathrm{control}}$ (grey), $\lg F_{\mathrm{soak}}$ (white), and $\lg F_{\mathrm{soak}}$ (dark grey) with 95 % bootstrap confidence intervals (method = LMG, $b = 1000$).

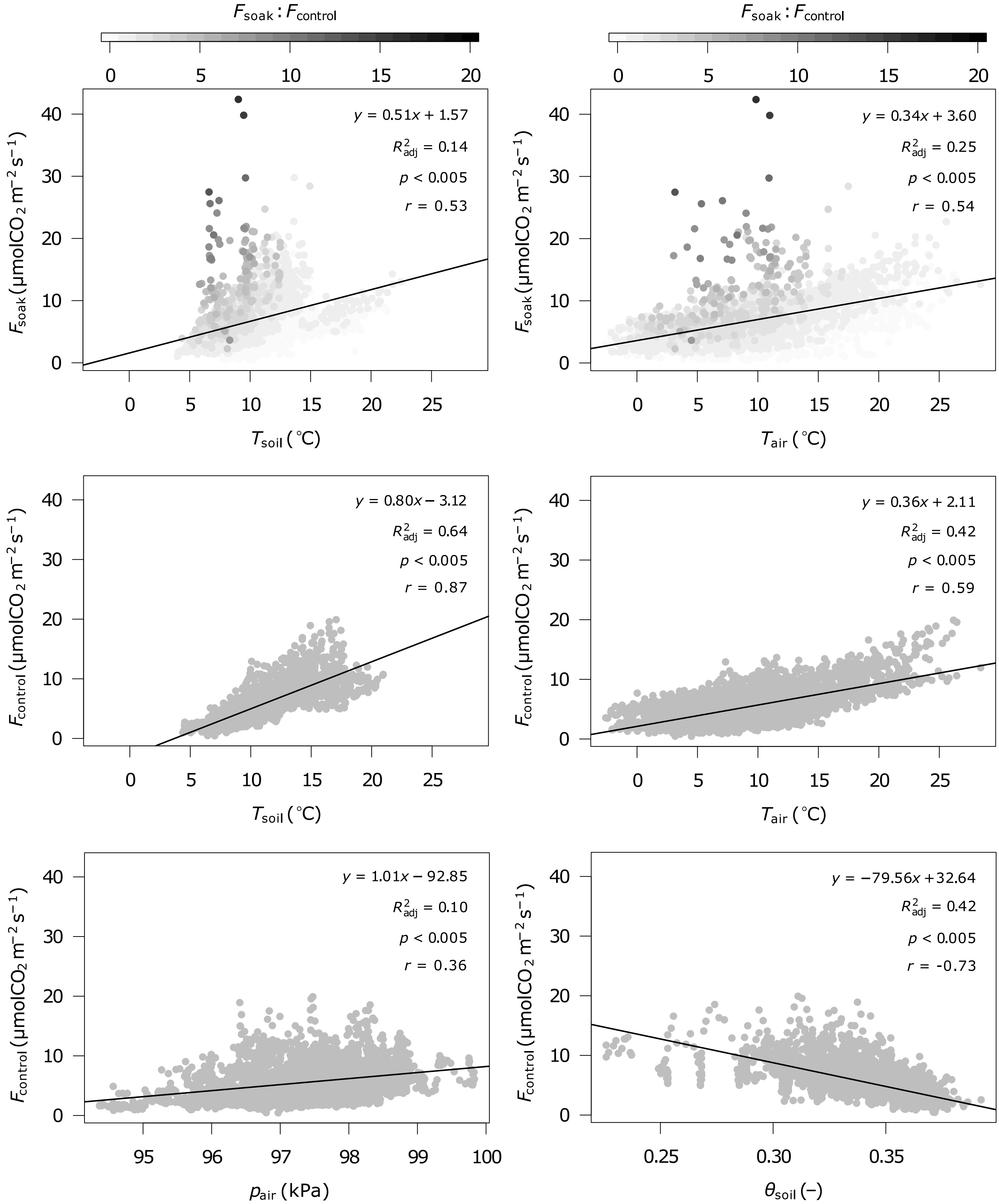

Figure 4: Dependence of measured fluxes (F_{soak} and F_{control}) on environmental factors (air temperature T_{air}, soil temperature T_{soil}) and of F_{control} on atmospheric pressure p_{air} and soil volumetric water content θ_{soil}. Plots for F_{soak} are color-coded by $F_{\text{soak}} : F_{\text{control}}$ ratio.

List of Tables

Table 1: Statistics for comparing soakaway and control soil gas fluxes using Welch's two-sample t-tests for unequal variances

	soakaway[a]		control[a]				
	mean	SD	mean	SD	t	df	p
non–October, $T_{\mathrm{soil}} < 10\,°\mathrm{C}$	5.17	3.69	3.00	1.35	20.99	1837	<0.05
non–October, $T_{\mathrm{soil}} \geq 10\,°\mathrm{C}$	8.00	3.20	7.33	2.66	6.69	3330	<0.05
October, $T_{\mathrm{soil}} < 10\,°\mathrm{C}$	36.52	10.94	4.13	0.72	11.44	14	<0.05
October, $T_{\mathrm{soil}} \geq 10\,°\mathrm{C}$	53.85	19.62	5.14	0.93	56.60	522	<0.05

[a] fluxes in $\mu\mathrm{mol\,CO_2\,m^{-2}\,s^{-1}}$; SD = standard deviation, t = test statistic, df = degrees of freedom, p = p–value

Table 2: Statistics for comparing day and night time soil gas fluxes using Welch's two-sample t-tests for unequal variances

	day time[a]		night time[a]				
	mean	SD	mean	SD	t	df	p
soakaway, non–October	7.37	3.73	5.89	3.52	11.50	3098	<0.05
soakaway, October	51.14	18.72	55.07	20.18	−2.33	516	<0.05
control, $T_{soil} < 10\,°C$	3.09	1.51	2.96	1.23	1.85	1152	<0.07
control, $T_{soil} \geq 10\,°C$	7.41	2.76	5.89	1.81	15.68	2238	<0.05

[a] fluxes in $\mu mol\, CO_2\, m^{-2}\, s^{-1}$; SD = standard deviation, t = test statistic, df = degrees of freedom, p = p–value

Table 3: Comparing results from this and previous studies of CO_2 emissions from septic systems

Study	Emissions [g CO_2 cap^{-1} d^{-1}]	
Diaz-Valbuena et al. (2011)	septic tank^a	33.3 ± 2.7
	vent pipe^a	335.0 ± 2.1
	leach field^a	negligible
Truhlar et al. (2016)	vent pipe^a	160.0 ± 3.2
	sand filter^b	120 ± 83
	leach field^b	130 ± 120
This study (all measurements)	soakaway^b	155 ± 208
This study (excluding October)	soakaway^b	78 ± 42
This study (only October)	soakaway^b	602 ± 222

^{a} geometric mean ± SD; ^{b} arithmetic mean ± SD.