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Impact of reduced tillage and cover cropping on the greenhouse gas budget of a maize/soybean rotation ecosystem

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ABSTRACT

Agricultural ecosystems have been viewed with the potential to sequester atmospheric carbon dioxide (CO_2) by increasing soil organic carbon (SOC) through reduced tillage and cover cropping practices. There remains considerable uncertainty, however, regarding the carbon (C) sink/source potential of these systems and few studies have examined C dynamics in conjunction with other important greenhouse gases. The objective of this study was to evaluate the impact of an alternative management scenario (reduced tillage and cover cropping) on ecosystem respiration (R_E) and nitrous oxide (N_2O) and methane (CH_4) fluxes in a maize (*Zea mays* L.)/soybean (*Glycine max* L.) rotation ecosystem in east-central Minnesota, United States. The control treatment was managed using fall tillage with a chisel plow in combination with a tandem disk, and the experimental treatment was managed using strip tillage and a winter rye (*Secal cereale*) cover crop. Over the two-year study period (2004–2005), cumulative R_E was 222.7 g C m^{-2} higher in the alternatively managed treatment as a result of increased decomposition of the cover crop residue. N_2O fluxes were similar in both treatments during the 2004 growing season and were 100.1 mg N m^{-2} higher in the conventional treatment during the 2005 growing season after nitrogen (N) fertilization. N fertilization and fertilizer type were the dominant factors controlling N_2O fluxes in both treatments. CH_4 fluxes were negligible in both treatments and often below the detection limit. Cumulative growing season N_2O losses in the control and experimental treatments, which totalled 38.9 ± 3.1 and 26.1 ± 1.7 g C m^{-2} when converted to CO_2 equivalents, were comparable to the annual estimates of net ecosystem CO_2 exchange in both treatments. This study further supports that N_2O losses are an important component of the total greenhouse gas budget of agroecosystems. It also suggests that spring cover cropping, without residue removal, has limited C sequestration potential. The results from this study, however, may not necessarily represent equilibrium conditions in the experimental treatment. Rather, they are a measure of the transient response of the system after tillage conversion and cover crop addition. It is expected that the soil microbes will continue to adjust to the reduction in tillage and increased C inputs. Therefore, continued, long-term monitoring is needed to confirm whether the results are representative of equilibrium conditions.

1. Introduction

The intensive management of agricultural soils has resulted in the depletion of soil carbon (C) stocks and has increased atmospheric carbon dioxide (CO_2) levels. Reversing this trend by increasing the C sink potential of agricultural soils could help to offset some of the rise in atmospheric CO_2 concentrations. It has been suggested that conservation tillage (Lal and Bruce, 1999; Lal, 2003, 2004) and cover cropping during fallow periods (Baker and Griffis, 2005) could increase the amount of C sequestered in agricultural soils. A review of soil organic carbon (SOC) studies from West and Post (2002) concluded that conservation tillage could, on average, sequester 0.60 ± 0.14 t C ha^{-1} y^{-1}. Several recently published studies, however, have found little to no difference in SOC in conventional and reduced-tillage systems (Dolan et al., 2006; Venterea et al., 2006; Baker et al., 2007; Blanco-Canqui and Lal, 2008).

While SOC measurements allow researchers to estimate soil C gains and losses, they provide little information about the dynamic exchange processes. If management strategies are to be improved to increase C uptake, then a comprehensive understanding of how C cycles through agroecosystems needs to be developed. In addition to SOC measurements, soil respiration (R_S), ecosystem

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respiration (R_E), and net ecosystem CO_2 exchange (F_N) measurements are needed to examine how C is cycled between terrestrial ecosystems and the atmosphere. Recent observations of F_N in conservation tillage maize/soybean systems have shown little or no C gains (Baker and Griffis, 2005; Verma et al., 2005). In contrast, Hollinger et al. (2005) reported a net C gain of 90 g m^{-2} y^{-1} in a conservation tillage maize/soybean system. However, Dobermann et al. (2006) have argued that weed growth could have been the main cause of the increased C sequestration. A better understanding of how R_S and R_E contribute to C losses in agricultural systems would help to explain the large variation observed in budget estimates and could provide insight regarding how to better manage these pools for C sequestration.

While much attention has focused on understanding how conservation tillage will impact C sequestration, understanding the potential impact on nitrous oxide (N_2O) and methane (CH_4) emissions is also critical given that the global warming potential (i.e. how much a given mass of gas contributes to global warming) of N_2O and CH_4 are 296 and 23 times larger than that of CO_2 over a 100 year period, respectively (Ramuswamy et al., 2001). A number of studies have reported that conservation tillage systems have higher N_2O emissions when compared to conventionally tilled systems (Robertson et al., 2000; Mummey et al., 1998; Ball et al., 1999). Interpreting the impact of tillage on N_2O emissions, however, can be complicated by nitrogen (N) fertilization. Venterea et al. (2005, 2009) have shown that fertilizer type can have a large impact on N_2O emissions. Similarly, McSwiney and Robertson (2005) and Wagner-Riddle et al. (2007) have shown that fertilizer application rates also influence N_2O fluxes and can potentially be reduced by using best management practices (BMPs). Therefore, understanding the connection between N fertilization and tillage is critical when investigating the impact of tillage on N_2O losses.

It has also been argued that conservation tillage may increase CH_4 oxidation rates as a result of improved soil structure, which allows for greater rates of CH_4 diffusion into the soil profile (Smith et al., 2001). Current research, however, has found no significant difference in CH_4 uptake/loss between conventional and conservation tillage systems (Venterea et al., 2005; Robertson et al., 2000).

The goal of this research, therefore, was to improve the understanding of how C cycles through maize/soybean agroecosystems in east-central Minnesota, United States, and to evaluate whether an alternative management strategy (reduced tillage and cover cropping) has a significant impact on CO_2, N_2O, and CH_4 losses. Two key questions were addressed: (1) are R_S, R_E, N_2O and CH_4 fluxes different in a conventionally tilled and reduced-tillage system with spring cover cropping? and (2) what factors are responsible for the differences in R_E, R_S, N_2O, and CH_4 fluxes?

2. Materials and methods

2.1. Site description

Research was conducted in two adjacent fields at the University of Minnesota Rosemount Research and Outreach Center, MN (44°45′ N, 93°04′ W). The soils at the site are a Waukegan silt loam (fine-silty over skeletal mixed, superactive mesic Typic Hapludoll) consisting of a silt loam surface layer 0.5–1.8 m thick overlying a layer of sand and gravel >20 m thick. The FAO classification is Chernozem. The field site has been under cultivation for the past 125 years; however, prior to cultivation, it consisted of an upland dry prairie (Griffis et al., 2005). The two fields are directly adjacent to each other with a road separating them. Both fields were managed using a maize (*Zea mays* L.)/soybean (*Glycine max* L.) rotation, with soybeans planted in 2004 and maize planted in 2005. The southernmost field was tilled conventionally using fall tillage with a chisel plow used in combination with a tandem disk and was designated as the control treatment. The northern field was tilled using an alternative technique consisting of strip tillage and a cover crop of winter rye (*Secale cereale*) and was designated as the experimental treatment. Tillage intensity in the experimental treatment was reduced in 2001 when chisel plowing was replaced with strip tillage and the cover crop was changed from oats (*Avena sativa*) to winter rye in 2003. The soybeans were planted into the rye and the rye was killed with a herbicide approximately ten days later. At the beginning of the growing season in 2005, anhydrous ammonia was knifed into the soil in the control treatment and urea was broadcast on the experimental treatment at a rate of 112 kg actual N per hectare.

2.2. Micrometeorological flux measurements

F_N was measured in both fields using the eddy covariance (EC) approach. The EC system consisted of a 10 m mast instrumented with a 3D sonic anemometer-thermometer (CSAT3, Campbell Scientific Inc., Logan, UT) and an open-path infrared gas analyzer (Li7500, LiCor, Lincoln, NE) located in the center of both the control and experimental treatments. Raw signals (sonic temperature, wind fluctuations, water vapor, and carbon dioxide concentrations) were recorded at 10 Hz and the mean covariances were computed over 30 min intervals.

Ancillary measurements at each tower included: (1) net radiation (NR Lite Net Radiometer, Kipp & Zonen, Netherlands; also computed by summing the downwelling and upwelling short and longwave radiation components); (2) soil heat flux and soil temperature; (3) soil water content; and (4) leaf area index (LAI). Downwelling and upwelling long and short wave radiation were measured using up and downward-facing pyranometers and pyrgeometers, respectively (Eppley Laboratories, Newport, RI). Soil heat fluxes were measured using soil heat flux plates (Huskeflux, the Netherlands and REBS, Seattle, WA) installed at 10 cm and corrected caliorimetrically. Soil temperature and volumetric water content were measured at 8 depths ranging from 0.05 to 1 m using copper-constantan thermocouples (Omega Engineering, Stamford, CT) and time-domain reflectometry (TDR 100, Campbell Scientific, Logan, UT). Soil and radiation measurements were made at 30 or 60 s intervals and averaged over 30 min. Soil water content measurements were made every 30 min. A more detailed site description can be found in Baker and Griffis (2005).

A quality control procedure similar to that used by the CarboEurope community was used to assess all half-hourly fluxes. Three parameters were used to evaluate the fluxes: (1) the integral turbulence parameter (α); (2) the stationarity parameter (β); and (3) a coordinate deviation parameter (ε) (Foken and Wichura, 1996; Thomas and Foken, 2002; Gockede et al., 2004; Foken et al., 2004). A quality parameter was also added in which fluxes were flagged if over 10% of the 10 Hz measurements were deemed to be outliers. These quality control criteria were then used to assign individual quality flags and were grouped to get a final quality flag for assessing the fluxes (Foken et al., 2004). Fluxes with final flag values between 1 and 3 were considered of high quality and used for further analysis in this study.

Nighttime fluxes were first screened using a box-plot filter that used a centered window to examine 10, non-overlapping, half-hourly time periods. Fluxes within the 10-point window that were greater than 25% of the inner quartile range were removed assuming that EC measurements usually contain approximately 20% random error (Wilson et al., 2002; Massman and Lee, 2002). The box-plot filter was used in addition to the CarboEurope method because a few outliers still remained after applying the methodology.

2.3. Soil chamber measurements

Static soil chambers were used to measure R_S, N_2O, and CH_4 fluxes. The chambers and bases were constructed from rectangular, stainless steel 53 cm × 32 cm × 8.6 cm and were covered with a reflective insulating material (Refletix, Markleville, IN) to prevent chamber heating (Venterea et al., 2005). A 6.4 mm × 10 cm stainless steel vent tube was used to equalize the pressure inside the chamber with the ambient atmosphere to attenuate pumping action caused by pressure differences (Hutchinson and Moiser, 1981; Gaumont-Guay et al., 2006). A sampling port constructed from a brass tube fitting with a male o-seal connector (part# B-200-1-OR, Swagelock, Solon, OH) was inserted into the top of the chamber and a rubber septum was inserted into the 3.2 mm end of the connector to prevent air leakage. The bases were inserted into the soil to a depth of approximately 6–7 cm.

No mechanical devices were used to mix the air inside the chamber. Theoretical and experimental evidence has shown that even without mechanical mixing, forced mixing usually occurs inside of the chamber as the result of temperature and pressure driven mass flow (Hutichinson et al., 2000). To negate possible problems associated with a lack of mixing, a 4-inlet hypodermic tubing reducer (part# STCM-13-20/4, Small Parts Inc., Miami Lakes, FL) was connected to the 2.3 mm end of the connector inside the chamber and teflon tubing, 0.8 mm × 23.5 cm, was connected to the 4 inlets on the manifold and fixed to the inside of the chamber. This allowed air to be pulled from different points within the chamber, to provide a representative sample. A rubber gasket made of ethylene propylene diene terpolymer (EPDM) lined the bottom of the chamber to prevent air exchange between the chamber and the atmosphere and spring clips were used to hold the chamber tight to the base during sampling.

In 2004, soil fluxes were measured in the control treatment from DOY 166 to DOY 281 and in the experimental treatment from DOY 189 to DOY 234. Fewer measurements were made in the experimental treatment because the winter rye was too tall for the chambers. The rye was cut in the late spring to prevent the standing rye residue from shading the emerging soybeans. In 2005, fluxes were measured in both treatments from DOY 146 to DOY 242. Measurements were taken between 8:00 AM and 4:00 PM (CST) and sampling alternated between morning and afternoon for each of the treatments. During the summer of 2004, 24 chambers were installed in each field; 12 within the row and 12 centered between the rows. The chambers were initially placed at distances of 30, 50, and 70 m along four transects N, S, E and W of the EC tower located in the middle of each field. Midway through the growing season, the chambers to the E and W of the tower were moved so that they were located along two transects, one SE and one NW of the tower. At the beginning of August those chambers were moved so that they were located on two transects NE and SW of the tower. Multiple chambers were placed around each EC tower because the flux footprint extended out approximately 100 m upwind of each tower and varied depending on wind direction and atmospheric stability. Multiple measurements were also made in each treatment to account for the spatial variability in soil fluxes, and are not intended to represent replicated measurements in each treatment.

In 2005, a 160 m × 160 m grid was established around each tower. The grid was then divided into four 80 m × 80 m blocks, which were then subdivided into sixteen 20 m × 20 m cells of which two cells were randomly chosen from each block. In the experimental treatment, only one chamber was placed between the rows. In the control field, chambers were placed between the rows to provide better spatial coverage since ammonia was knifed into the soil between the rows at the beginning of the growing season. Analysis of the 2004 data indicated that a minimum of 11 and 30 chambers were needed to obtain errors about the mean that were approximately 20% of the mean daily CO_2 and N_2O fluxes, respectively. Time limitations, however, did not allow 30 chambers to be sampled in each treatment. The necessary CH_4 sample size was not calculated because too few daily CH_4 fluxes could be resolved in 2004 to obtain an accurate estimate.

During 2004, 4 air samples per chamber were collected at 0, 20, 40, and 60 min intervals. The total sampling time was reduced to 40 min at the beginning of July 2004 after it was determined that there was no difference in fluxes calculated using 40 and 60 min sampling times ($p < 0.05$). During the summer of 2005, air samples were collected at 0, 10, and 20 min intervals to minimize the effect of increasing gas concentration gradients in the chamber headspace, which can reduce the flux (Hutchinson and Moiser, 1981). The sampling time was increased to 0, 15, and 30 min midway through the summer because more travel time was needed to move between the individual chambers.

At each measurement interval, approximately 12 ml of air was removed from the chamber using a 20 ml syringe (Norm-Ject, Tuttlingen Germany) and a 23 GA needle (Becton Dickinson, Franklin Lakes, NJ) and compressed into 9 ml headspace vials sealed with butyl-rubber septa (Alltech, Deerfield, IL). The vials were over pressurized to prevent ambient air from leaking into them and were vented using a 23 GA needle before analysis to equalize all vials to ambient pressure (Venterea et al., 2005). The samples were analyzed simultaneously for CO_2, N_2O, and CH_4 using two gas chromatographs. One was equipped with a thermal conductivity detector for CO_2 and an electron capture detector for N_2O. The other was equipped with a flame ionization detector for CH_4.

To account for gradient suppression related to increased gas concentration inside the head space, fluxes were calculated following Hutchinson and Moiser (1981). If the gas concentration inside the chamber did not increase at a decreasing rate then fluxes were calculated using a linear method following Gaumont-Guay et al. (2006).

Systematic error estimates for the CO_2, N_2O, and CH_4 fluxes were calculated using the error associated with the replicate ambient air samples, standards, and the measured rate of gas diffusion out of the chamber during a 40 min measurement period. All estimates were calculated for a worst-case scenario. CH_4 diffusion was not measured because the difference in the CH_4 mixing ratio between the chamber and the atmosphere was insignificant. The rate of gas diffusion between the chamber and atmosphere was estimated by sealing the base of the chamber and increasing the internal mixing ratio to ≈1333 μmol mol^{-1} CO_2 and ≈3.5 μmol mol^{-1} N_2O. These mixing ratios were approximately equal to the highest mixing ratios measured at 40 min during the 2-year study. The rate of diffusion out of the chamber was then measured using the same techniques used to measure the soil gas fluxes. The CO_2 diffusion rate was also measured using an infrared gas analyzer (Li-6262, Lincoln, NE) plumbed into a closed loop. Over the 40 min period, the chamber CO_2 mixing ratio changed by approximately 2 μmol mol^{-1} and there was a negligible change in the N_2O mixing ratio. When the systematic errors were propagated through the flux calculations, it was found that under worst-case conditions, CO_2, N_2O, and CH_4 fluxes could be underestimated by 5.5%, 3.3%, and 4.3%, respectively. These error estimates, however, do not account for systematic errors that result when the chamber modifies near-surface atmospheric mixing processes and the interfacial layer between the soil and atmosphere. When combined, these factors could have lowered post-deployment gas exchange rates, and could have caused the fluxes to be underestimated by as much as 30% (Hutichinson et al., 2000).

2.4. Determining ecosystem respiration

R_E was assumed to be equal to F_N at times where incoming solar radiation was equal to 0 W m^{-2} or when air temperature (T_a) and 5 cm soil temperature (T_5) were less that 0 °C (i.e. during nighttime and non-growing season conditions) (Barr et al., 2004). A simple model was then used to estimate daytime and nighttime respiration for the entire experimental period (Barr et al., 2004):

$$R_E = F(T,t) = \frac{r_w(t) r_1}{1 + \exp[r_2(r_3 - T)]} \quad (1)$$

where r_1 (μmol CO_2 m^{-2} s^{-1}), r_2 (°C^{-1}), and r_3 (°C) are empirically fit coefficients, T is either T_5 or T_a, and r_w is a dimensionless tuning parameter that accounts for factors other than temperature that influence R_E, such as phenology and soil water content. The numerator represents the baseline R_E and the denominator modifies R_E for changes in temperature. The model was separately fit to the growing and non-growing season data for each treatment to account for the influence of autotrophic and heterotrophic respiration on R_E. The growing season for the experimental treatment was defined as DOY 191–284 in 2004 and DOY 172–285 during 2005. For the control treatment, the growing season was defined as DOY 195–282 in 2004 and 171–285 in 2005. All other data were assumed to be non-growing season data.

The function was first fitted to the measured R_E and T_5 or T_a for the growing and non-growing season periods, and the coefficients r_1–r_3 were determined for each period using a least-squares fitting method. Secondly, the coefficient r_w was computed by performing a linear regression on the modeled and measured R_E and was assumed to be equal to the slope of the regression line. Lastly, final R_E was computed using T_5 or T_a, the original coefficients and r_w. Modeled R_E versus measured R_E is shown in Figs. 1 and 2.

2.5. Statistics

All analyses were performed using Matlab version 2006b (TheMathWorks Inc., Natick, CT). When the soil fluxes were very small, it was often difficult to resolve a change in gas concentration within the chambers during the measurement period. This often resulted in inaccurate flux estimates. To quality control the chamber flux measurements, we examined the linear regressions of all of the individual chamber measurements by plotting their R^2 values on a histogram. Based on the distribution of values over the two-year period, it was determined that fluxes with R^2 values less than 0.7 were unreliable and were not included in subsequent analyses.

Mean daily soil fluxes were calculated using the filtered measurements and all error terms/bars represent the standard error about the mean. Cumulative soil CO_2, N_2O, and CH_4 losses were calculated by linearly interpolating between the mean daily flux measurements and by integrating the area under the gap filled curve. Total error for the interpolated sum was estimated using a Monte-Carlo simulation (Griffis et al., 2003).

A Wilcoxon Rank Sum Test was used to determine whether fluxes were statistically different. Comparisons were made using all of the valid measured fluxes and not the daily means. No statistics were performed on the CH_4 measurements because low fluxes resulted in relatively large errors. An unequal number of soil flux measurements were collected from the control and experimental treatments in 2004. Consequently, soil fluxes were compared and cumulative soil fluxes were computed from DOY 189 to 234 in 2004 and DOY 146 to 242 in 2005 when measurements were made in both treatments.

3. Results and discussion

3.1. Ecosystem respiration: budget and seasonal patterns

From the beginning of the growing season in 2004 through fall 2005, cumulative R_E for the experimental treatment was significantly greater ($p < 0.05$) than cumulative R_E for the control (Table 1). These results run contrary to the hypotheses proposed by Lal and Bruce (1999) and Desjardins et al. (2005) where they speculated that limiting tillage would slow C decomposition and reduce C losses. As expected, the additional contributions of autotrophic respiration and the residues from the rye cover crop in 2004 increased cumulative R_E. Sampling of the rye after cutting indicated aboveground dry matter of 440 ± 146 g m^{-2} (n = 12). When growing and non-growing season losses were summed together for the experimental and control

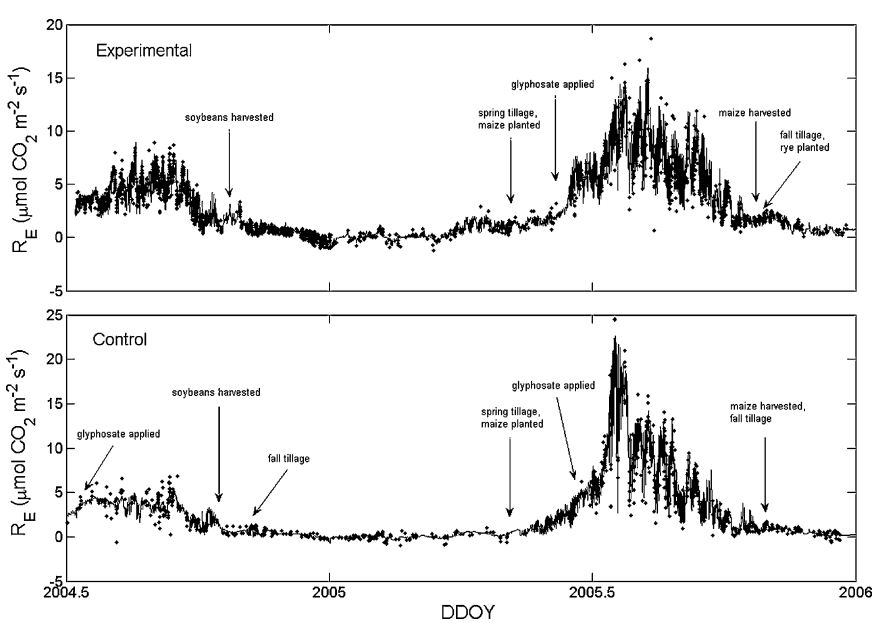

Fig. 1. Comparison of modeled (solid line) to measured (points) R_E for the experimental and control treatments from the beginning of the 2004 growing season to the end of 2005.

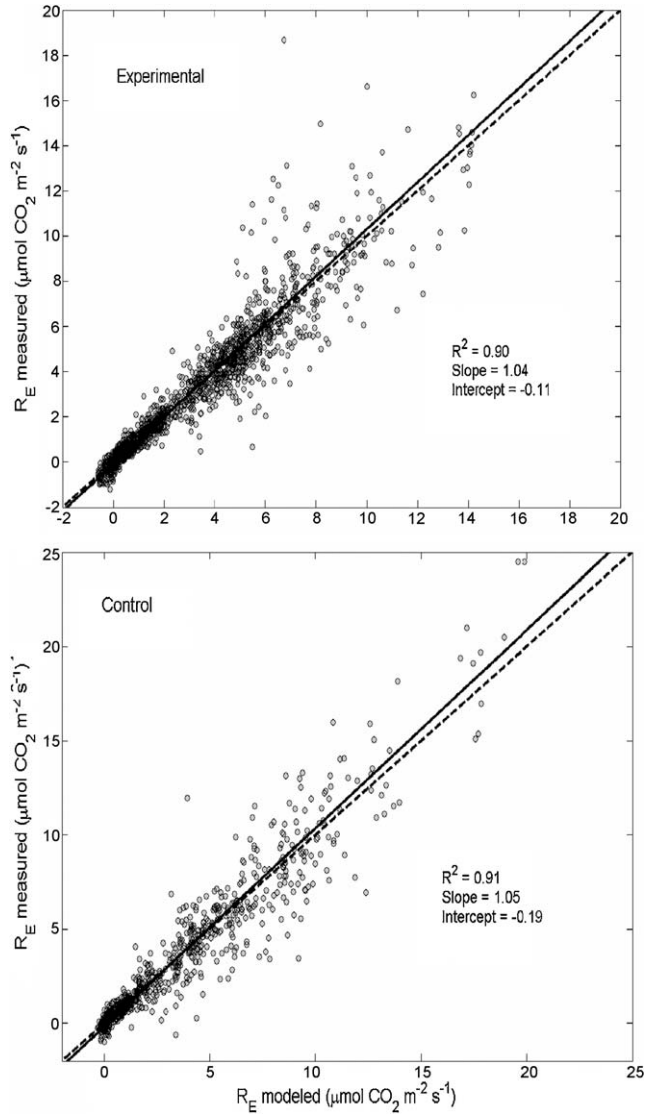

Fig. 2. Comparison of measured R_E to modeled R_E for the experimental and control treatments. The dashed line represents a 1:1 relationship.

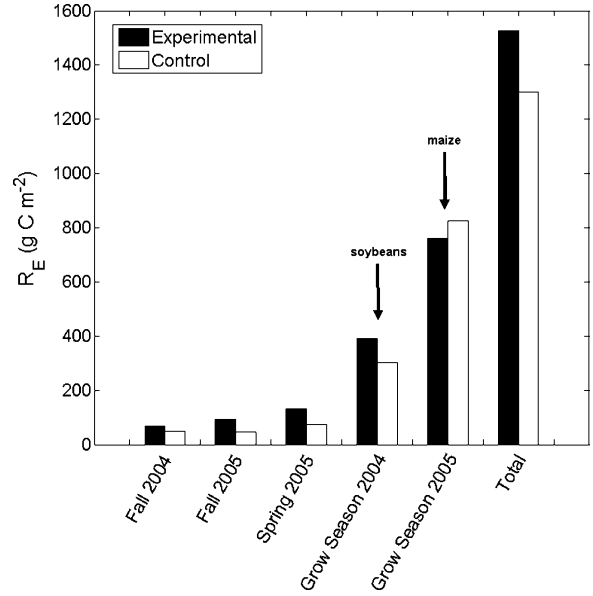

Fig. 3. Cumulative R_E for different periods during the 2004 and 2005 season. Total losses were integrated from the start of the 2004 growing season until the end of 2005.

treatments, the experimental treatment lost 220.7 g C m^{-2} more than the control (Table 1). This indicates near-complete respiration of the rye residue, assuming a C content of 50%.

Seasonal changes have a large impact on the amount of C sequestered and lost in natural and managed ecosystems (Black et al., 1996; Griffis et al., 2004; Barr et al., 2004; Baker et al., 2007; Flanagan and Johnson, 2005; Verma et al., 2005). Fig. 3 shows the cumulative R_E for the individual growing and non-growing periods during 2004 and 2005. Cumulative R_E was highest during the growing seasons, and C losses were 389 ± 13.5 and 525.1 ± 16.8 g C m^{-2} larger in both treatments during the maize growing season than during the soybean growing season. The ratios of maize R_E to soybean R_E were 2.0 and 2.7 for the experimental and control treatments. The growing season losses in the experimental treatment were smaller than those found for other reduced-tillage systems. For example, Verma et al. (2005) reported losses of approximately 900.0 g C m^{-2} in a rainfed maize system and 700.0 g C m^{-2} for a rainfed soybean system.

Table 1
Comparison of cumulative R_E for the experimental and control treatments during 2004 and 2005.

	Cumulative R_E (g C m^{-2})		
	Experimental	Control	SIG
Fall 2004	69.0 ± 2.0	47.9 ± 1.5	**
Fall 2005	93.5 ± 2.4	44.7 ± 1.2	**
Spring 2005	154.1 ± 3.9	79.5 ± 2.5	**
Grow 2004	409.4 ± 8.9	302.7 ± 6.8	**
Grow 2005	799.3 ± 16.2	827.8 ± 18.1	**
Total	1525.3 ± 19.2	1302.6 ± 19.8	**

** Values that are significantly different ($p < 0.05$). Standard errors were calculated by assuming the half-hourly EC measurements had 20% error.

To better understand how seasonality and management affected R_E, mean daily values were compared and R_E was integrated from the beginning of the 2004 growing season to the end of 2005 (Fig. 4). R_E was elevated in the experimental treatment with respect to the control for the majority of 2004 and 2005, likely owing to the decomposition of cover crop residues. Only in July 2005 was R_E, on average, higher in the control treatment. When the seasonal differences between treatments were compared, cumulative C losses were statistically higher in the experimental treatment during all seasons except for the 2005 growing season (Table 1). During the 2005 growing season, C losses were approximately 28.5 g C m^{-2} higher in the control than in the experimental treatment. During the 2004 growing season, the difference between treatments was much larger with the experimental treatment losing approximately 106.7 g C m^{-2} more than the control. The increase in R_E in the control treatment in 2005 coincides approximately with the time that N fertilizer was applied to both treatments. N addition could have stimulated decomposition and R_E; however, numerous studies have found that N fertilization does not significantly influence SOC turnover (Dolan et al., 2006) or R_S (Liu et al., 2006; Ding et al., 2007). We hypothesize that the increase in R_E was related to increased growth and higher maintenance respiration after fertilization.

Similar to growing season R_E, R_S was also statistically higher in the experimental treatment in 2004 (Fig. 5, panel A) and in the control in 2005 (Fig. 5, panel B). Cumulative R_S was 143.0 ± 2.2 g C m^{-2} in the control treatment in 2004 whereas the cumulative R_S in the experimental was 159.6 ± 2.2 g C m^{-2}

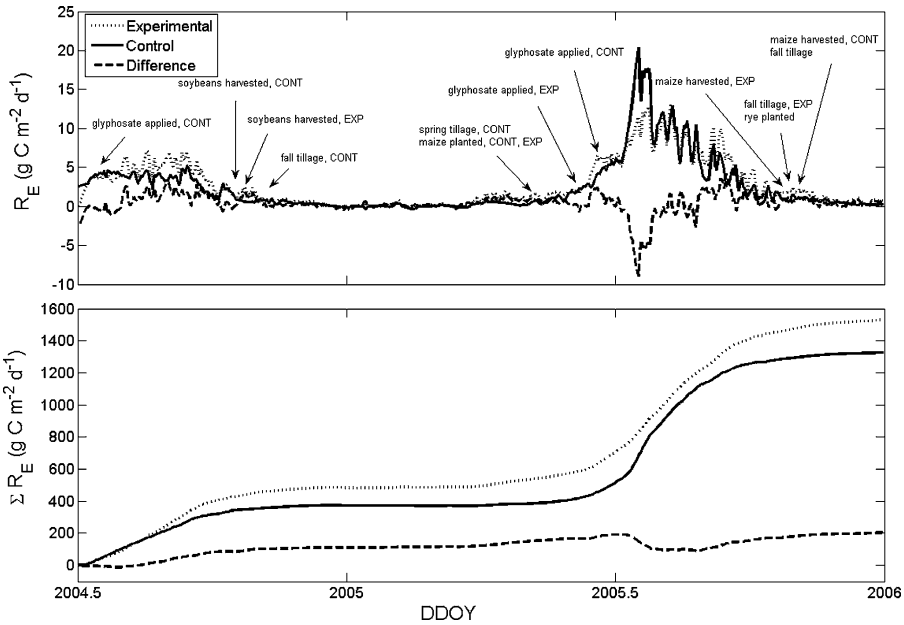

Fig. 4. Daily C losses for the experimental and control treatments and the difference in daily C losses (top panel); cumulative C losses for the experimental and control treatments and the difference in cumulative losses between treatments (difference = experimental − control) (bottom panel).

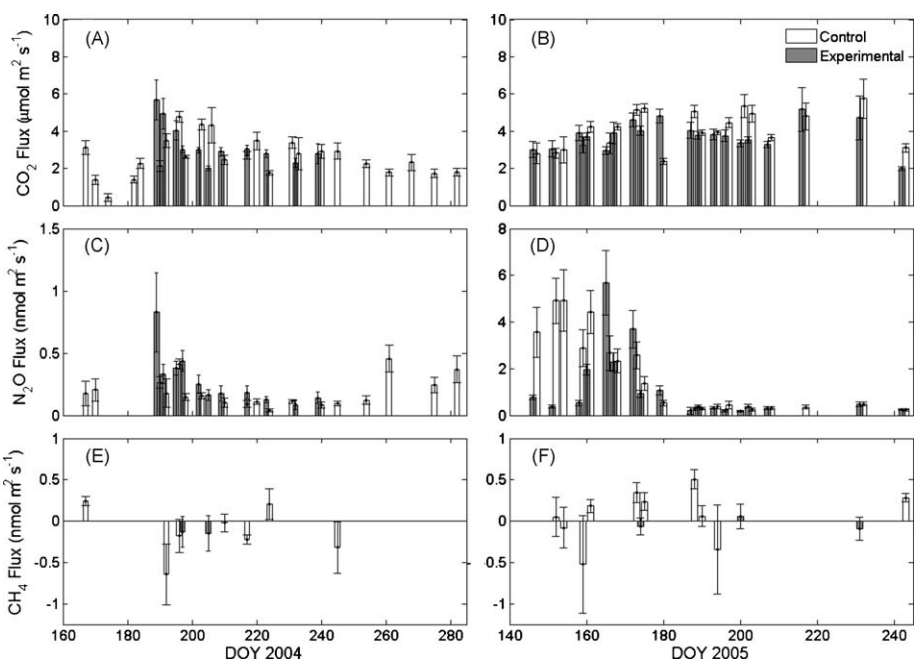

Fig. 5. Mean daily R_S (top panels), N_2O (middle panels), and CH_4 (bottom panels) fluxes in the control and experimental treatments for the 2004 and 2005 growing and non-growing season. The error bars represent the standard error about the mean for the daily measurements. There are an unequal number of daily measurements for the different gases and treatments because fluxes could not be resolved for every day and therefore were not included in the final data set. Also, more measurements were collected in the control treatment in 2004 because of problems getting the chambers established in the experimental treatment. Anhydrous ammonia and urea were added to the experimental and control treatments at a rate of 112 kg actual N per hectare, respectively, just prior to the first round of measurements in 2005.

respectively. In 2005, cumulative R_S was 394.2 ± 5.8 in the experimental treatment and 423.9 ± 5.1 g C m^{-2} in the control.

Differences in baseline R_E (i.e. $r_w(t)r_1$ in Eq. (1)) appeared to be the main factor driving differences in C losses between the two treatments. Overall, baseline R_E was higher in the experimental treatment the majority of the time with the exception of the end of the 2005 growing season when a large increase in baseline R_E was observed in the control (Fig. 6). Differences in baseline R_E between the two treatments tracked the differences in cumulative C losses with the exception of the 2005 growing season when C losses were higher in the control treatment and baseline R_E was higher in the experimental the majority of time. Increased C inputs likely had the greatest influence on baseline R_E in the experimental treatment.

3.2. Partitioning ecosystem respiration

R_S can contribute as much as 60% to R_E for temperate forest ecosystems (Davidson et al., 2006). Similarly, Jassal et al. (2007) found that the ratio of R_S to R_E varied between 0.52 and 0.86 for a

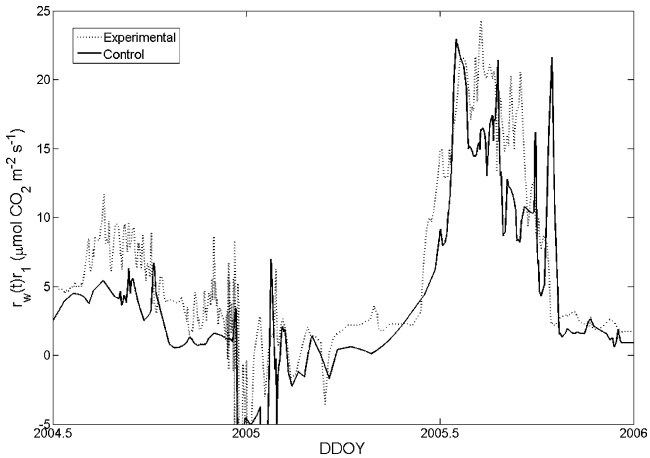

Fig. 6. Baseline respiration in the experimental and control treatments for 2004 and 2005. Baseline respiration was assumed to be equal to $r_w(t)r_1$ in Eq. (1).

Douglas fir stand. To better understand how management impacts R_E, estimates of above ground and below ground respiration were determined by comparing R_S to R_E. R_S was assumed to be equal to the sum of below ground heterotrophic and autotrophic respiration. Differences between R_S and R_E were highly variable between treatments and growing seasons; however, it can been seen that in 2005, minimum R_S/R_E ratios of 30–40% occurred in mid-July, around the time in which the maize passed peak growth (Fig. 7). In 2004, minimum R_S/R_E ratios of 50% occurred in mid-August in the experimental treatment around the time in which the soybeans passed peak growth. In 2004, ratios were highly variable in the control treatment; however, R_S/R_E was around 50–60% if only the values after leaf out (DOY 190) were used.

R_S/R_E ratios computed from the cumulative C losses for the 2004 and 2005 growing seasons were 70.7 ± 4.2% and 59.3 ± 7.7% for the experimental treatment and 80.9 ± 3.7% and 57.9 ± 8.2% for the control, respectively (Fig. 8). The soybean growing season ratios were similar and the maize growing season ratios were approximately 10% smaller than those computed from annual estimates of R_S and R_E for a boreal aspen forest in Griffis et al. (2004). Ratios for the soybean growing season were approximately 11.4% and 23% higher than R_S/R_E for the maize growing season. When the cumulative growing season

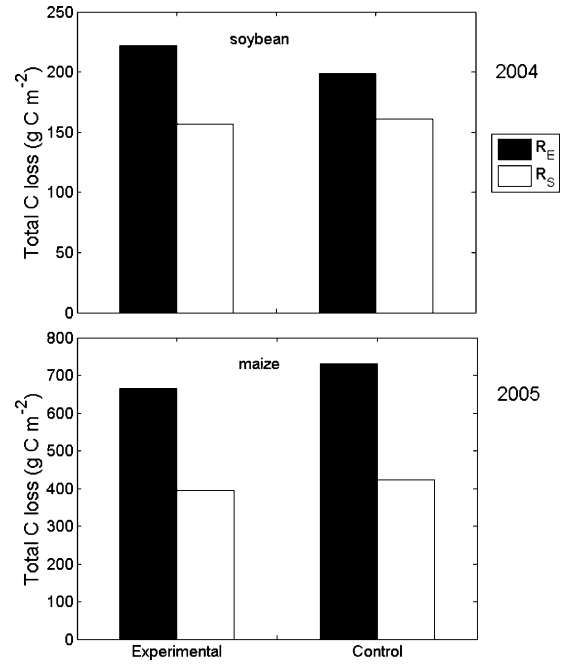

Fig. 8. Cumulative R_S and R_E for the soybean (2004) and maize (2005) rotations. Cumulative losses were calculated from DOY 189 to 234 in 2004 and DOY 146 to 242 in 2005 when soil fluxes were collected in both treatments.

R_E estimates are multiplied by their respective R_S/R_E ratios, cumulative R_S was 289.4 ± 12.1 and 244.6 ± 9.1 g C m^{-2} and cumulative above-ground R_A was 120.0 ± 5.0 and 58.1 ± 2.2 g C m^{-2} 2 for the experimental and control treatments in 2004, respectively. In 2005, cumulative R_S was 474.0 ± 36.5 and 479.3 ± 39.3 g C m^{-2} and cumulative above-ground R_A was equal to 325.3 ± 25.0 and 348.6 ± 28.6 g C m^{-2} for the experimental and control treatments, respectively. R_S and above-ground R_A were significantly higher in both treatments during the maize phase of the rotation.

Compared to above-ground R_A, R_S exhibited a larger degree of control on R_E in both treatments during 2004 and 2005. The average difference between R_S and above-ground R_A for both treatments in 2004 and 2005 was approximately 178 and 140 g C m^{-2}, indicating that R_S was a larger component of R_E

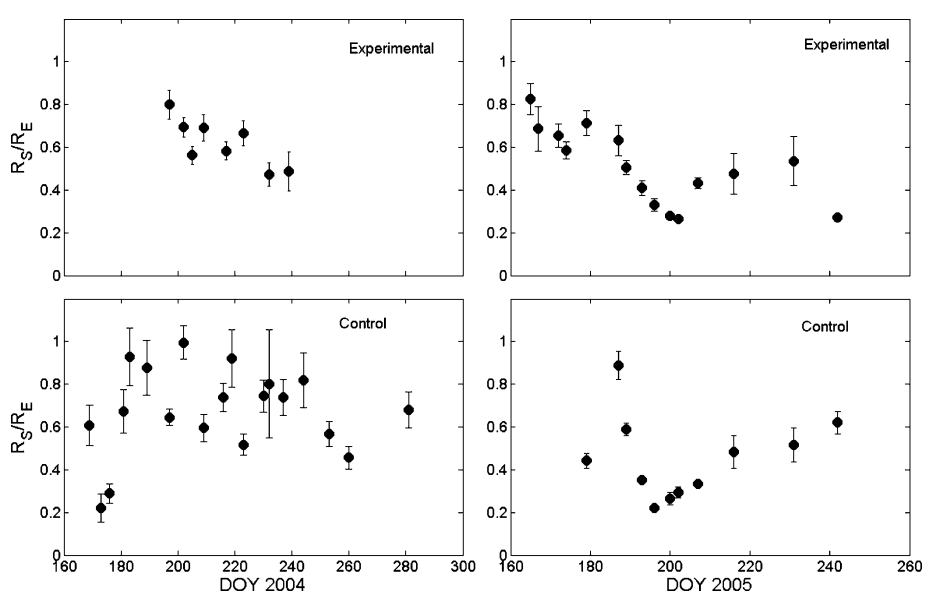

Fig. 7. R_S/R_E ratios for the different treatments during the soybean (2004) and maize (2005) growing seasons. The errors bars represent the mean error for the ratios and were calculated using the standard error about the mean for the R_S measurements and 20% error for the R_E measurements.

during the soybean phase of the rotation. The largest treatment differences were seen in 2004 when cumulative R_S and above-ground R_A were approximately 44.8 and 60.2 g C m^{-2} higher in the experimental treatment. Increased R_S in the experimental treatment in 2004 was most likely the product of increased respiration from the decomposition of the cover crop residues. If it is assumed that respiration from the soybeans was similar in both treatments, the lower R_S/R_E ratio in the experimental treatment in 2004 indicates that C turnover is relatively fast in covercropping systems and the decomposition of standing residues can make significant contributions to C losses. Limiting these losses by harvesting the cover crop therefore could decrease above ground C losses and improve the net C gain of reduced tillage agroecosystems that have cover cropping.

3.3. Soil N_2O and CH_4 exchange

There was no significant difference in N_2O fluxes in 2004 (Fig. 5, panel C); however, N_2O fluxes were significantly higher in the control treatment in 2005 after N fertilizer was applied to both treatments (Fig. 5, panel D). CH_4 fluxes for 2004 (Fig. 5, panel E) and 2005 (Fig. 5, panel F) were very small and were often below the detectable limit of the measurement technique. As a result, few CH_4 fluxes could be resolved, which resulted in several days where no daily averages could be determined. High spatial variability and a low number of CH_4 measurements also resulted in large standard errors about the mean for the daily measurements. As a consequence, no statistical analyses were performed on the CH_4 fluxes. Overall, CH_4 was deemed to be a negligible component of the greenhouse gas budget for the two treatments. In contrast, Venterea et al. (2005) and Robertson et al. (2000) found that agroecosystems were a measurable weak sink for CH_4.

From DOY 189 to 234 in 2004, cumulative N_2O–N loss in the control treatment was 16.3 ± 0.8 and 26.7 ± 1.4 mg N m^{-2} in the experimental treatment. When converted to CO_2 equivalents, cumulative N_2O losses were 2.1 ± 0.2 and 3.4 ± 0.4 g C m^{-2} in the control and experimental treatments, respectively. From DOY 146 to 242 in 2005, cumulative N losses were 306.5 ± 10.5 mg N m^{-2} in the control treatment and 206.4 ± 5.6 mg N m^{-2} in the experimental treatment. Overall, cumulative N_2O losses were approximately 18.8 and 7.7 times higher during the maize growing season than during the soybean growing season, however, no N fertilizer was applied during the soybean growing season.

In CO_2 equivalents, cumulative N_2O losses were 38.9 ± 3.1 and 26.1 ± 1.7 g C m^{-2} respectively. Baker and Griffis (2005) estimated that net C losses were approximately 80–90 g C m^{-2} during 2002 and 2003. Assuming that losses were similar in 2004 and 2005 (approximately 40–45 g C m^{-2} y^{-1}), then the global warming potential (GWP) of N_2O emission was similar to CO_2 losses in both treatments.

We cannot conclude that tillage had any effect on N_2O fluxes. Treatment differences were not observed until after N fertilizer was applied to both treatments. After fertilization, N_2O fluxes remained elevated for approximately 40 days in the control and 20 days in the experimental treatment. These results suggest that N fertilization was the primary factor regulating N_2O fluxes in both treatments. Wagner-Riddle et al. (2007) and McSwiney and Robertson (2005) also reported similar findings. The type of N fertilizer could have also played a key role in regulating N_2O fluxes. Venterea et al. (2005, 2009) measured significantly higher N_2O fluxes in treatments fertilized with anhydrous ammonia as compared to urea. The direct injection of anhydrous ammonia into the soil could have increased the amount of N available for nitrification and denitrification, which in turn increased N_2O fluxes in the control treatment. Venterea (2007) found that high soil nitrite (NO_2^-) levels increased N_2O losses and it is possible that anhydrous ammonia was more easily converted to NO_2^-. Soil N data, however, were not available from 2004 to 2005 to substantiate these hypotheses. Reducing fertilizer application rates and switching to a slow release N fertilizer is expected to reduce N_2O fluxes.

4. Conclusions

1. Changing from a conventional to alternatively managed maize/soybean rotation ecosystem (strip tillage with spring cover crop) caused R_E to increase by 222.7 g C m^{-2}. Decomposition of the rye cover crop was the main factor that increased cumulative R_E and R_S in the experimental treatment.
2. Cumulative N_2O losses, expressed in CO_2–C equivalents, were 12.8 g C m^{-2} higher in the conventional treatment during one of the two study years. N fertilization was the main factor that increased N_2O losses in both treatments and N fertilizer type likely had a large influence on the differences between the two treatments.
3. CH_4 oxidation or production was negligible in either treatment indicating CH_4 is a small component of the greenhouse gas budget of both systems.
4. N_2O losses, when expressed in CO_2–C equivalents, totaled 38.9 ± 3.1 and 26.1 ± 1.7 g C m^{-2} in the conventional and alternatively managed treatments after fertilization, comparable to the net ecosystem CO_2 exchange of both systems.

The results from this study do not necessarily represent equilibrium conditions in the experimental treatment. Rather, they are a measure of the transient response of the system after tillage conversion and cover crop addition. Dick (1992) has suggested that the biological potential of soils may initially be diminished when conventionally managed agroecosystems are converted to alternatively managed systems. We therefore expect that soil microbial populations will continuously adjust to these treatment effects. As a result, future long-term monitoring is needed to confirm whether the results from this study represent a short-term response of the system, or long-term changes in C cycling.

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