Tillage Effects on Soil Carbon Balance in a Semiarid Agroecosystem

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Tillage systems may affect soil C sequestration, with a potential impact on crop productivity or organic matter mineralization. We evaluated crop yield, C inputs to the soil, and in situ CO_2–C fluxes under no-till and conventional tillage (disk tillage) during the 3- to 6-yr period from the installation of an experiment in an Entic Haplustoll of the Semiarid Pampean Region of Argentina to elucidate the mechanisms responsible for possible management-induced soil organic matter changes. Yield and biomass production were greater under no-till than disk tillage for all the crops included in the rotation (oat [*Avena sativa* L.] + hairy vetch [*Vicia villosa* Roth ssp. *villosa]*, corn [*Zea mays* L.], wheat [*Triticum aestivum* L.], and oat). This result was attributed to the higher soil water content under no-till. Carbon inputs to the soil averaged 4 Mg C ha^{-1} yr^{-1} under no-till and 3 Mg C ha^{-1} yr^{-1} under disk tillage. Soil temperature was similar between tillage systems and CO_2–C emission was about 4 Mg C ha^{-1} yr^{-1}, with significant but small differences between treatments (~0.2 Mg C ha^{-1} yr^{-1}). Carbon balance of the soil was nearly equilibrated under no-till; meanwhile, greater C losses as CO_2 than inputs in crop residues were measured under conventional tillage. Organic C in the soil was 5.4 Mg ha^{-1} higher under no-till than the disk tillage treatment 6 yr after initiation of the experiment. Results showed that in our semiarid environment, C sequestration occurred under no-till but not conventional tillage. The sequestration process was attributed to the effect of the tillage system on crop productivity rather than on the mineralization intensity of soil organic pools.

Abbreviations: BLUP, best linear unbiased prediction.

Soils can act as sources or sinks for C depending on management, which can impact global warming (Lal et al., 1999; Follett et al., 2001). Reviews of published results from field experiments performed worldwide have shown that increases in soil C content usually result under no-till in comparison with tilled soils (West and Post, 2002; Alvarez, 2005). The magnitude of these increments is variable, depending on time since initiation of the experiments, with organic C levels being 8 to 15% higher in the upper layer of no-till soils. The absolute amount of C sequestration under no-till could not been associated with climatic conditions in some studies (West and Post, 2002; Alvarez, 2005), but others have suggested that sequestration is higher in semiarid environments (VandenBygaart et al., 2003). In relative terms, there is agreement that more C may be sequestered in semiarid environments than in humid areas (VandenBygaart et al., 2003, Steinbach and Alvarez, 2005). The mechanisms by which these increments are produced are not fully known.

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Soil organic C levels may be greater in no-till than under tillage even when crop biomass production and residue mass returned to the soil are similar between both management regimes (Alvarez et al., 1995a, 1998). One possible mechanism responsible for the increase of soil C in no-till soils when compared with tilled ones is a lower organic matter mineralization rate under no-till. After tillage, soil CO_2–C efflux is greater than under untilled scenarios (Dao, 1998; Rochette and Angers, 1999). This increase of soil CO_2–C efflux has a combination of different causes: the disruption of aggregates and mineralization of aggregate-protected organic matter (Kristensen et al., 2000), the temperature rise of tilled soil due to residue incorporation by machines, which also leads to an increased mineralization of organic pools (Alvarez et al., 2001), and the more rapid decomposition of buried residues under tillage management (Curtin et al., 1998; Alvarez et al., 2001). The consequence of these processes is that, some years after no-till adoption, aggregate-protected particulate organic matter is higher in no-till soils than in tilled ones (Bossuyt et al., 2002; Denef et al., 2004).

Soil water content had been reported to be higher in no-till systems than in plowed soils (Rasse and Smucker, 1999; Karamanos et al., 2004). This is a consequence of higher infiltration rates and lower evaporation under no-till (Unger and McCalla, 1980; Franzluebbers, 2002). Crop production and residue returned to the soil may also be greater under no-till in semiarid environments (Campbell et al., 1997; Hernanz et al., 2002), a phenomenon that has been attributed to the effect of tillage systems on soil water content (Unger and McCalla,

SOIL & WATER MANAGEMENT & CONSERVATION

1980). The greater the amount of residues returned to the soil, the greater the soil organic C level (Duiker and Lal, 1999). Consequently, another possible mechanism by which soil C increases in no-till in relation to tillage management is due to the effects of tillage systems on soil available water content.

The Pampas is a vast plain of around 50 Mha, which runs from 28 to 40° S in Argentina (Alvarez and Lavado, 1998). Mean annual rainfall ranges from 200 mm in the west to 1200 mm in the east and the mean annual temperature ranges from 14°C in the south to 23°C in the north. Agriculture is performed in the semiarid and humid portions of the region on well-drained soils, mainly Mollisols formed on loess-like materials. Nearly 50% of the area is devoted to agriculture, with soybean [*Glycine max* (L.) Merr.], corn, and wheat as the main crops (Hall et al., 1992). Since 1990, the use of no-till has increased exponentially, with around 50% of agricultural lands now cropped under this management (INDEC, 2003). The region is considered one of the most suitable areas for grain crop production in the world (Satorre and Slafer, 1999).

In the semiarid portion of the Pampas, agriculture was introduced around 60 yr ago using low-external-input farming systems based on cattle grazing and harvest crop rotations (Viglizzo et al., 2001). The cropped area increased gradually due to economic reasons, and partially as a consequence of an increase in rainfall during the last few decades (Viglizzo et al., 1995). A net loss of nutrients (Bernardos et al., 2001) and soil organic C occurred (Quiroga et al., 1996; Hevia et al., 2003); conservation tillage systems were introduced recently in this area (Bernardos et al., 2001) to prevent soil degradation. Soil organic C increases under no-till in these soils (Steinbach and Alvarez, 2005) but the processes involved are not fully understood. The impact of tillage management on organic matter mineralization has not been studied and local reports of grain yield response to tillage systems are scarce and contradictory across experiments (Buschiazzo et al., 1998; Díaz-Zorita et al., 2002).

Our objectives were (i) to determine the effect of no-till and a conventional tillage system used in the semiarid portion of the Pampas on the intensity of soil CO_2–C emissions, (ii) to develop a model suitable for prediction of soil CO_2–C efflux, (iii) to evaluate tillage system effects on grain yield and residue mass production of a common rotation used in the region, and (iv) to calculate the soil C balance. We hypothesized that soil CO_2–C efflux would be greater in tilled soil than under no-till due to aggregate disruption and higher temperature, and that under no-till, greater soil water content would result in larger biomass production and C inputs to the soil.

MATERIALS AND METHODS

Study Site

The experimental site is located at the Anguil Experimental Station of the Instituto Nacional de Investigaciones Agropecuarias, located in the Semiarid Pampean Region of Argentina (36°30′ S, 63°49′ W, 165 m above sea level). Average rainfall for the period 1913 to 2000 was 664 mm and increased during the last three decades (1970–2000) to 760 mm, with common interannual variability ranging from 400 to 1000 mm and 75% occurring in spring and summer. Annual rainfall was 775 mm in 1998, 938 mm in 1999, 606 mm in 2000, 1150 mm in 2001, 754 mm in 2002, and 450 mm in 2003. During the period from August 2000 to July 2003, annual rainfall averaged 829 mm. Mean annual air temperature was 16°C.

The soil is an Entic Haplustoll with a surface A horizon of 18 cm and a petrocalcic horizon present at 100- to 120-cm depth. The upper 25 cm of the profile is carbonate free and has a loamy texture with 530 g kg^{-1} sand, 94 g kg^{-1} clay, 20.5 g kg^{-1} organic matter, 1.06 g kg^{-1} organic N, 14.9 mg kg^{-1} extractable P (Bray 1), and pH (soil/water) 6.1. Soil field capacity (34.5 kPa) was 280 mm and wilting point (1.5 MPa) was 110 mm in the upper 100 cm of the profile (determined by using porous membrane plates [Klute, 1986]). Before the start of the experiment, the field had been under continuous cropping with conventional tillage (disk tillage) for 20 yr, under a 4-yr rotation: wheat, natural grassland, sorghum [*Sorghum bicolor* (L.) Moench.], and sorghum. In 1998, before the installation of the experiment, an area of 33 by 220 m was delimited in the experimental field and subdivided into 36 units of approximately 200 m^2 each. Samples were taken in the center of each unit to 100-cm depth for an initial characterization of the soil and for the determination of possible spatial autocorrelation, discussed below.

Experimental Design

The tillage experiment was installed with two treatments: no-till, where weeds were controlled using glyphosate [*N*-(phosphonomethyl) glycine], and disk tillage, where the soil was tilled to 15- to 18-cm depth with a disk plow 3 mo before sowing and the seedbed refined using a harrow disk. Each tillage treatment was applied to one single plot (33 by 110 m) that was divided into three subplots (11 by 110 m) for sampling. The crop sequence used was similar to those widespread in the region: wheat in 1998, sunflower (*Helianthus annuus* L.) in 1999, oat + hairy vetch in 2000, corn (*Zea mays* L.) in 2001/2002, wheat in 2002, and oat (as green manure) in 2003. Weeds were controlled with selective herbicides in both tillage treatments. Nitrogen fertilizer (45–50 kg N ha^{-1}) was applied to the oat + hairy vetch, corn, and wheat crops. Harvest was performed mechanically. In spite of the lack of true replications, the experiment was used for CO_2–C evolution and crop C input estimation because it was the only available tillage experiment in the Semiarid Pampean Region of Argentina.

Measurements

In July 2000 and August 2003, six soil samples were taken by subplot from the upper 25-cm soil layer in 5-cm intervals with 244-cm^3 cylinders and bulk density was determined. Samples were ground to pass a 0.5-mm sieve and organic C was also determined (Amato, 1983) to calculate C stock on an areal basis. The remaining straw and root materials were also evaluated by harvesting the remaining straw on the soil surface in 25- by 25-cm microplots, six per subplot. Buried straw and roots were determined on soil samples by a combination of washing and flotation (Böhm, 1979). Residues were dried, weighed, and C was determined (Amato, 1983). Between July 2000 and August 2003, soil CO_2–C fluxes were measured in the field by the static chamber method (Alvarez et al., 1998), with a sampling interval of 38 ± 9 d. Cylindrical polyvinyl chloride chambers, 11-cm diameter by 15-cm height, were pushed 5 cm into the soil, six chambers per subplot, during periods when the soil was bare. During the cycles of crop growth, six long chambers per subplot, 11-cm diameter by 40-cm height, were pushed into the soil to 30 cm before crop emergence and kept free of plants to avoid the inclusion of roots in the chambers. The respiration measured with these long cylinders was considered soil respiration free of root respiration (Alvarez et al., 1996). All chambers were sealed with polyethylene and covered with aluminum foil for insulation.

Vials containing 5 mL of 1 mol L^{-1} NaOH solution captured the CO_2–C emitted by the soil during 48-h periods. The surface area of the vials was 25% of the chamber area. The NaOH solution was titrated against HCl using phenolphthalein (Alvarez et al., 1995b). Chambers sealed in the base were used for blank determinations and CO_2–C captured in these chambers was subtracted from soil respiration. On the dates on which CO_2–C efflux was determined, three soil samples were also taken by subplot to 100-cm depth by 20-cm layers to determine soil water content by the gravimetric method. Soil bulk density was also determined on these samples with a cylinder as indicated above. Nitrate N was analyzed to 60-cm depth with 2 mol L^{-1} KCl extraction and the phenoldisulfonic acid method (Bremmer, 1965). Soil temperature at the 10-cm depth was measured in each subplot with standard thermometers covered with caps. As minimum and maximum soil temperatures occur earlier at the surface and later at deeper layers (Baumhardt et al., 2000; Elias et al., 2004), we registered soil temperature at 0900 and 1600 h, and averaged the results to obtain a gross estimation of the daily mean in the upper soil layers.

Crop production of aboveground biomass was measured by harvesting six microplots (10 m^2 each) per subplot at physiological maturity. Root biomass to the 20-cm depth was determined by removing six columns (3000 cm^3 each) per microplot in the row and in the middle of the furrows. The soil was dispersed in water and sieved though 0.5-mm mesh to retain roots. The plant material was dried at 60°C, ground, and C was determined by wet digestion (Amato, 1983). Total root biomass to 100-cm depth was estimated assuming that roots in the 0- to 20-cm layer accounted for 60% of the total belowground biomass (Jackson et al., 1996). Rhizodeposition, defined as root-derived C remaining in the soil at harvest, originating from decomposition of dead roots, exudates, and sloughed root cells, was estimated to be 8% (Swinnen et al., 1994; Kisselle et al., 2001). Total belowground C input was calculated as the sum of C in roots and rhizodeposition.

Decomposition of corn and wheat residues was evaluated by the litter bag technique (Andrén and Paustian, 1987). Litter bags of 20 by 20 cm and 4-mm mesh size, containing 50 g dry straw, were installed on the soil surface in the no-till treatment or buried to 15-cm depth in the disk tillage treatment and left in the field for up to 900 d. At variable time intervals, six bags were taken from each subplot, the remaining straw material was washed to remove contaminating soil and dried (60°C), and C in the residues was analyzed (Amato, 1983).

Data Analyses

The main goal of all analyses was to test for differences between tillage treatments. Because soil variation is not random (Webster, 2000) and using pseudoreplications inflates the degrees of freedom and increases the probability of Type I error (Hurlbert, 1984; Johnson, 2006), a key issue of all analyses was the use of spatial covariance matrices to reflect the stochastic association among samples, which in turn affects the degrees of freedom for the tests of differences between tillage and no-till. First, we checked for lack of difference in the subplots between both treatments to ensure that all differences at the end of the experiment were due to the treatments. In doing so, the data from the initial soil sampling in 1998 was analyzed with a one-way model with treatment effects and an anisotropic power covariance structure using PROC MIXED (SAS Institute, 1999). Coordinates were the positions of the samples in a two-dimensional grid.

Tillage effects on soil C mineralization were tested using daily and cumulative CO_2–C emissions estimated from a semiparametric mixed model by fitting the day of measurement using B-spline bases (Cantet et al., 2005). To compare treatment effects on C flux, the emission of CO_2–C was estimated by fitting different curves, each with a different covariance matrix, for each treatment. The model can then be written as

$$y_{itk} = \sum_{j=1}^{4} B_{ij}^{(tk)} b_j + e_{itk} \qquad [1]$$

where y_{itk} is the observation of CO_2–C emission from treatment i (i = 1, 2) at time t (t = 1, .., 32) from position k (k = 1, .., 18); $B_{ij}^{(tk)}$ are the cubic B-spline coefficients for observation y_{itk}. There are four cubic B-spline coefficients that add up to 1 for each date of measure. The b_j values are the curve "parameters," which are actually treated as random variables, and e_{ikt} is the error term. The covariance between two different b_j values separated by time lag w, b_j and b_{j+w}, is cov(b_t, b_{t+w}) = $(1 - w/n_x)\sigma_b^2$, where n_x is the number of knots (equal to eight in this application) or supporting points of the B-spline function and σ_b^2 is the variance of the b_j values. This covariance structure, described in detail elsewhere (Cantet et al., 2005), has the property of mimicking a linear decay of the correlation and is more parsimonious than an AR(1) process: there is only one variance parameter (σ_b^2) to estimate for each curve. The variance components (σ_{b1}^2, σ_{b2}^2, σ_e^2) were estimated by restricted maximum likelihood (Patterson and Thompson, 1971) using the expectation–maximization algorithm. Best linear unbiased predictions (BLUPs, Henderson, 1984) of daily values of all these variables were calculated and added to obtain cumulative CO_2–C emission. The difference between the BLUPs of the two tillage systems among sampling dates were tested using the F test with the degrees of freedom corrected by the method of Kenward and Roger (1997). The fitting of the model, the estimation of variance components, and the calculation of the BLUPs and the F tests were performed with a program specially written in FORTRAN. The program is available from R.J.C. Cantet on request. Soil moisture, temperature, and NO_3–N evolution under the tillage treatments were contrasted by the same methodology.

To characterize the dynamics of residue decomposition, we used the following double-exponential decay model (Andrén and Paustian 1987):

$$C_{rem} = C_L \exp\left(-k_L t\right) + C_R \exp\left(-k_R t\right) \qquad [2]$$

where C_{rem} is the remaining residue C, C_L is the labile C fraction in the residue, C_R is the recalcitrant C fraction in the residue, k_L and k_R are decomposition constants, and t is the time of measurement. The remaining residue mass was estimated for each date that CO_2–C flux was measured by using estimated values from the adjusted model. The model was fitted using weighted least squares as implemented in PROC NLIN (SAS Institute, 1999). Comparison of treatments and type of material effects on decomposition was performed by contrasting the parameters of the models using the t-test, as Mueller and Zhao (1995) showed that the estimated parameters of a nonlinear model are asymptotically normal under mild conditions. Remaining residue C was estimated for each date respiration was measured by fitting decomposition models and C inputs from crops. Estimated residue C was the sum of the remaining C from all crops.

Relationships between CO_2–C emission and environmental variables were tested with multiple linear regression techniques using the adjusted R^2 statistic as a decision criterion. Different models were developed for CO_2–C emission estimation. The linear and quadratic functions were tested, as well as a quadratic polynomial surface response model. The covariate model used for surface response (Shen et al., 2003) was

$$CO_2\text{–}C = a_0 + a_1v_1 - a_2v_1^2 + a_3v_2 - a_4v_2^2 + a_5v_1v_2 + \ldots + a_{n-2}v_x - a_{n-1}v_x^2 + a_nv_{x-1}v_x \quad [3]$$

where a_0, …, a_n are regression coefficients and v_0, …, v_n are explanatory variables (soil moisture, temperature, NO_3, organic C, estimated residue C remaining, and tillage treatment as a categorical variable).

The model incorporates linear and quadratic terms for assessing the effects of explanatory variables and their interactions on the response variable. Only the linear term and the interactions were tested for the categorical variable associated with tillage treatments. This formulation has been extensively used in fertilizer experiment evaluation, and positive linear effects and negative quadratic effects are expected (Colwell, 1994). A combination of stepwise, forward and backward regression adjustments was used to obtain the most parsimonious model with the largest R^2. Terms were kept in the final model if the *F* tests were significant at $P < 0.05$ and the entire regression at $P < 0.001$. For simplicity, variables that increased R^2 <0.05% were left out of the model. Autocollineality among explanatory variables was checked by means of the variance inflation factors (Neter et al., 1990). Slopes and intercepts of predicted vs. observed CO_2–C data were compared using the *t*-test.

Crop yields and soil organic C were analyzed with PROC MIXED, using a one-way model with fixed treatments and a spatial power covariance structure. Coordinates used were sampling locations within treatment × block cells. Degrees of freedom for hypothesis testing were corrected by the method of Kenward and Roger (1997). Soil C balance was calculated as the difference between the BLUPs of cumulative CO_2–C emitted from the soil and crop C inputs. The difference between C inputs and outputs was tested by means of a one-way model with fixed treatment effects and the spatial power covariance structure in PROC MIXED.

Initial and final measures of soil C were analyzed with a fixed model with a spatial linear covariance structure, and PROC MIXED was used to fit the model. Effects in the model equation were tillage system, moment of sampling, and soil depth nested within moment of sampling. The differences between tillage systems at the five soil depths sampled were tested using linear contrasts, with degrees of freedom corrected by the procedure suggested by Kenward and Roger (1997).

RESULTS

Initial Characterization of the Site

Means and standard deviations of all variables measured in the soil for the two treatments were similar (Table 1). Patterns of spatial variability were detected for all the characteristics initially evaluated, indicating that measurements were not independent. The spatial covariability was taken into account in all models of analyses, and in all cases the degrees of freedom for the *F* tests were corrected using the procedure of Kenward and Roger (1997). No differences were found, however, between the means of the subplots subjected to no-till and disk tillage ($P < 0.05$). Consequently, further differences may be mainly attributed to treatment effects rather than to plot effects.

Table 1. Soil characteristics of no-till and disk tillage plots before the installation of the experiment (n = 18 by treatment).

Soil variable	No-till plot		Disk tillage plot	
	Avg.	SE	Avg.	SE
Petrocalcic horizon depth, cm	113	4.19	107	4.48
Soil bulk density (0–25 cm), g mL^{-1}	1.18	0.00236	1.18	0.00943
Sand (0–25 cm), g kg^{-1}	512	11.1	546	13.3
Sand (25–100 cm), g kg^{-1}	537	4.90	543	6.93
Silt (0–25 cm), g kg^{-1}	390	8.23	364	13.2
Silt (25–100 cm), g kg^{-1}	393	5.49	380	8.16
Clay (0–25 cm), g kg^{-1}	98.4	5.52	89.5	4.01
Clay (25–100 cm), g kg^{-1}	70.1	3.51	76.7	3.06
Cation exchange capacity (0–25 cm), cmol$_c$ kg^{-1}	18.7	0.209	19.5	0.483
Ca, cmol$_c$ kg^{-1}	12.0	0.212	12.4	0.462
Mg, cmol$_c$ kg^{-1}	0.634	0.148	0.772	0.176
K, cmol$_c$ kg^{-1}	2.81	0.087	2.34	0.077
Na, cmol$_c$ kg^{-1}	0.589	0.060	0.323	0.115
H, cmol$_c$ kg^{-1}	2.72	0.209	3.82	0.375
pH in water, 1:2.5 (0–25 cm)	6.16	0.0735	6.07	0.119
pH in water, 1:2.5 (25–100 cm)	7.1	0.0773	7.41	0.117
Extractable P, Bray 1 (0–25 cm), mg kg^{-1}	15.4	0.898	14.4	0.818
Organic C (0–25 cm), g kg^{-1}	12.1	0.467	11.7	0.486
Organic N (0–25 cm), g kg^{-1}	1.06	0.0257	1.05	0.0218

Soil Water, Temperature, Nitrate, and Carbon Dioxide Flux Evolution

Seasonal fluctuation of soil water content showed similar tendencies under both tillage treatments, with maximum values in the winter and minimum values in the summer (Fig. 1). Higher soil water content ($P < 0.05$) was found under no-till than under disk tillage in the 0- to 20-cm depth on 31 sampling dates, in the 0- to 60-cm depth on 17 sampling dates, and in the 0- to 100-cm depth on 15 sampling dates. Significant differences in soil water content under no-till ranged from 4 to 14 mm in the 0- to 20-cm layer, from 2 to 26 mm in the 0- to 60-cm depth, and from 3 to 36 mm in the 0- to 100-cm layer. During the critical periods of sowing and flowering, the soil had more available water under no-till than under disk tillage. Extreme dry conditions occurred under both tillage treatments during the summer when soil water content dropped below the wilting point in each of the three growing seasons. Average soil water content during the whole experimental period, estimated by averaging daily BLUPs of the variable, resulted in a significantly higher ($P < 0.05$) level for no-till than for tillage.

No significant differences in soil temperature were detected between tillage systems for most sampling dates, although in some of them the temperature tended to be higher under disk tillage (Fig. 2). Only on Date 1, during the initial stages of the growth cycle of corn, did the temperature differ between treatments ($P < 0.05$), being approximately 4°C higher under disk tillage than under no-till. The average temperature during the entire experimental period, which was estimated by averaging daily BLUPs of the variable, was equal to 15.9°C for no-till and 16.8°C for disk plow, the difference being nonsignificant.

Soil NO_3 variation with time showed trends similar to water content (Fig. 3). Higher levels of NO_3 were found during

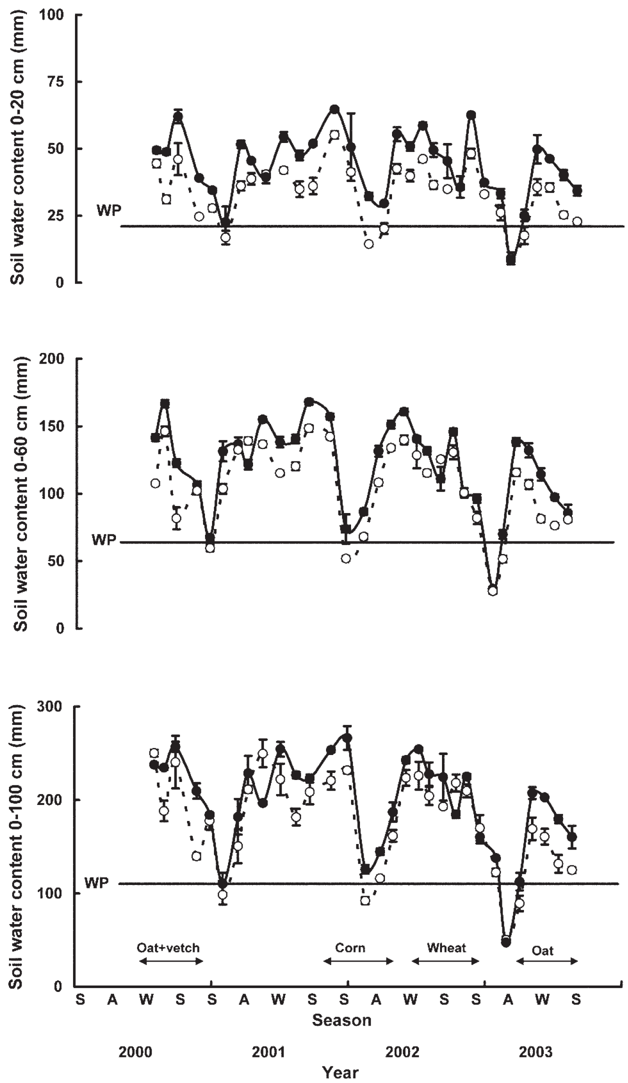

Fig. 1. Evolution of soil water content as a function of season of the year. S, A, W, and S are summer, autumn, winter, and spring, respectively; filled circles are no-till, empty circles are disk tillage; WP is wilting point. Crop growing cycles are indicated by horizontal arrows. Vertical bars represent standard errors.

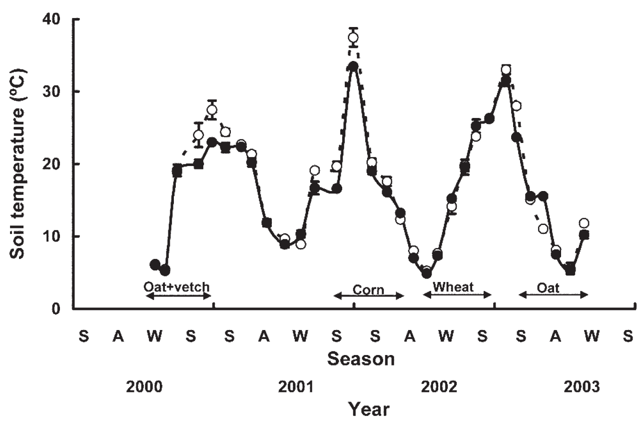

Fig. 2. Evolution of mean soil temperature (10 cm) as a function of season of the year. S, A, W, and S are summer, autumn, winter, and spring, respectively; filled circles are no-till, empty circles are disk tillage. Crop growing cycles are indicated by horizontal arrows. Vertical bars represent standard errors.

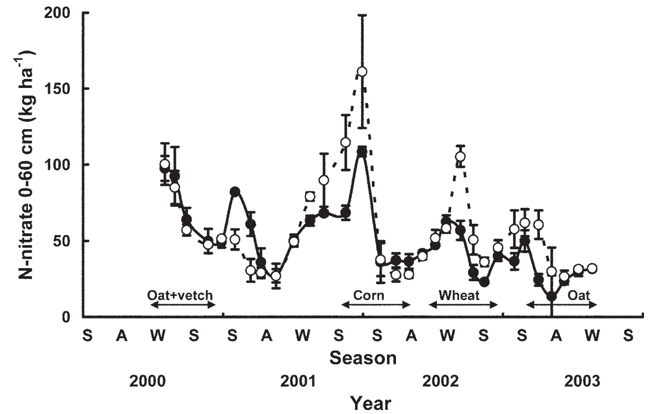

Fig. 3. Evolution of NO_3–N as a function of season of the year. S, A, W, and S are summer, autumn, winter, and spring, respectively; filled circles are no-till, empty circles are disk tillage. Crop growing cycles are indicated by horizontal arrows. Vertical bars represent standard errors.

the initial stages of crop development, which then decreased as the growing season advanced. Significant differences ($P < 0.05$) between treatments were detected on 16 sampling dates, and on 14 of them disk tillage yielded a higher value of soil NO_3 than no-till.

Soil CO_2–C efflux showed a strong seasonal fluctuation (Fig. 4), with low values in the winter (common range 4–10 kg C ha^{-1} d^{-1}) and high values during the end of the spring–summer period (common range 12–19 kg C ha^{-1} d^{-1}). Significant differences ($P < 0.05$) were detected in CO_2–C emissions between treatments in only 6 out of the 31 sampling dates, with higher CO_2–C emission rates under disk tillage. These differences occurred during the tillage period before corn sowing and during the initial stages of the growth of corn. No differences were detected in CO_2–C emissions between tillage treatments during the growing cycles of the winter crops. Disk tillage tended to produce more CO_2–C than no-till in the first 2 yr of the study; this trend reverted during the last year of the experiment. The estimated total CO_2–C emission, which was calculated by adding all daily BLUPs of CO_2–C production, were 4.04 Mg C ha^{-1} yr^{-1} under no-till and 4.27 Mg C ha^{-1} yr^{-1} under disk plow ($P < 0.05$).

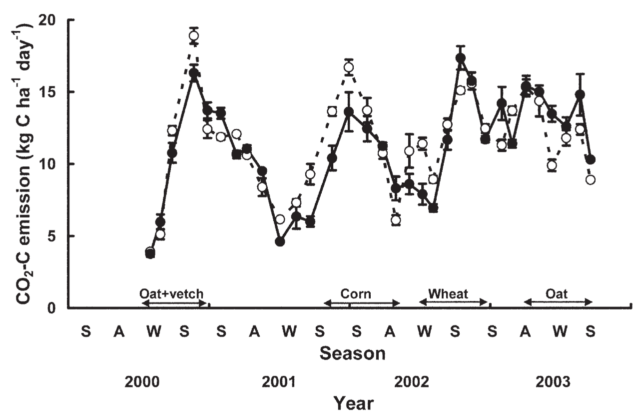

Fig. 4. Evolution of CO_2–C emission from soil as a function of season of the year. S, A, W, and S are summer, autumn, winter, and spring, respectively; filled circles are no-till, empty circles are disk tillage. Crop growing cycles are indicated by horizontal arrows. Vertical bars represent standard errors.

Table 2. Regression and determination coefficients for labile and recalcitrant C and their decomposition constants (*k*) of the double exponential model from residue decomposition data.

Tillage system	Regression coefficient				R^2
	Labile C	k_L	Recalcitrant C	k_R	
No-till	32.2	0.0133 a†	67.4 a	0.00114 a	0.955
Disk tillage	33.5	0.0432 b	66.5 a	0.00218 b	0.962

† Different letters indicate significant differences between tillage treatments ($P < 0.01$).

Residue Decomposition and Carbon Dioxide Flux Model

Decomposition kinetics were well depicted by the double-exponential decay model (Table 2). After an initial phase of rapid decomposition, the process became slower. Residue decomposition was faster under disk tillage than under no-till (Fig. 5). No significant differences were detected between decomposition rates of corn and wheat materials (data not presented), but the decomposition constants of both labile and recalcitrant residue C components (k_L and k_R) were significantly greater when residues were incorporated into the soil in the disk tillage treatment than under no-till. The partition of C between these two pools was similar in both materials, the recalcitrant pool being double the labile one.

Carbon dioxide efflux was positively correlated with soil temperature and estimated residue C remaining in the soil. On the other hand, CO_2 values were negatively correlated with soil water content and NO_3–N in the soil (Table 3). The explanatory variable more correlated to CO_2–C emission was soil temperature, accounting for 30% of the variation. The CO_2–C emission increased during the warmer periods of the year. Due to the decrease in soil water content during the summer, temperature and water content were negatively correlated ($R^2 = 0.30$, $P < 0.001$). Consequently, the negative relationship between CO_2–C efflux and soil water content seemed to be an indirect effect on the CO_2–C efflux from drying of the soil during the summer, rather than the impact of an excess of soil water content on CO_2–C emission.

Multiple regression techniques allowed development of a model that accounted for 55% of the variation in CO_2–C efflux (Fig. 6):

Table 3. Coefficients of determination and estimated parameters for the regression of soil CO_2–C efflux on soil properties (*n* = 186).

Variable	Intercept	Linear term	Quadratic term	R^2
Tillage treatment†	–‡	–	–	–
Soil C	–	–	–	–
Crop residue C	9.9	0.00059	–	0.07
Soil temperature	5.7	0.48	−0.0068	0.31
Soil water content (0–20 cm)	15	−0.095	–	0.12
Soil water content (0–60 cm)	15	–	−0.00022	0.20
Soil water content (0–100 cm)	11	−0.048	−0.00023	0.23
NO_3–N (0–20 cm)	15	−0.28	0.0032	0.17
NO_3–N (0–60 cm)	16	−0.14	0.00074	0.14

† Tillage treatments were taken as a categorical variable.
‡ Nonsignificant term ($P < 0.05$)

$$CO_2\text{–C} = 3.7 + 0.0031\text{RC} - 0.00000028\text{RC}^2 + 0.48\text{ST} - 0.012\text{ST}^2 - 0.059\text{NN} + 0.029\text{TSNN} - 0.000015\text{RCNN} + 0.0027\text{STNN} \quad [4]$$

where RC is C in the residue, ST is the temperature of the soil, NN is the level of NO_3–N in the 0- to 60-cm layer, and TS is the tillage system (no-till, 0; disk tillage, 1).

The model predicts a strong positive effect of temperature on CO_2–C efflux and also positive but weak effects of tillage system and remaining C in the residue. Soil NO_3–N had a small and negative effect on CO_2–C emission.

Carbon Inputs and Soil Carbon Balance

Grain yield, biomass production, and C inputs to the soil were greater under no-till than under disk tillage for all the crops included in the rotation (Table 4). Average C inputs differed ($P < 0.05$) between tillage treatments, rounding to 4 Mg C ha^{-1} yr^{-1} under no-till and 3 Mg C ha^{-1} yr^{-1} under disk tillage. Belowground C input accounted for about 35% of these amounts. Neither was the ratio of straw biomass/grain biomass affected by tillage, averaging 2.0 for oat + hairy vetch and wheat and 1.0 for corn, nor was the ratio straw biomass/root biomass influenced by treatment, with average values of 1.2 to 1.4 for oat + hairy vetch and corn, and 2.3 to 2.7 for wheat and oat.

The estimated C balance was negative when the soil was managed under tillage. This result was a consequence of lower C inputs to the soil in the tillage treatment that were not equiv-

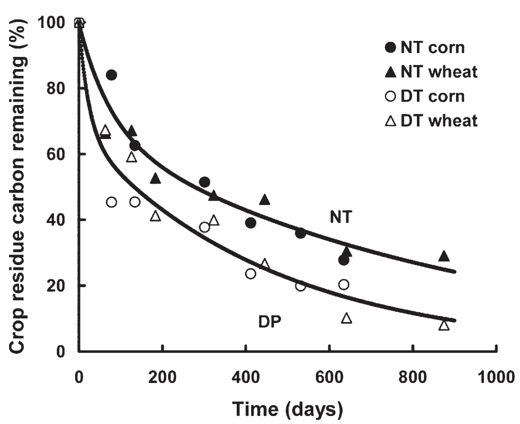

Fig. 5. Crop residue C remaining as a function of time under different tillage systems. NT is no-till, DT is disk tillage. Lines represent double exponential decay model fits to each tillage system.

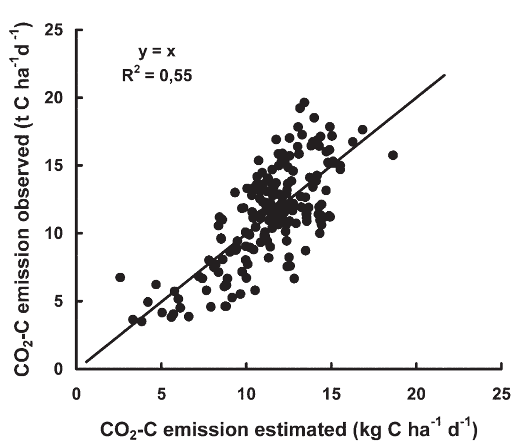

Fig. 6. The CO_2–C emission observed in the field vs. the CO_2–C emission estimated by the model from Eq. [1].

Table 4. Dry matter production and C inputs from crops under two tillage systems.

Crop and year	Tillage system	Straw		Roots + rhizodeposition		Grain	
		Dry matter	C	Dry matter	C	Dry matter	C
		kg ha^{-1}					
Oat + hairy vetch 2000	Disk tillage	1,940 a†	720 a	1,390 a	460 a	970 a	340 a
	No-till	2,990 b	1130 b	2,130 b	800 b	1,500 b	520 b
Corn 2001/2002	Disk tillage	6,290 a	2330 a	5,240 a	1790 a	6,140 a	2170 a
	No-till	8,030 b	3320 b	6,670 b	2440 b	8,160 b	2990 b
Wheat 2002	Disk tillage	5,330 a	1930 a	2,100 a	690 a	2,600 a	920 a
	No-till	5,880 b	2400 b	1,990 a	690 a	3,340 b	1130 b
Oat 2003	Disk tillage	2,130 a	790 a	890 a	350 a		
	No-till	2,470 b	930 b	1,080 b	450 b		
Total	Disk tillage	15,700 a	5770 a	9,600 a	3300 a	9,720 a	3440 a
	No-till	19,400 b	7780 b	11,900 b	4370 b	13,000 b	4640 b

† Different letters indicate significant differences between tillage treatments ($P < 0.01$).

alent to CO_2–C emissions (Fig. 7). Under no-till, the soil C balance was near equilibrium during the experimental period. Tillage had no impact on the overall flux of CO_2–C during the 3-yr study, but the increase in biomass production under no-till determined C inputs that counteracted CO_2–C emissions.

Soil Carbon Stock

There were no detectable changes of soil C content between the 2000 and 2003 sampling dates. The level of soil C for the no-till treatment, however, was significantly higher ($P < 0.01$) than for the tillage treatment up to 15-cm depth (soil layers of 0–5, 5–10, and 10–15 cm, Fig. 8); the differences in deeper layers (25–20 and 20–25 cm) were not significant ($P > 0.05$). Organic C sequestered in the 0- to 25-cm stratum was 32.3 Mg C ha^{-1} under no-till and 26.9 Mg C ha^{-1} under disk tillage ($P < 0.05$). Soil bulk density showed no differences between sampling dates. It was greater under no-till than under disk tillage ($P < 0.05$) only in the 0- to 5-cm layer, with no differences in deeper layers (data not presented). When bulk density for the entire soil layer at 0 to 25 cm was considered, there was no difference between tilled and untilled soils. Consequently, when soil C stocks were calculated on an equivalent soil mass basis, similar results were obtained as when expressing data to a fixed depth.

DISCUSSION

A major goal of no-till in semiarid to subhumid environments is to produce surface residue covering for the control of erosion and for water conservation (Unger et al., 1988). In these environments, runoff and evaporation must be minimized and soil water available for transpiration maximized. Mulch cover induces increased infiltration and reduced surface evaporation (Unger and McCalla, 1980; Unger et al., 1988).

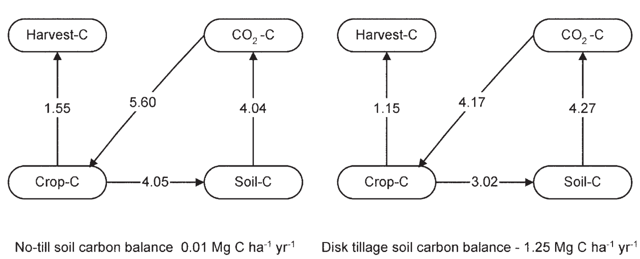

Fig. 7. Soil C balance under two tillage systems. Numbers on arrows indicate C fluxes between pools (Mg ha^{-1} yr^{-1}).

The temperature is lower in soils with surface mulch, especially during the warm periods of the year and before crops shade the soil (Grant et al., 1990), leading to reduced evaporation. In our experiment, the presence of surface mulch contrasted between tillage treatments since the initiation of the field measurements. Nevertheless, the mulch produced only minor differences in soil temperature between treatments because the fallow periods occurred in the winter. During the summers, soil was covered by crops or post-harvest residues under both tillage treatments, except during the initial phase of the corn crop. Consequently, differences in the availability of soil water appeared to be mainly the consequence of greater infiltration under no-till than tilled soil.

In temperate and cold-temperate humid ecosystems, temperature greatly impacts soil respiration (Alvarez and Alvarez, 2001; Tufekcioglu et al., 2001). In arid and semiarid regions, soil moisture restricts respiration and interacts with temperature in the regulation of CO_2–C emission (Frank et al., 2002; Sánchez et al., 2003). In some cases in arid environments, temperature may be the only abiotic factor correlated with soil CO_2–C flux when the rainy season coincides with summer and the variations in soil water content are not very high during the year (Kessavalou et al., 1998). We did not detect significant positive moisture effects on soil CO_2–C fluxes or interactions between temperature and moisture in the Pampean environment of our experiment. The rainy season in the semiarid Pampas falls during the end of the spring–summer period. Soil moisture decreased in the summer due to the higher atmospheric water demand, but not enough to generate a strong detectable CO_2–C efflux reduction during the 3-yr experimental period.

The tillage effect on the annual CO_2–C emitted by the soil depends on environmental conditions. In some cases, it has been reported that there was an increase in soil respiration in plowed soils compared with no-till soils (Kessavalou et al., 1998; Curtin et al., 2000), which had been attributed to a more intense C turnover in plowed soils (Franzluebbers et al., 1998) or a slower residue decomposition rate under no-till (Fortin et al., 1996). In other situations, no effect of tillage treatments on soil respiration was observed (Alvarez et al., 1995b, 1998) or it was reported that higher CO_2–C effluxes occurred under no-till than under plowed soils (Franzluebbers et al., 1995; Wagal et al., 1998). This latter result has been interpreted as the consequence of labile C compound accumulation in no-till soil after some years from installation and the subsequent mineralization of that C (Alvarez et al., 1998). We found significant but small differences between tillage treatments in annual CO_2–C fluxes, possibly as the result of the small differences in soil temperature under the rotation used. Only during the fallow period and the initial stages of the corn growing season was CO_2–C emission higher in the tilled soil than the no-till soil. This period coincided with some higher soil temperatures in the disk tillage treatment. Since the installation of the experiment and the initiation of CO_2–C measurements (3 yr), significant effects of tillage treatment on soil C stocks were found. Consequently, mineralizable C pools were possibly greater under no-till, and this factor counteracts tillage effects on soil C mineralization. Crop root contribution to soil respiration may be as high as 40 to 60% of total soil respiration during periods of intense growth (Akinremi et al., 1999; Rochette et al., 1999). In our CO_2–C efflux measurements, roots were excluded from respiration chambers and CO_2–C was emitted mainly from decomposing residues and organic C, so the overall mineralization of organic soil components was of the same magnitude under both tillage systems.

Our estimations of residue C remaining in the soil may be biased because we applied decomposition models fitted to wheat and corn residue data to all the inputs produced during the experiment. The decomposition rate is faster as the C/N ratio of the decomposing material increases (Silver and Miya, 2001) because C mineralization is regulated by the residue N content (Jensen et al., 2005). The C/N ratios of residues in the soil from wheat, corn, and oat materials at the initiation of CO_2–C measurements were similar, ranging from 40 to 50. Only hairy vetch residues had a lower C/N ratio (~30), but much more of the biomass produced by the oat + hairy vetch crop was from oat, so the bias in remaining C estimations seemed to be small.

The negative relationship observed between CO_2–C emission and NO_3–N in the soil is not clear. One possible explanation is that when the soil NO_3–N level is high, microbial biomass becomes more efficient in producing living cells, growth yield rises, and maintenance respiration decreases (Knapp et al., 1983). Conversely, when the NO_3–N level is low, more polysaccharides are synthesized and then consumed by respiration, increasing soil CO_2–C efflux.

Tillage may induce an increase in soil mineral N content (Groffman et al., 1987; Davies et al., 2001) as the consequence of two possible mechanisms: the intensification of organic N mineralization (Grace et al., 1993) and a more rapid straw decomposition and N liberation (Lachnicht et al., 2004). In our experiment, the N content from NO_3 was significantly greater (10–50 kg N ha^{-1}) at the initial phases of growth of corn, wheat, and oat in the tilled soil than under no-till. Nitrogen fertilization did not compensate these differences in soil fertility between treatments. In spite of the higher N availability under the disk tillage treatment, crop yield and biomass were greater under no-till. This result may be due to the fact that soil productivity in this semiarid environment is mostly dependent on water availability, so that no-till produces higher yields and C inputs to the soil.

The results of the C balance from the current research must be taken with caution. Carbon inputs to the soil are difficult to measure and great uncertainties exist, especially in the estimation of belowground C allocation and in the separation of the root and microbial sources of CO_2–C (Rees et al., 2005). For example, the root biomass distribution with depth may be influenced by the tillage treatment, with deeper root systems under conventional tillage (Dwyer et al., 1996; Martino and Shaykewich, 1994). Assuming that the root mass in the 0- to 20-cm soil layer represented 50% of the total roots under disk tillage and 70% under no-till, rather than an equal proportion of 60% for both tillage treatments as assumed in this research, the soil C balance would be −0.23 Mg ha^{-1} under no-till and −1.03 Mg ha^{-1} under disk tillage. In temperate arable and grassland ecosystems, total soil respiration (microbial plus root respiration) ranges from 4 to 26 Mg C ha^{-1} yr^{-1}, and net ecosystem exchange of C is about one order of magnitude lower, with negative values in most cases (Rees et al., 2005). Under the humid climatic scenarios of the Rolling Pampas, a neighboring region to the Semiarid Pampas with rich organic matter soils, the CO_2–C efflux emitted from residue decomposition and organic matter mineralization has been estimated to range from 7.8 to 11.6 Mg C ha^{-1} yr^{-1} depending on management, with a soil C balance that is negative under tilled and no-till situations (Alvarez et al., 1995b, 1998). The low CO_2–C efflux of the Semiarid Pampas soil may be attributed to the low soil organic matter content. A consequence of this low C efflux is that inputs necessary to counterbalance the loss of C through decomposition and mineralization of organic substrates are small. Because of the method used in our experiment to prevent the inclusion of root respiration, some of the CO_2–C trapped in the long chambers used during the growing seasons of the crops may have been produced by the respiration of deep roots below 30-cm depth. Contribution to CO_2–C determinations by these deep roots biases the estimation of soil C decomposition and mineralization, and seems to be on the order of 0.2 to 0.3 Mg C ha^{-1} yr^{-1} (Alvarez et al., 1998), leading to an overestimation of our soil C losses.

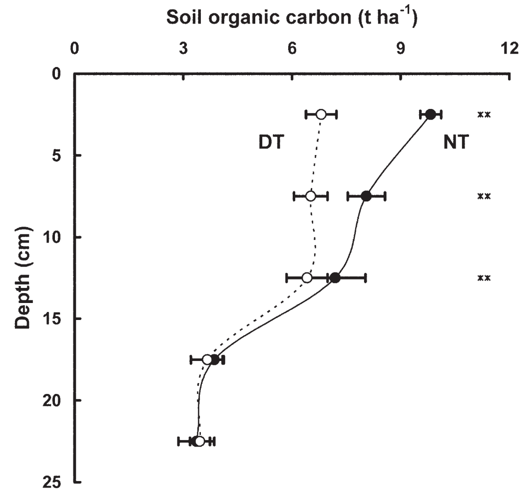

Fig. 8. Soil organic C distribution as a function of depth. Values are means of 2000 and 2003 sampling dates. NT is no-till, DP is disk tillage. Bars represent standard errors. ** Significant difference between treatments (P < 0.01).

Soil bulk density is usually higher under no-till than under tilled soils, both at the surface and throughout the entire tillage depth (Chen et al., 1998). Soil densification under no-till is texture dependent, with lower increases in bulk density induced by no-till in sandy soils (Chen et al., 1998). The soil of our experiment had high sand-plus-silt content and was not susceptible to compaction after the introduction of no-till, except in the upper 5-cm layer.

Tillage incorporates crop residues, producing an even distribution throughout the tillage depth; meanwhile, under no-till, residue inputs are concentrated in the upper centimeters (Staricka et al., 1991). As a consequence, soil C increases near the surface of no-till soil, mainly due to the accumulation of labile organic matter components (Alvarez et al., 1995a; Wander et al., 1998; Balesdent et al., 2000). Taking into account the entire tillage depth, the C sequestration rate under no-till has been described in some revisions as an S-shape phenomenon related to time, which peaks around 5 to 15 yr after the beginning of no-till and reaches steady-state conditions around 10 to 25 yr (West and Post, 2002; Alvarez, 2005; Steinbach and Alvarez, 2005). An overall increase in soil C under no-till was detected in our experiment 3 yr after its introduction. The estimations of the C balance showed losses of C from tilled soil between 3 and 6 yr of the experiment and no change under no-till, but these effects were not detected when contrasting C stock data in the soil. Soil C stock estimations were highly variable, with coefficients of variation ranging from 7 to 22% between subplots of the same tillage treatment and sampling date. This high dispersion was not the result of variability in soil bulk determinations (CV 2–4%), but was due to variability in the concentration of C. This variation prevented the detection of significant trends in soil C changes with time. Conversely, CO_2–C flux and C input determinations were much more sensitive at detecting differences in soil C dynamics and the results allowed predicting tillage effects on the level of organic matter in the soil.

The soil C balance was affected by tillage system through the effects of management on crop productivity. Carbon dioxide emitted by the degradation processes of organic soil components was not affected by tillage under our environmental conditions. It is expected that the adoption of no-till in this area will sequester C in comparison to tilled soils, in which C losses may occur.

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REFERENCES

Akinremi, O.O., S.M. McGinn, and H.D.J. McLean. 1999. Effects of soil temperature and moisture on soil respiration in barley and fallow plots. Can. J. Soil Sci. 79:5–13.

Alvarez, R. 2005. A review of nitrogen fertilization and conservation tillage effects on soil C storage. Soil Use Manage. 21:38–52.

Alvarez, R., and C.R. Alvarez. 2001. Temperature as regulator of soil carbon dioxide production in the humid pampa of Argentina. Biol. Fertil. Soils 34:282–285.

Alvarez, R., C.R. Alvarez, and G. Lorenzo. 2001. CO_2–C fluxes following tillage from a Mollisol in the Argentine rolling pampa. Eur. J. Soil Biol. 37:161–166.

Alvarez, R., R. Díaz, N. Barbero, O.J. Santanatoglia, and L. Blotta. 1995a. Soil organic carbon, microbial biomass and CO_2–C production from three systems. Soil Tillage Res. 33:17–28.

Alvarez, R., and R.S. Lavado. 1998. Climate, organic matter and clay content relationships in the Pampa and Chaco soils, Argentina. Geoderma 83:127–141.

Alvarez, R., M. Russo, P. Prystupa, J. Sheiner, and L. Blotta. 1998. Soil carbon pools under conventional and no-tillage systems in the Argentine rolling pampa. Agron. J. 90:138–143.

Alvarez, R., O. Santanatoglia, and R. García. 1996. Plant and microbial contribution to soil respiration under zero and disc tillage. Eur. J. Soil Biol. 32:173–177.

Alvarez, R., O.J. Santanatoglia, and R. Garcia. 1995b. Soil respiration and carbon inputs from crops in a wheat–soyabean rotation under different tillage systems. Soil Use Manage. 11:45–50.

Amato, M. 1983. Determination of ^{12}C and ^{14}C in plant and soil. Soil Biol. Biochem. 15:611–612.

Andrén, O., and K. Paustian. 1987. Barley straw decomposition in the field: A comparison of models. Ecology 68:1190–1200.

Balesdent, J., C. Chenu, and M. Balabane. 2000. Relationship of soil organic matter dynamics to physical protection and tillage. Soil Tillage Res. 53:215–230.

Baumhardt, R.L, R.J. Lascano, and S.R. Evett. 2000. Soil material, temperature, and salinity effects on calibration of multisensor capacitance probes. Soil Sci. Soc. Am. J. 64:1940–1946.

Bernardos, J.N., E.F. Viglizzo, V. Jouvet, F.A. Lértora, A.J. Pordomingo, and F.D. Cid. 2001. The use of EPIC model to study the agroecological change during 93 years of farming transformation in the Argentine Pampas. Agric. Syst. 69:215–234.

Böhm, W. 1979. Methods of studying root systems. Ecol. Stud. 33:115–124.

Bossuyt, H., J. Six, and P.F. Hendrix. 2002. Aggregate-protected carbon in no-tillage and conventional tillage agroecosystems using carbon-14 labeled plant residue. Soil Sci. Soc. Am. J. 66:1965–1973.

Bremmer, J.M. 1965. Inorganic forms of nitrogen. p. 1179–1237. *In* C.A. Black et al. (ed.) Methods of soil analysis. Part 2. Agron. Monogr. 9. SSSA and ASA, Madison, WI.

Buschiazzo, D.E., J.L. Panigatti, and P.W. Unger. 1998. Tillage effects on soil properties and crop production in the subhumid and semiarid Argentinean Pampas. Soil Tillage Res. 49:105–116.

Campbell, C.A., B.G. McConkey, V.O. Biederbeck, R.P. Zentner, S. Tessier, and D.L. Hahn. 1997. Tillage and fallow frequency effects on selected soil quality attributes in a coarse-textured Brown Chernozem. Can. J. Soil Sci. 77:497–505.

Cantet, R.J.C., A.N. Birchmeier, A.W. Canaza Cayo, and C. Fioretti. 2005. Semiparametric animal models via penalized splines as alternatives to contemporary groups. J. Anim. Sci. 83:2482–2494.

Chen, Y., S. Tessier, and J. Rouffigant. 1998. Soil bulk density estimations for tillage systems and soil textures. Trans. ASAE 41:1601–1610.

Colwell, J.D. 1994. Estimating fertilizer requirements: A quantitative approach. CAB Int., Wallingford, UK.

Curtin, D., F. Selles, H. Wang, C.A. Campbell, and V.O. Biederbeck. 1998. Carbon dioxide emissions and transformation of soil carbon and nitrogen during wheat straw decomposition. Soil Sci. Soc. Am. J. 62:1035–1041.

Curtin, D., H. Wang, F. Selles, B.G. McConkey, and C.A. Campbell. 2000. Tillage effects on C fluxes in continuous wheat and fallow–wheat rotations. Soil Sci. Soc. Am. J. 64:2080–2086.

Dao, T.H. 1998. Tillage and crop residue effects on carbon dioxide evolution and C storage in a Paleustoll. Soil Sci. Soc. Am. J. 62:250–256.

Davies, M.G., K.A. Smith, and A.J.A. Vinten. 2001. The mineralisation and fate of nitrogen following ploughing of grass and grass–clover swards. Biol. Fertil. Soils 33:423–434.

Denef, K., J. Six, R. Merckx, and K. Paustian. 2004. Carbon sequestration in microaggregates of no-tillage soils with different clay mineralogy. Soil Sci. Soc. Am. J. 68:1935–1944.

Díaz-Zorita, M., G.A. Duarte, and J.H. Grove. 2002. A review of no-till systems and soil management for sustainable crop production in the subhumic and semiarid Pampas of Argentina. Soil Tillage Res. 65:1–18.

Duiker, S.W., and R. Lal. 1999. Crop residue and tillage effects on carbon sequestration in a Luvisol in central Ohio. Soil Tillage Res. 52:73–81.

Dwyer, L.M., B.L. Ma, D.W. Stewart, H.N. Hayhoe, D. Balchin, J.L.B. Culley, and M. McGovern. 1996. Root mass distribution under conventional and conservation tillage. Can. J. Soil Sci. 76:23–28.

Elias, E.A., R. Cichota, H.H. Torriani, and Q. de Jong van Lier. 2004. Analytical soil-temperature model: Correction for temporal variation of daily amplitude. Soil Sci. Soc. Am. J. 68: 784–788.

Follett, R.F., J.M. Kimble, and R. Lal (ed.). 2001. The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect. Lewis Publ., Boca Raton, FL.

Fortin, M.C., P. Rochette, and E. Pattey. 1996. Soil carbon dioxide fluxes from conventional and no-tillage small-grain cropping systems. Soil Sci. Soc. Am. J. 60:1541–1547.

Frank, A.B., M.A. Liebig, and J.D. Hanson. 2002. Soil carbon dioxide fluxes

in northern semiarid grasslands. Soil Biol. Biochem. 34:1235–1241.
Franzluebbers, A.J. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil Tillage Res. 66:197–205.
Franzluebbers, A.J., F.M. Hons, and D.A. Zuberer. 1995. Tillage induced seasonal changes in soil physical properties affecting soil CO_2 evolution under intensive cropping. Soil Tillage Res. 34:41–60.
Franzluebbers, A.J., F.M. Hons, and D.A. Zuberer. 1998. In situ and potential CO_2 evolution from a Fluventic Ustochrept in southcentral Texas as affected by tillage and cropping intensity. Soil Tillage Res. 47: 303–308.
Grace, P.R., I.C. MacRae, and R.J.K. Myers. 1993. Temporal changes in microbial biomass and N mineralization under simulated field cultivation. Soil Biol. Biochem. 25:1745–1753.
Grant, R.F., R.C. Izaurralde, and D.S. Chanasyk. 1990. Soil temperature under conventional and minimum tillage: Simulation and experimental verification. Can. J. Soil Sci. 70:289–304.
Groffman, P.M., P.F. Hendrix, and D.A. Crossley. 1987. Nitrogen dynamics in conventional and no-tillage agroecosystems with inorganic fertilizer or legume nitrogen inputs. Plant Soil 97:315–332.
Hall, A.J., C.M. Rebella, C.M. Ghersa, and J.P. Culot. 1992. Field crop systems of the Pampas. p. 413–450. *In* C.J. Pearson (ed.) Field Crop Ecosyst. World 18. Elsevier, Amsterdam.
Henderson, C.R. 1984. Applications of linear models in animal breeding. Univ. of Guelph, Guelph, ON, Canada.
Hernanz, J.L., R. López, L. Navarrete, and V. Sánchez-Girón. 2002. Long-term effects of tillage systems and rotations on soil structural stability and organic carbon stratification in semiarid central Spain. Soil Tillage Res. 66:129–141.
Hevia, G.G., D.E. Buschiazzo, E.N. Hepper, A.M. Urioste, and E.L. Antón. 2003. Organic matter in size fractions of soils of the semiarid Argentina: Effects of climate, soil texture and management. Geoderma 116:265–277.
Hurlbert, S.H. 1984. Pseudoreplicacion and the design of ecological field experiments. Ecol. Monogr. 54:187–211.
INDEC. 2003. National Institute of Statistics and Censuses. Available at www.indec.mecon.ar/ (verified 1 May 2008). INDEC, Buenos Aires.
Jackson, R.B., J. Canadell, J.R. Ehleringer, H.A. Mooney, O.A. Sala, and E.D. Schulze. 1996. A global analysis of root distributions for terrestrial biomes. Oecologia 108:389–411.
Jensen, L.S., T. Salo, F. Palmason, T.A. Breland, T.M. Henriksen, B. Stenberg, A. Pedersen, C. Lundström, and M. Esala. 2005. Influence of biochemical quality on C and N mineralization from a broad variety of plant materials in soil. Plant Soil 273:307–326.
Johnson, D.H. 2006. The many faces of replication. Crop Sci. 46:2486–2491.
Karamanos, A.J., D. Bilalis, and N. Sidiras. 2004. Effects of reduced tillage and fertilization practices on soil characteristics, plant water status, growth and yield of upland cotton. J. Agron. Crop Sci. 190:262–276.
Kenward, M.G., and J.H. Roger. 1997. Small sample inference for fixed effects from restricted maximum likelihood. Biometrics 53:983–997.
Kessavalou, A., A.R. Mosier, J.W. Doran, R.A. Drijber, D.J. Lyon, and O. Heinemeyer. 1998. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter wheat–fallow tillage management. J. Environ. Qual. 27:1094–1104.
Kisselle, K.W., C.J. Garrett, S. Fu, P.F. Hendrix, D.A. Crossley, D.C. Coleman, and R.L. Potter. 2001. Budgets for root-derived C and litter-derived C: Comparison between conventional tillage and no-tillage soils. Soil Biol. Biochem. 33:1067–1075.
Klute, A. 1986. Water retention: Laboratory methods. p. 635–662. *In* A. Klute et al. (ed.) Methods of soil analysis. Part 1. Physical and mineralogical methods. 2nd ed. SSSA Book Ser. 5. SSSA, Madison, WI.
Knapp, E.B., L.F. Elliott, and G.S. Campbell. 1983. Carbon, nitrogen and microbial biomass interrelationships during the decomposition of wheat straw: A mechanistic simulation model. Soil Biol. Biochem. 15:455–461.
Kristensen, H.L., G.W. McCarty, and J.J. Meisinger. 2000. Effects of soil disturbance on mineralization of organic soil nitrogen. Soil Sci. Soc. Am. J. 64:371–378.
Lachnicht, S.L., P.F. Hendrix, R.L. Potter, D.C. Coleman, and D.A. Crossley. 2004. Winter decomposition of transgenic cotton residue in conventional-till and no-till systems. Appl. Soil Ecol. 27:135–142.
Lal, R., J.M. Kimble, R.F. Follett, and C.V. Cole. 1999. The potential of U.S. cropland to sequester C and mitigate the greenhouse effect. Lewis Publ., Boca Raton, FL.
Martino, D.L., and C.F. Shaykewich. 1994. Root penetration profiles of wheat and barley as affected by soil penetration resistance in field conditions. Can. J. Soil Sci. 74:193–200.
Mueller, H.G., and P.L. Zhao. 1995. On a semiparametric variance function model and a test for heteroscedasticity. Ann. Stat. 23:946–967.
Neter, J., W. Wasserman, and M.H. Kutner. 1990. Applied linear statistical models. 3rd ed. Richard D. Irwin, Homewood, IL.
Quiroga, A.R., D.E. Buschiazzo, and N. Peinemann. 1996. Soil organic matter particle size fractions in soils of the semiarid Argentinean Pampas. Soil Sci. 161:104–108.
Patterson, H.D., and R. Thompson. 1971. Recovery of inter-block information when block sizes are unequal. Biometrika 58:545–554.
Rasse, D.P., and J.M. Smucker. 1999. Tillage effects on soil nitrogen and plant biomass in a corn–alfalfa rotation. J. Environ. Qual. 28:873–880.
Rees, R.M., I.J. Bingham, J.A. Baddeley, and C.A. Watson. 2005. The role of plants and land management in sequestering soil C in temperate arable and grassland ecosystems. Geoderma 128:130–154.
Rochette, P., and D.A. Angers. 1999. Soil surface carbon dioxide fluxes induced by spring, summer, and fall moldboard plowing in a sandy loam. Soil Sci. Soc. Am. J. 63:621–628.
Rochette, P., L.B. Flanagan, and E.G. Gregorich. 1999. Separating soil respiration into plant and soil components using analysis of the natural abundance of carbon-13. Soil Sci. Soc. Am. J. 63:1207–1213.
Sánchez, M.L., M.I. Ozores, M.J. López, R. Colle, B. De Torre, M.A. García, and I. Pérez. 2003. Soil CO_2 fluxes beneath barley on the central Spanish plateau. Agric. For. Meteorol. 118:85–95.
SAS Institute. 1999. SAS/STAT user's guide. Version 8 ed. SAS Inst., Cary, NC.
Satorre, E.H., and G.A. Slafer. 1999. Wheat production systems of the Pampas. p. 333–348. *In* E.M. Satorre and G.A. Slafer (ed.) Wheat: Ecology and physiology of yield determination. Haworth Press, New York.
Shen, J.L., R.F. Zhang, Z. Rengel, and C. Tang. 2003. Orthogonal polynomial models to describe yield response of rice to nitrogen and phosphorus at different levels of soil fertility. Nutr. Cycling Agroecosyst. 65:243–252.
Silver, W.L., and R.K. Miya. 2001. Global patterns in root decomposition: Comparisons of climate and litter quality effects. Oecologia 129:407–419.
Staricka, J.A., R.R. Allmaras, and W.W. Nelson. 1991. Spatial variation of crop residue incorporated by tillage. Soil Sci. Soc. Am. J. 55:1668–1674.
Steinbach, H.S., and R. Alvarez. 2005. Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in pampean agroecosystems. J. Environ. Qual. 35:3–13.
Swinnen, J., J.A. van Veen, and R. Merckx. 1994. Rhizosphere carbon fluxes in field-grown spring wheat: Model calculations based on ^{14}C partitioning after pulse-labelling. Soil Biol. Biochem. 26:171–182.
Tufekcioglu, A., J.W. Raich, T.M. Isenhart, and R.C. Schultz. 2001. Soil respiration within riparian buffers and adjacent crop fields. Plant Soil 229:117–124.
Unger, P.W., G.W. Langdale, and R.I. Papendick. 1988. Role of crop residues in improving water conservation and use. p. 69–100. *In* W.L. Hargrove (ed.) Cropping strategies for efficient use of water and nitrogen. ASA Spec. Publ. 51. ASA, CSSA, and SSSA, Madison, WI.
Unger, P.W., and T.M. McCalla. 1980. Conservation tillage systems. Adv. Agron. 33:1–58.
VandenBygaart, A.J., E.G. Gregorich, and D.A. Angers. 2003. Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. Can. J. Soil Sci. 83:363–380.
Viglizzo, E.F., F. Lértora, A.J. Pordomingo, J.N. Bernardos, Z.E. Roberto, and H. Del Valle. 2001. Ecological lessons and applications from one century of low external-input farming in the Pampas of Argentina. Agric. Ecosyst. Environ. 83:65–81.
Viglizzo, E.F., Z.E. Roberto, M.C. Filippin, and A.J. Pordomingo. 1995. Climate variability and agroecological change in the central Pampas of Argentina. Agric. Ecosyst. Environ. 55:7–16.
Wagal, R., K.R. Brye, S.T. Gower, J.N. Norman, and L.G. Bundy. 1998. Land use and environmental factors influencing soil surface CO_2 flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. Soil Biol. Biochem. 30:1501–1509.
Wander, M.M., M.G. Bidart, and S. Aref. 1998. Tillage impacts on depth distribution of total and particulate organic matter in three Illinois soils. Soil Sci. Soc. Am. J. 62:1704–1711.
Webster, R. 2000. Is soil variation random? Geoderma 97:149–163.
West, T.O., and W.M. Post. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: A global data analysis. Soil Sci. Soc. Am. J. 66:1930–1946.