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Carbon footprint of spring wheat in response to fallow frequency and soil carbon changes over 25 years on the semiarid Canadian prairie

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ABSTRACT

Growing interest in environmental quality has provided a strong incentive to examine how farming practices affect agricultural products' carbon footprints (CF), an environmental quality indicator. This study determined (i) the CF of spring wheat (*Triticum aestivum* L.) grown in different cropping systems over 25 years, and (ii) the effect of soil organic carbon (SOC) changes over years on wheat CF. Wheat was grown in four cropping systems: (a) fallow-wheat (FW), (b) fallow-wheat-wheat (FWW), (c) fallow-wheat-wheat-wheat-wheat-wheat (FWWWWW), and (d) continuous wheat (ContW), in replicated field plots in Saskatchewan, Canada. Wheat CF was calculated at a system level with measured variables coupled with modeling approaches. Over the 25-year period, the soil under the ContW system gained organic C of 1340 kg CO_2 eq ha^{-1} annually, or 38%, 55%, and 127% more than those gained in the FWWWWW, FWW, and FW systems, respectively. The SOC gain more than offset the greenhouse gas (GHG) emissions occurred during wheat production, leading to negative emission values at −742 kg CO_2 eq ha^{-1} annually for ContW, and −459, −404, and −191 kg CO_2 eq ha^{-1} for FWWWWW, FWW, and FW systems, respectively. Wheat in the ContW system produced the highest grain yield and gained highest SOC over the years, leading to the smallest (more negative) CF value at −0.441 kg CO_2 eq kg^{-1} of grain, significantly lower than the CF values from the three other systems (−0.102 to −0.116 kg CO_2 eq kg^{-1} of grain). Without considering the SOC gain in the calculation, wheat CF averaged 0.343 kg CO_2 eq kg^{-1} of grain and which did not differ among cropping systems. Wheat is the largest agricultural commodity in Saskatchewan, and the way the crop is produced has significant impacts on environmental quality, reflected by its carbon footprint. Cropping systems with decreased fallow frequency was shown to significantly enhance soil carbon gains over the years, increase annualized crop yields, and effectively lower the carbon footprint of this important commodity.

1. Introduction

Increasing emissions of anthropogenic greenhouse gases (GHG) are causing significant changes to the global climate (Ruddiman, 2003; IPCC, 2006). Agricultural activities and related farming practices contribute a large proportion of the total national GHG emissions of many countries (Janzen et al., 2006). For example, in Canada, agriculture emitted approximately 62 M tons of CO_2 equivalents (CO_2 eq) in 2008, amounting to about 8% of Canada's total emissions (Environment Canada, 2010). Nearly two-thirds of agricultural emissions occur as N_2O, which has 298 times the global warming potential of CO_2 (IPCC, 2006). Major contributors to the emissions in agriculture are from inputs of inorganic fertilizers, manures, and plant litter, and those from various biological and soil processes. Furthermore, agriculture is involved in the processing of crops into food products, and marketing of food products to consumers; each of these steps generates GHGs (Dyer et al., 2010). One of the key strategies to reducing GHG emissions in the production of field crops is to adopt improved farming practices (Gan et al., 2011a,b).

In arid and semiarid areas, water is the main factor limiting crop productivity (Campbell et al., 1997, 2007b); this is especially true in areas where potential evapotranspiration (PE) exceeds precipitation (Pr) during the growing season (Bullock et al., 2010). In the North American Great Plains region, producers have traditionally used the practice of tilled summerfallow as a means to

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conserve extra water in the soil for the subsequent crop, control weeds and increase soil available nutrients (Campbell et al., 1990). However, in recent years, the area devoted to summerfallow has been steadily decreasing (Campbell et al., 2002) with the adoption of minimum and zero tillage (Larney and Lindwall, 1994) together with increased crop diversification with oilseeds (such as *Brassica napus* canola, *Brassica juncea* mustard) and pulse crops (such as *Pisum sativum* dry pea and *Lens culinaris* lentil) (Gan et al., 2010, 2011b). When compared with continuous cropping systems, long-term rotations with frequent tilled summerfallow have been shown to reduce soil organic matter quantity (Campbell et al., 2005) and quality (McGill et al., 1986; Janzen et al., 1997). Also, frequent summerfallowing negatively impacts soil biochemical characteristics such as soil microbial biomass and potential C and N mineralization in the soil (Franzluebbers and Arshad, 1996; Campbell et al., 1997).

There is a growing interest in reducing the carbon footprint of agricultural products, i.e., the total GHG emission associated with the amount of grain produced (Williams and Wikstrom, 2011). Farm products may in the near future be valued based on their carbon footprint, with mandatory "eco-labeling" of their carbon footprint. Policy-makers, the general public, and producers urgently wish to know how farming practices could be improved in order to produce high-quality and affordable food in sufficient quantity while ensuring the product has a low carbon footprint. There is a huge knowledge gap in regard to whether or not the practice of frequent summerfallowing in farming systems affects the carbon footprint of the grains produced. The objectives of this study were to determine (i) the carbon footprint of spring wheat grown in cropping systems with different frequencies of summerfallowing over 25 years, and (ii) the effect of SOC changes over years on the carbon footprint of spring wheat produced on the semiarid Canadian prairie.

2. Materials and methods

2.1. Experimental design and crop management

Details of the experimental design and plot management of this field trial have been reported previously (Campbell et al., 2005); therefore, only a review pertinent to this paper is presented. The experiment was initiated in 1966 at the Agriculture and Agri-Food Canada Research Centre at Swift Current (50°17′N, 107°48′W, elevation 883 m) on an Aridic Haploboroll soil (Orthic Brown Chernozem, Ayers et al., 1985). The experiment consisted of 12 crop rotation treatments, established on 81 plots in a randomized complete-block design with three replicates. In this paper, we discuss four of the rotation systems: (1) summerfallow-wheat (FW), (2) fallow-wheat-wheat (FWW), (3) fallow-wheat-wheat-wheat-wheat-wheat (FWWWWW), and (4) continuous wheat (ContW). The fallow frequency of these systems was taken as 50, 33, 17, and 0%, respectively. All phases of each system were present every year and each rotation was cycled on its assigned plots. Each plot is 10.5 m by 40 m.

Fertilizer N and P were applied to wheat based on the soil NO_3–N (0–0.6-m depth) and soil P (0–0.15-m depth) levels of individual plots measured the previous fall (Campbell et al., 2004; Selles et al., 2011). From 1967 to 1990 we used N rates recommended by the soil-testing laboratory of the University of Saskatchewan with N applied to bring total mineral N (soil N + fertilizer N) to 65 kg N ha^{-1}. Starting in 1991, rates of N fertilization were increased to meet targets of 90 kg N ha^{-1} of available N in the soil for wheat grown on summer fallow, and 73 kg N ha^{-1} for wheat grown on stubble, as per new Saskatchewan Soil Testing Laboratory recommendation guidelines. Fertilizer N (primarily ammonium nitrate) was applied by broadcasting in spring prior to seedbed preparation (Selles et al., 2011). Phosphorus (P) fertilizer (monoammonium phosphate) was applied with the seed to all cropped treatments at the rate of 9–10 kg P ha^{-1} $year^{-1}$. The crop received in-crop weed control as required using recommended herbicides at label rates. In the fall, after harvest, 2,4-D was applied to all plots to control winter annual weeds. Full-sized farm equipment was used for all cultural and tillage operations. Plots being planted generally received one preseeding tillage operation with a heavy-duty sweep cultivator and mounted harrow to prepare the seedbed, whereas plots being fallowed received two to five tillage operations with a heavy-duty cultivator and/or rod weeder to control weeds (Campbell et al., 2005). All the other cropping practices such as planting and harvesting were the same as those adopted locally (Campbell et al., 2007b).

2.2. Data collection

Grain yield was determined by cutting a swath 5 m wide and 40 m long through the middle of the cropped plots and after air drying for several days, the grain was harvested using a conventional combine equipped with an automated weighing system. An area of 2.32 m^2 was also hand harvested in each plot to determine straw/grain ratios and harvest index. Although plots were established in 1966, the complete measurement of crop straw was started in 1985 and up to the present. Therefore, in this study we only used the data from 1985 to 2009, and crop productivity and treatment effect prior to 1985 were detailed previously by Campbell et al. (2007b). The N concentrations in grain and straw were measured using standard micro-Kjeldahl method. Root dry weight was estimated using the model developed by Gan et al. (2009) where root biomass was a proportion of straw biomass varying with growing conditions.

Soil carbon was measured in 1985, 1990, 1993, 1996, 2003, and 2009. At each measurement, two soil samples were taken at random within the central part of each plot. The soil in the depths of 0–15 and 15–30 cm was sampled with a 5-cm diameter probe (Campbell et al., 2007a). The soils of the two cores per depth in each plot were bulked, air dried and sieved (<2 mm). Representative subsamples of the <2 mm soil were ground with a roller mill (<153 μm) and a 20-mg subsample analyzed for organic C with an automated combustion technique (Carlo ErbaTM, Milan, Italy) at 1000 °C. Organic C values were obtained after pre-treatment of the soil with phosphoric acid to remove inorganic C, and the SOC concentrations converted to weight per volume basis using measured bulk densities. The annualized soil C gains or losses for each rotation system were calculated using the differences of soil C values measured in 2009 and those in 1985 and divided by 25 (except FWWWWW rotation system which was modified into a more diversified system in 2005 when an oilseed and legume crop were included).

2.3. Calculations of GHG emissions and carbon footprint

Wheat carbon footprint depends on grain yield and total GHG emissions associated with the crop production, including CO_2 emissions from energy use and N_2O emissions from non-energy sources. Using site-specific data coupled with empirical modeling, we estimated GHG emissions derived from (1) crop residue decomposition, (2) on-farm application of inorganic fertilizers, (3) manufacture, storage, and transportation of inorganic N and P fertilizers to the farm gate, (4) manufacture of herbicides and pesticides and their on-farm application to the crops, (5) various farming operations such as spraying pesticides, planting and harvesting the crop, and tillage operations, (6) soil C gains or losses in various cropping systems during the 25-year period, and (7) emissions of N_2O

from summerfallow areas. The system boundary was from cradle-to-farm gate (Gan et al., 2012).

The amount of direct and indirect N_2O emissions is related to the quantity of N applied to the crop and is affected by environmental conditions (Gregorich et al., 2005). Following a well-recognized model of Rochette et al. (2008), which measures N_2O fluxes from Canadian farmlands, we estimated N_2O emission factors based on the ratio of growing season precipitation (Pr) to PE. Growing season of 1 May to 30 October was used in this case, because crop residue decomposition and soil-related emissions continue until the soil is frozen (Rochette et al., 2008):

$$\mathrm{EF} = \frac{0.022Pr}{\mathrm{PE}} - 0.0048, \tag{1}$$

where EF is the emission factor with a unit of kg N_2O–N kg^{-1} of N, and Pr/PE is the ratio of precipitation to PE during the growing season (Table 1).

Soil mineral N particularly in the form of nitrate in the rooting zone is subject to leaching (Campbell et al., 2004), and this N can be leached out of the rooting zone and/or undergone further transformations to be emitted as N_2O. We estimated the fraction of N subject to leaching ($\mathrm{FRAC_{LEACH}}$) as:

$$\mathrm{FRAC_{LEACH}} = \frac{0.3247Pr}{\mathrm{PE}} - 0.0247. \tag{2}$$

Using the method developed by the Intergovernmental Panel on Climate Change (IPCC, 2006), emissions of N_2O from inorganic N applications were estimated as:

$$\mathrm{CO_2\ eq_{SNF}} = Q_{\mathrm{SNF}} \times \{(\mathrm{FRAC_{GASM}} \times \mathrm{EF_{VD}}) + \mathrm{EF} + (\mathrm{FRAC_{LEACH}} \times \mathrm{EF_{LEACH}})\} \times \frac{44}{28} \times 298 \tag{3}$$

where $\mathrm{CO_2\ eq_{SNF}}$ is the total emissions from the inorganic N fertilizer application (kg CO_2 eq ha^{-1}), Q_{SNF} is the quantity of synthetic N fertilizer applied (kg N ha^{-1}), $\mathrm{FRAC_{GASM}}$ is the fraction of inorganic N fertilizer that volatilized as NH_3– and NO_x–N ($\mathrm{FRAC_{GASM}}$ = 0.1 kg N kg^{-1} N), $\mathrm{EF_{VD}}$ is the N_2O emission factor for volatilized NH_3– and NO_x–N ($\mathrm{EF_{VD}}$ = 0.01 kg N kg^{-1} N), $\mathrm{EF_{LEACH}}$ is the N_2O emission factor for nitrate leaching ($\mathrm{EF_{LEACH}}$ = 0.0075 kg N kg^{-1} N), 44/28 is the conversion coefficient from N_2O–N to N_2O, and 298 is the global warming potential of N_2O for the 100-year period (IPCC, 2006).

The N contained in the crop residue (after harvest) provided an additional source of N for nitrification and denitrification, leading to N_2O emissions. The quantity of crop residue N (Q_{CRD}) was obtained using the aboveground and belowground crop residue biomass values multiplied by their respective N concentrations. Using the formula of Gan et al. (2011a,b), emissions from crop residue decomposition were calculated as:

$$\mathrm{CO_2\ eq_{CRD}} = Q_{\mathrm{CRD}} \times \{\mathrm{EF} + (\mathrm{FRAC_{LEACH}} \times \mathrm{EF_{LEACH}})\} \times \frac{44}{28} \times 298 \tag{4}$$

During the fallow period, no fertilizer is applied; however, several other factors may stimulate N_2O emissions, such as higher soil water content, temperature and available carbon and soil N. Field studies have shown that N_2O emissions during the fallow period is proportional to the emission from continuously cropped fields (Rochette et al., 2008). In order to account for these emissions not captured by the default IPCC input-driven approach, we estimated the effect of summerfallow on N_2O emissions as:

$$\mathrm{CO_2\ eq_{Fallow}} = \mathrm{CO_2\ eq_{SNF}} + \mathrm{CO_2\ eq_{CRD}} \tag{5}$$

where $\mathrm{CO_2\ eq_{Fallow}}$ is the emissions of N_2O resulting from summerfallow (kg CO_2 eq ha^{-1}); $\mathrm{CO_2\ eq_{SNF}}$ is the emissions of N_2O resulting from inorganic nitrogen application for ContW (kg CO_2 eq ha^{-1}), and $\mathrm{CO_2\ eq_{CRD}}$ is the emissions of N_2O from crop residue decomposition for ContW (kg CO_2 eq ha^{-1}).

Based on available data from the studies conducted in North America, we estimated emissions from the manufacture, transportation, storage, and delivery of fertilizers to the farm gate using an emission factor of 4.8 kg CO_2 eq kg^{-1} of N and 0.73 kg CO_2 eq kg^{-1} of P_2O_5, multiplied by the amount of N and P fertilizers applied on a per hectare basis.

Herbicides and fungicides were used in the production of wheat crops. Although emission factors for each individual pesticides used in wheat production are not readily available, we assumed that the emission during the process of manufacture, transportation, storage, and field application were similar among pesticides within a similar category. Thus, an average emission factor of 23.1 kg CO_2 eq kg^{-1} of a.i. (active ingredient) was used for herbicides and 14.3 kg CO_2 eq kg^{-1} of a.i. for fungicides. The emissions associated with various farming operations such as tillage, planting, spraying, and harvesting were estimated using factors of 14, 14, 5, and 37 kg CO_2 eq ha^{-1}, respectively.

Soil carbon is an important factor in influencing the carbon footprint of the cropping systems, as it changed substantially over time or varied significantly among cropping systems. In our study, the annualized soil C gain or loss for each of the rotation systems, except for FWWWWW, was determined as follow:

$$\Delta C = \frac{\mathrm{SOC_{2009}} - \mathrm{SOC_{1985}}}{25} \times \frac{44}{12} \tag{6}$$

where ΔC is the annual change in SOC since 1985 (kg CO_{2eq} ha^{-1} year^{-1}); $\mathrm{SOC_{2009}}$ and $\mathrm{SOC_{1985}}$ are the amount of SOC in the 0–15-cm soil in 2009 and 1985, respectively; 25 is the duration of the study period, and 44/12 is the coefficient converting C into CO_2. The FWWWWW treatment was converted to a diversified cereal–oilseed–cereal–legume system in 2005, and thus the annual change in SOC for FWWWWW was calculated using the following equation:

$$\Delta C = \frac{\mathrm{SOC_{2003}} - \mathrm{SOC_{1985}}}{19} \times \frac{44}{12} \tag{7}$$

where ΔC is the amount of change in SOC (kg C ha^{-1} year^{-1}); $\mathrm{SOC_{2003}}$ is the amount of SOC in the 0–15-cm soil in 2003, and 19 is the duration of the study period for this rotation.

To facilitate the discussion we grouped emissions into five categories: (i) emissions related to inputs of N including manufacture, storage and transportation, as well as field application of N fertilizer; (ii) emission from crop residue decomposition; (iii) emissions related to inputs from non-N sources such as manufacture, storage and transport of phosphorus, herbicides, and fungicides; (iv) emissions from various farming operations such as seeding, tilling, harvesting, spraying; and (v) emission during the summerfallow period. Carbon footprints for each rotation system were calculated based on an entire rotation cycle, including all phases of the rotation, with and without consideration of the changes in SOC during the 25-year study period.

$$\mathrm{CF} = \frac{\sum_i \sum_j (\text{Emission Category}_{i,j} + \Delta C)}{\sum_i \text{Grain Yield}_i} \tag{8}$$

where CF is carbon footprint of a rotation (kg CO_2 eq kg^{-1} of grain), Emission $\text{Category}_{i,j}$ is emissions from a jth emission category in an ith rotation phase (kg CO_2 eq ha^{-1}), GrainYield_i is the grain yield of wheat from an ith phase of a rotation (kg ha^{-1}), and ΔC is the amount of change in SOC (kg C ha^{-1} year^{-1}) when this factor was included in the CF calculation.

Table 1
Precipitation (Pr), evapotranspiration (PE), N_2O emission factors (EF), and N-leaching factor ($FRAC_{LEACH}$) under dry-, normal-, and wet-conditions over the period of 25 years in Swift Current, Saskatchewan, Canada.

Year	Precipitation (mm)	Mean T (°C)	PE	Pr/PE	EF	$FRAC_{LEACH}$
Dry						
2001	147[a]	14.5	664	0.221	0.0001	0.047
2007	166	14.4	634	0.262	0.0010	0.060
1985	177	12.6	589	0.299	0.0018	0.072
1987	187	14.1	636	0.293	0.0017	0.070
1990	204	13.7	624	0.327	0.0024	0.081
2003	209	14.7	641	0.326	0.0024	0.081
1988	215	15.3	670	0.307	0.0020	0.075
Mean	186	14.2	641	0.291	0.0016	0.069
Std error	9	0.3	13	0.014	0.0003	0.004
Normal						
2009	226	12.9	598	0.378	0.0035	0.098
1994	240	14.2	619	0.388	0.0037	0.101
1997	247	14.0	623	0.397	0.0039	0.104
1998	268	14.9	653	0.411	0.0042	0.108
2005	269	13.1	562	0.478	0.0057	0.130
1999	276	12.8	546	0.506	0.0063	0.139
2006	277	14.0	621	0.445	0.0050	0.119
1992	278	12.2	554	0.501	0.0062	0.138
1996	318	12.4	545	0.584	0.0081	0.165
2000	325	13.5	588	0.552	0.0074	0.154
1986	329	13.2	569	0.578	0.0079	0.163
1989	332	13.5	569	0.583	0.0080	0.164
2004	332	11.9	518	0.641	0.0093	0.183
Mean	286	13.3	582	0.495	0.0061	0.136
Std error	10	0.2	11	0.024	0.0005	0.007
Wet						
2008	351	13.3	584	0.601	0.0084	0.170
1991	364	13.4	589	0.618	0.0088	0.176
1995	380	12.8	551	0.689	0.0104	0.199
1993	401	12.2	544	0.738	0.0114	0.215
2002	419	12.0	531	0.790	0.0126	0.231
Mean	383	12.7	559	0.687	0.0103	0.198
Std error	12	0.3	11	0.035	0.0008	0.011

[a] Weather data were for the entire growing season – 1 May–31 October.

2.4. Data management and statistical analysis

In the study, the length of the rotation cycles varied among rotation systems. For example, the FWW system had a 3-year cycle (fallow in yr-1, wheat in yr-2 and yr-3), while the FWWWWW system had a 6-year cycle. To simplify the interpretation of the results, we determined all crop yield, SOC, and GHG emission factors for the entire period of the rotation cycle and, then, annualized values were compared on a per-rotation basis. For example, the annualized grain yield for the FWW rotation system was the sum of the 2 years of wheat crops (i.e., wheat after summerfallow plus wheat after wheat) divided by three (including zero yield in the fallow phase). All phases of the rotation occurred each year, therefore, all crop-related variables were averaged across cropped phases of the rotations; this allowed the concentration of the analysis on a per-rotation system basis.

Furthermore, during the course of the 25 study years, weather conditions varied substantially. To avoid possible confounding effect of highly variable annual weather conditions with the effect due to the systematic changes in rotation over years, we categorized the 25 study years into three categories of environmental conditions – dry, normal, and wet (Table 1); this was based on growing season precipitation (Pr), evapotranspiration (PE), and the ratio of Pr to PE which affects carbon footprint estimates significantly (Eqs. (1) and (2)). In the 7 dry years, the Pr averaged 186 mm, PE 641 mm, Pr/PE = 0.291, and mean air temperature 14.2 °C. In the 5 wet years, the Pr averaged 383 mm, PE 560 mm, Pr/PE = 0.036, and mean temperature 12.7 °C. The remaining 13 years had weather conditions near the long-term normal, with Pr = 286 mm, PE = 582 mm, Pr/PE = 0.496, and mean temperature 13.3 °C. Preliminary analysis revealed significant treatment effects between dry-, normal-, and wet-conditions, whereas the effect of cropping systems was insignificant within each environmental category. Therefore, the treatment effects were determined mainly for each category rather than each individual year. Data from the original plots were analyzed as a randomized complete block design with years as main plot and the rotation systems as subplots, using the restricted maximum likelihood (REML) method (Littell et al., 1996). In all analyses, significant effects were declared at $P < 0.05$.

3. Results

3.1. Wheat productivity and N inputs

Over the 25-year period, the yield of wheat varied largely from year to year; severe drought occurred in 1988 which caused yields to be less than 250 kg ha^{-1}, whereas under more favorable conditions such as in 2004 wheat yields averaged in excess of 2400 kg ha^{-1}. Overall, wheat yields were directly proportional to growing-season (1 May–31 August) precipitation with each mm of precipitation increasing grain yield by an average of 5.26 kg ha^{-1} (Fig. 1A). There were significant yield increases during the period from 1985 to 1991, and thereafter wheat yields have been in a narrow range with a slight decline in the last few years (Fig. 1B). The greatest yield increase from 1985 to 1991 was largely due to the change of fertilization program where the target of soil available N was increased from 65 kg N ha^{-1} to 90 kg N ha^{-1}. The magnitude of yield change during the 25-year period was nearly parallel to the change of fertilization. Based on the annualized grain yield for each

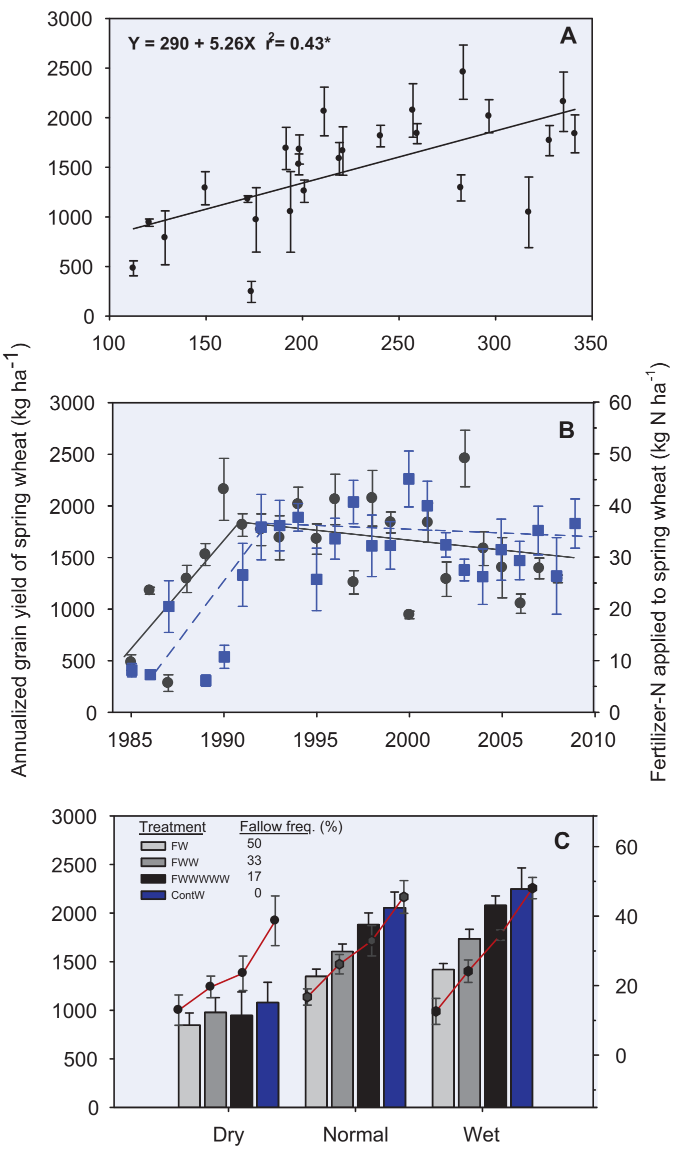

Fig. 1. Annualized grain yield of spring wheat (A) in relation to growing-season (1 May–31 August) precipitation (mm), (B) varying (solid line) from 1985 to 2009 and nearly paralleling to fertilizer-N (dashed line) applied to the wheat crop, and (C) affected by fallow-frequency treatments under dry-, normal-, and wet-conditions, in Swift Current, Saskatchewan (bars are standard errors, and the three line graphs show the amounts of fertilizer-N applied to the crop).

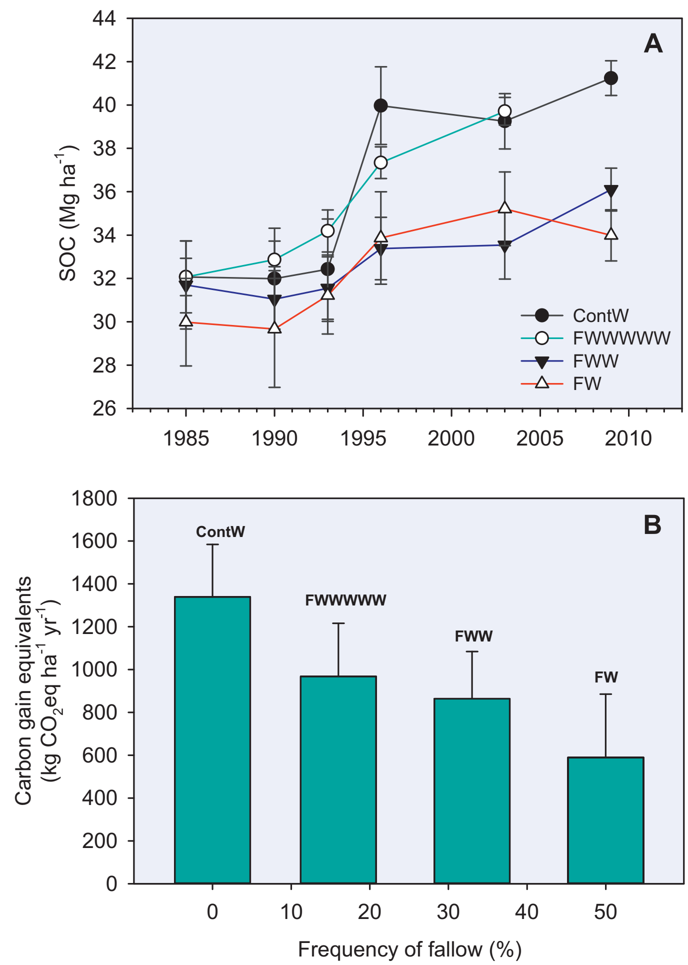

Fig. 2. Effects of fallow frequency in cropping systems on (A) soil organic carbon and (B) carbon gain equivalents for spring wheat at Swift Current, Saskatchewan, over the 25-year period.

Data from 1985 to 2003 were adapted from Campbell et al. (2007a,b).

rotation system (i.e., zero yield in the fallow phase was accounted in the yield calculation), fallow frequency interacted with water availability in affecting annualized grain yield (Fig. 1C). In the dry years, wheat in the FW system had lowest annualized grain yield whereas wheat in the three other systems did not differ in yield, averaging 962 kg ha^{-1}. In normal years, wheat yield differed significantly among the four rotation systems; with the ContW system producing 2055 kg ha^{-1} of grain annually, or 9%, 28%, and 52% more than was produced by the FWWWWW, FWW, and FW systems, respectively. Similarly, in the wet years, the ContW system produced 8%, 30%, and 59% more grain than wheat in the three other respective systems. Harvest index did not differ greatly among cropping systems within a year, but it was typically higher in dry years, averaging 0.395, compared with 0.372 in normal years and 0.386 in wet years.

On the basis of an entire cropping system, annualized N fertilizer rates applied to the wheat crops varied among cropping systems and among water availability categories (Fig. 1C). In dry years, the ContW system received an annualized N fertilizer rate of 39 kg N ha^{-1}, or 64% more than that in the FWWWWW system, twice that in the FWW systems, and nearly triple that in the FW system. The FW system received the lowest N fertilization since no fertilizer was applied in the fallow phase. Similarly, in normal and wet years, the ContW system received an annualized N fertilizer rate of 47 kg N ha^{-1}, which was 38%, 86%, and 227% more than the N applied to the FWWWWW, FWW, and FW systems, respectively.

3.2. Soil organic carbon gains/losses

Previous analyses have showed that the amount of SOC changes with time in both the 0–15-cm and 15–30-cm depths, but was only affected significantly by rotation system in the 0–15-cm depth (Campbell et al., 2007a). Therefore, in the present analysis, only the amount of SOC in the 0–15-cm depth was used to evaluate the cropping system effects on the respective carbon footprints. The year x cropping systems interaction on SOC was significant (ANOVA not presented), and the interaction was mainly due to the rate of the change in SOC that varied among rotation systems (Fig. 2A). Soil C changed little before 1993, between 1993 and 1996 the amount of SOC increased significantly for most of the cropping systems, and since 1996, SOC increased little or none. The quantities of SOC in the ContW and FWWWWW systems were consistently greater than those in the FWW and FW systems during the entire study period, whereas the FWW system had a similar amount of SOC as the FW system, except in 2009 when the FW system had the lowest SOC among the four rotation systems.

To estimate total GHG emission from various cropping systems, we quantified SOC changes and the rate of C gains or losses in soils during the 25-year period. Using the measured and previously-established bulk densities, we converted the C concentrations to a mass per fixed depth basis, a method similar to that used by Campbell et al. (2007a). The amount of C gain or loss was then converted into CO_2 equivalents (Fig. 2B), thus permitting us to include or exclude C sequestration in the GHG emission calculation. It was not surprising that the soil under the ContW system gained SOC at 1340 kg CO_2 eq ha^{-1} annually, the highest among the four rotation systems. This was double that gained in the FWW or FW systems (averaging 726 CO_2 eq ha^{-1}). The differences in SOC gains between FWW and FW were rarely significant statistically due to large variation in the data. Comparing among cropping systems, there was a general trend that decreased fallow frequencies in the rotation increased the rate of SOC gains over time (Fig. 2B).

3.3. Emission contributors and environmental effects

The N_2O emission factors differed largely among environmental conditions. On average, the N_2O emission factor in dry years was at 0.0016 (±0.0003); it was significantly higher in normal years (0.0061 ± 0.0005) and the highest in wet years (0.0103 ± 0.0008) (Table 1). This difference in N_2O emission factor was largely attributable to the differences in Pr/PE ratio, which increased from 0.29 in dry years to 0.50 in normal years and 0.69 in wet years. Similarly, the higher Pr/PE ratio resulted in a greater emission factor for potential leaching of N into underground, which was increased from 0.0698 in dry years to 0.1363 in normal years and to 0.1986 in wet years (Table 1).

Many factors contributed to the GHG emissions in the production of wheat crops. Among them was the environment, which had the largest effect (Table 2). Averaged across the four cropping systems, the annualized GHG emission in dry years was 305 kg CO_2 eq ha^{-1}, less than half of the emission in wet years and about 58% of the total emission that occurred in normal years.

Nitrogen, phosphorus, and pesticides are the main inputs in the production of wheat crops. Averaged over cropping systems, the estimated indirect emissions from the manufacture, transportation, storage, and delivery of inorganic N fertilizer accounted for 26.7% of the total emissions, and this together with the direct emissions from the N fertilizer itself accounted for another 18.5% of the total emissions (Table 2). In dry years, the direct and indirect emissions associated with N fertilizer use represented 43% of the total annual emissions, compared to 45% in normal years, and 48% in wet years. The emissions from the use of N fertilizer averaged 22.8 times the emission associated with P fertilizer use, 11.2 times the emission from the pesticides use, and 2.5 times the emission from various cultural and tillage operations.

The decomposition of wheat crop residues contributed direct and indirect emissions, averaging 53 kg CO_2 eq ha^{-1}, or 9.3% of the total emission (Table 2). This portion varied among the environmental conditions, largely due to the variation in Pr/PE ratio (Table 1) which influenced the amount of wheat straw and root biomass produced. Wheat grown in dry years produced an average straw biomass of 1453 kg ha^{-1}, significantly less than that produced in normal (2880 kg ha^{-1}) and wet (3024 kg ha^{-1}) years, resulting in the corresponding emission from crop residue decomposition at 3.4%, 12.3%, and 12.2% of the total emission, respectively.

Emissions associated with pesticide use were much lower than those from N fertilizer use (Table 2), while emissions from cultural and tillage operations were highly significant representing 35% of the total emissions in dry years, 22% in normal year, and 18% in wet years. In addition, emissions during the fallow period varied greatly between dry, normal, and wet years (Table 2), reflecting the increased tillage needed to control weeds during wetter growing seasons. Averaged across cropping systems, emission associated with fallowing activities accounted for 6.0% of the total in dry years, 13.2% in normal years, and 17.3% in wet years.

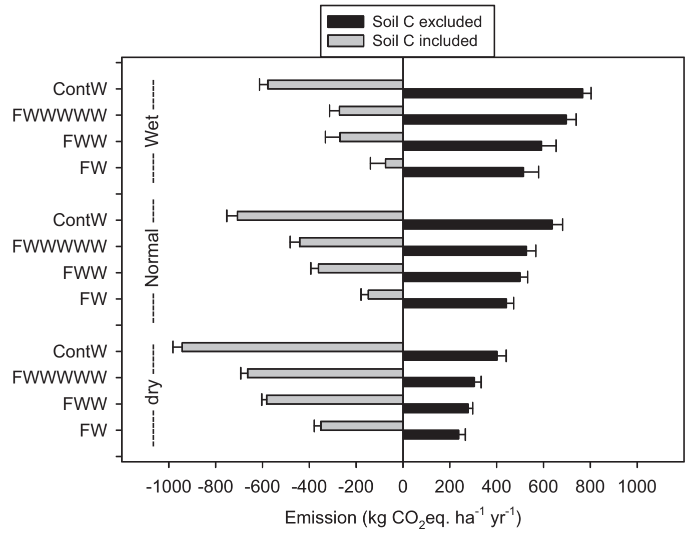

Fig. 3. Total GHG emission of spring wheat as a function of fallow-frequency and environmental condition, with and without soil organic carbon gain/loss included in the analysis, at Swift Current, Saskatchewan (bars are standard errors).

3.4. Effect of cropping systems on GHG emission and carbon footprint

Fallow frequency significantly impacted total GHG emissions, with the magnitude of the effect influenced by the prevailing environmental conditions (Fig. 3). The inverse relationship between fallow frequency and GHG emissions largely reflects that cropping the land more intensively requires a greater input of N fertilizer, the main contributor to GHG emissions. When SOC change was excluded from the calculations, the four cropping systems emitted an average of 642 kg CO_2 eq ha^{-1} per year in wet years, 22% more than was emitted in normal years, and 110% more than was emitted in dry years. However, when SOC change was included in the calculations, the emission values were negative, reflecting that the cropping systems were acting as net sinks for carbon storage, with the ContW system providing the greatest and FW the lowest carbon sequestration benefit. Furthermore, the effect of environmental conditions was reversed with emissions averaging −298 kg CO_2 eq ha^{-1} in wet years, −414 kg CO_2 eq ha^{-1} in normal years, and −635 kg CO_2 eq ha^{-1} in dry years.

By definition, the carbon footprint for the spring wheat production systems was directly related to grain yield and inversely related to total GHG emissions. In the present study, the carbon footprint values were also significantly influenced by the level of soil C sequestration. Without considering soil C sequestration, the carbon footprint was generally similar among cropping systems, averaging 0.343 kg CO_2 eq kg^{-1} of grain (Fig. 4). In all cases, increased grain yield was at the expense of the increased GHG emissions. As a result, the carbon footprint value was relatively constant across the four cropping systems and among environmental categories.

When soil C changes were included in the calculations, the carbon footprint values became negative and differed significantly among cropping systems and among environments (Fig. 4). Overall, the ContW system had the lowest carbon footprint, with little difference among the remaining three cropping systems. In dry

Table 2
Total emission and percent contribution by individual emission components for spring wheat grown in various fallow-frequency systems under dry- (Pr/PE = 0.291), normal- (Pr/PE = 0.496), and wet- (Pr/PE = 0.688) conditions, at Swift Current, Saskatchewan.

Environmental conditions/cropping system	Total emission	Emission relating to N				Emission relating to input other than N				Various farming oper.	Fallow effect[a]
	$kg\ CO_2\ eq\ ha^{-1}$	Fert.N prod. %	Fert. N appl.	Res. soil N	Subtotal	Fert. P prod. %	Herbi. prod.	Fung. prod.	Subtotal	%	$kg\ CO_2\ eq\ ha^{-1}$
Dry											
Fallow-W	238	22.1	5.9	3.8	31.7	3.5	9.7	0.4	13.6	41.4	13.3
Fallow-W-W	278	32.6	9.4	3.5	45.5	3.8	7.2	0.5	11.5	35.7	7.3
Fallow-W-W-W-W-W	305	35.0	10.5	3.3	41.8	4.4	9.4	0.4	12.2	33.8	3.3
Continuous-W	401	43.1	12.5	2.8	58.4	4.1	8.2	0.5	12.8	28.8	0.0
Mean	305	33.2	9.6	3.4	44.4	3.9	8.6	0.4	12.5	34.9	6.0
P-value	<0.01	<0.01	<0.01	0.27	0.02	0.80	0.85	0.75	0.80	<0.01	<0.01
LSD (0.05)	40	7	3	NS	15	NS	NS	NS	NS	5	3
Normal											
Fallow-W	441	17.6	12.4	9.9	40.0	1.9	4.7	0.3	6.9	24.6	28.5
Fallow-W-W	499	24.6	17.8	12.1	54.5	2.2	3.8	0.4	6.4	22.3	16.8
Fallow-W-W-W-W-W	526	28.1	22.2	12.1	52.8	2.7	5.4	0.4	7.2	21.4	7.7
Continuous-W	637	33.2	24.4	15.2	72.8	2.5	5.0	0.5	8.0	19.2	0.0
Mean	526	25.9	19.2	12.3	55.0	2.3	4.7	0.4	7.2	21.9	13.2
P-value	<0.01	<0.01	<0.01	0.01	<0.01	1	1	0.6	0.35	<0.01	<0.01
LSD (0.05)	34	2	2	3	11	NS	NS	NS	NS	2	3
Wet											
Fallow-W	514	10.8	14.0	10.9	35.8	1.7	3.5	0.2	5.4	21.0	37.8
Fallow-W-W	591	19.3	24.4	11.3	55.0	1.9	2.7	0.3	4.9	18.4	21.7
Fallow-W-W-W-W-W	697	23.7	31.0	14.2	55.1	2.0	3.2	0.2	4.3	16.1	9.7
Continuous-W	767	30.0	37.2	12.2	79.4	2.2	3.0	0.3	5.5	15.1	0.0
Mean	642	21.0	26.7	12.2	56.3	1.9	3.1	0.2	5.0	17.7	17.3
P-value	<0.01	<0.01	<0.01	0.78	<0.01	1	1	0.7	0.55	<0.01	<0.01
LSD (0.05)	81	4.8	5.1	NS	20.1	NS	NS	NS	NS	3	3

[a] Fallow effect represents all emissions during the fallow phase of the rotation.

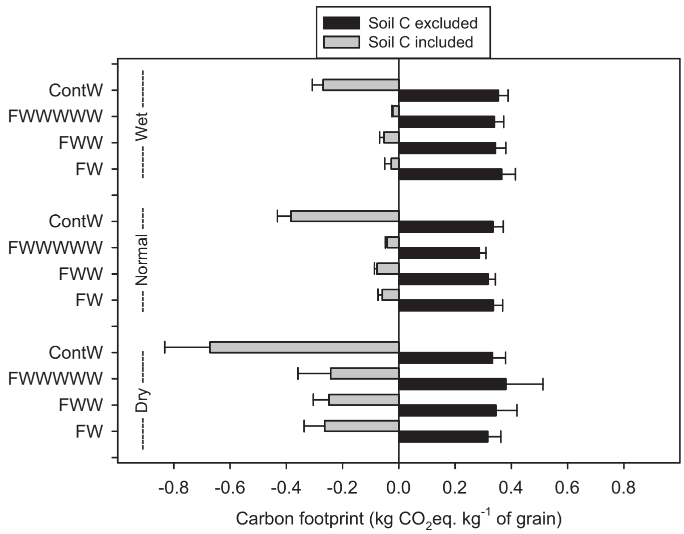

Fig. 4. Carbon footprint of spring wheat as a function of fallow-frequency and environmental condition, with and without soil organic carbon gain/loss included in the analysis, at Swift Current, Saskatchewan (bars are standard errors).

years, the carbon footprint averaged −0.357 kg CO_2 eq kg^{-1} of grain compared to −0.140 kg CO_2 eq kg^{-1} of grain in normal years and −0.093 kg CO_2 eq kg^{-1} of grain in wet years. The highest negative carbon footprint in dry years is attributable to the lowest emissions from least N fertilization and least crop residue decomposition which more than offset the low grain yields (Table 2).

4. Discussion

4.1. Fallow frequency and grain yields

The grain yield of wheat crop grown on fallow was greater than the yield of wheat grown on stubble; this was highly expected because more soil water was conserved in the fallow fields under the semiarid environment (Nielsen and Vigil, 2010). Increased soil moisture usually reduces heat stress to crop roots in the hot summer (Wang et al., 2008) and improves crop productivity (De Jong et al., 2008). However, annualized grain yield (i.e., the yield for the whole rotation system including the yield of zero in fallow phases) was significantly influenced by the frequency of wheat crop in the system. Among the four rotation systems evaluated, the ContW system had the greatest annualized grain yield. The lowered frequency of fallow in the rotation increased the annualized yield for the system. The increased grain yield of the wheat crop grown after fallow, compared with wheat after wheat, did not overcome the lost yield in the fallow phase. These findings are in agreement with those reported previously (Larney and Lindwall, 1994; Campbell et al., 2007b; De Jong et al., 2008; Nielsen and Vigil, 2010).

4.2. Soil C changes over years

The amount of SOC typically changes over years on the semiarid Canadian prairie, but cropping systems only affected the SOC in the 0–15-cm depth, with no effect on the SOC below 15-cm depth (Campbell et al., 2007a). In the present study, the SOC changed little before 1993 or after 1996 with the greatest increase in SOC occurred between 1993 and 1996 (Fig. 2). The mechanism responsible for such a great change in SOC during 1993 to 1996 was not known, but it was most likely related to (a) the higher rates of N fertilizer applied to the crops since 1991, 2 years before the large SOC increase was realized; and (b) the more favorable growing conditions leading to higher crops yields, and thus, more straw returned to the soil in the 1990s and beyond compared to the earlier years. This assumption was supported by the two nearly parallel lines of N fertilization and of grain yield presented in Fig. 1B. Beyond 1996, the amount of SOC generally did not change much; it is possible that SOC accrual due to improved crop management may have reached its plateau under the semiarid climate.

Soil carbon may decline over time with the use of intensive tillage (Campbell et al., 1996; Chivenge et al., 2007; Sainju et al., 2008), limited amounts of residues returned to the soil (Dam et al., 2004), or poor crop yield (Liang et al., 2003; Zentner et al., 2011). In the present study, tillage was carried out in all the systems evaluated, and two to five tillage operations were implemented to keep the fallow clean, but no evidence has shown that SOC is declining after reaching its accrual. In another trial nearby the present study where various intensities of tillage operation were implemented on the medium-textured silt loam soil, McConkey et al. (2003) found that tillage operation had minimal effect on carbon inputs, and thus minimal effects on SOC; the level of SOC remained constant in the FW system under no-till and conventional-tillage systems after 18 years of tillage implementation. Part of the reason is that the FW system has not changed over the past 90–100 years at the experimental site and thus soil C has been in a steady state. However, continuous cropping has provided greater C input to the soil (Campbell et al., 2007a) and has thus gradually increased soil C, especially labile soil quality attributes (Liang et al., 2003). On the semiarid Canadian prairie, tillage rarely affects soil C state (McConkey et al., 2003). After 7–11 years of treatments, tillage usually influences mineralizable C and microbial biomass in the 0–7.5 cm layer (Campbell et al., 2002), but the changes in minirealizable C and N do not necessarily increase soil available nutrients or crop yields (Liang et al., 2003). Proper fertilization is a key to improve crop net productivity and increase soil C inputs (Zentner et al., 2011), and such practices usually has little effect on CO_2 emissions in medium-textured soils (Beheydt et al., 2008; Malhi et al., 2009), but significantly increases C inputs to the soil (Zentner et al., 2011).

4.3. Roles of soil C in influencing the carbon footprint of spring wheat

In the Swift Current study, the value of carbon footprint for spring wheat was significantly influenced by soil carbon change. Including soil carbon change over the 25-year period in the calculation reversed the carbon footprint values from positive (undesirable) to negative (desirable). Overall, when soil carbon gains were excluded, the estimated carbon footprint of spring wheat was in a narrow range, from 0.325 to 0.359 kg CO_2 eq kg^{-1} of grain, and did not differ greatly among the cropping systems evaluated, nor did among the growing conditions experienced. When soil carbon change was included in the calculation, however, the estimated carbon footprint values for wheat became negative, and the magnitude of the negativity varied greatly with farming systems. The more intensified wheat cropping practices significantly increased soil carbon gains (despite not indefinitely), increased annualized grain production, and thus lowered the value of carbon footprint.

There is a growing discussion worldwide about policy-setting on the 'eco-labeling' requirements for agricultural products (Williams and Wikstrom, 2011). In some EU countries, particular attention has recently been on the carbon footprint of specific food products, as defined in the model of PAS 2050 (Carbon Trust, 2010). The PAS 2050 model has gained considerable credibility and may become an international standard upon which carbon footprint requirements are based (Finkbeiner and Berlin, 2009). The PAS model indicates that an assessment of carbon footprint must include GHG emissions arising from direct land-use change, but should not include those arising from soil carbon changes in existing agricultural systems. The model also indicates that offsetting schemes are not to be used to reduce the emissions associated with a particular product. However, the argument is that soil carbon change associated with farming practices should be considered in the calculation of carbon footprint, as some of the farming practices directly relating to soil carbon gain or loss have direct or indirect impacts on environmental quality (Ostle et al., 2009). Holos, a modeling program developed by Agriculture and Agri-Food Canada (AAFC, 2009), indicates the importance of estimating CO_2 from land use change and changes in soil carbon stocks. This whole-farm modeling software program estimates GHG emissions based on information entered by individual farms, including enteric fermentation and manure management, farming systems and energy use in crop production. Carbon storage and loss from changes in land use and management are also estimated, resulting in a whole-farm GHG estimate. More debates may be necessary whether or not soil carbon changes due to farming practices should be included in the calculation of the carbon footprints of grain products.

4.4. Main contributors to the carbon footprint of wheat

A close examination of emission contributors to carbon footprint of wheat production revealed that inorganic N fertilizers contributed the greatest percentage (45%) to total GHG emissions. Of this about 19% came from direct and indirect emissions of N_2O associated with the application of N fertilizers in the field, and the other 26% came from the manufacture, transport, storage, and delivery of N fertilizers to farm gate prior to on-farm use. Plant litters served as an important N source for nitrification and denitrification (Janzen et al., 2006), contributing directly and indirectly to N_2O emissions (Rochette et al., 2008). In our study, the decomposition of wheat straw and roots contributed an average of 10% to the total GHG emission. In comparison, the emissions due to farming operations (namely tillage, planting, spraying, harvesting etc.) contributed about 24%. However, increased GHG emissions due to increased N fertilization and other operations in wheat production did not necessarily lead to a higher carbon footprint, depending on whether or not the cropping practices resulted in greater grain yields. Previous studies have showed that the carbon footprints of agricultural products can be lowered significantly by adopting diversified cropping systems (Gan et al., 2011b), lowering fertilization to crops (Gan et al., 2011a), and using improved crop management practices (Gan et al., 2012). The present study indicate that on the semiarid Canadian prairie the key to lowering the carbon footprint of spring wheat is to increase grain yield, improve N use efficiency, and most importantly to enhance soil carbon gain through the use of cropping systems with reduced frequency of summerfallow.

5. Conclusions

Over the 25-year period (1985–2009), wheat producers on the semiarid Canadian prairie experienced record dry and wet growing conditions, with annualized grain yields ranging from 245 kg ha^{-1} (1988) to 2460 kg ha^{-1} (2004). The soils under the ContW production system gained a considerably large amount of carbon over the 25-year study period, averaging 1340 kg CO_2 eq ha^{-1} annually, which was 38% more than soil C gained in FWWWWW, 55% more than that gained in FWW, and 127% more than that gained in the FW system. Consequently, all wheat production systems evaluated in the study became net sinks for carbon sequestration, resulting in negative carbon footprint values, ranging from −0.116 kg CO_2 eq kg^{-1} of grain for wheat produced in the FW system to −0.441 kg CO_2 eq kg^{-1} of grain for wheat produced in the ContW systems. Without considering the SOC gain in the calculations, the carbon footprint of wheat averaged 0.343 kg CO_2 eq kg^{-1} of grain and which did not differ among cropping systems. This study showed that the way a wheat crop is produced has significant impacts on environmental quality, reflected by the carbon footprint of the grain produced. We conclude that more intensified cropping systems with low fallow frequencies must be adopted in order to effectively enhance soil carbon gains over years and thereby to lower the carbon footprint of wheat in semiarid climates.

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