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North American prairie wetlands are important nonforested land-based carbon storage sites

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Abstract

We evaluated the potential of prairie wetlands in North America as carbon sinks. Agricultural conversion has resulted in the average loss of 10.1 Mg ha⁻¹ of soil organic carbon on over 16 million ha of wetlands in this region. Wetland restoration has potential to sequester 378 Tg of organic carbon over a 10-year period. Wetlands can sequester over twice the organic carbon as no-till cropland on only about 17% of the total land area in the region. We estimate that wetland restoration has potential to offset 2.4% of the annual fossil CO₂ emission reported for North America in 1990.

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Keywords: Carbon sequestration; Wetlands; Prairie pothole region; Global climate change

1. Introduction

Concern over global climate change has stimulated much interest in identifying existing and potential carbon sinks. Atmospheric-based studies have provided

compelling evidence of a large terrestrial carbon sink in the Northern Hemisphere (Tans et al., 1990; Ciais et al., 1995) divided between North America and Eurasia (Schimel et al., 2001). Most of the North American sink is located below 51°N, south of the boreal forest zone (Fan et al., 1998). Land- and atmosphere-based studies have estimated that about half of the carbon sink in the United States is from nonforested areas (Pacala et al., 2001). However, land-based estimates have been unable to account for a large

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portion of the carbon sink identified by atmospheric-based studies. Though previous work has provided insight into the discrepancy between land- and atmosphere-based carbon sinks, land-based estimates need to be refined, especially for woody encroachment of western grasslands and for carbon storage in soils (Pacala et al., 2001). Woody encroachment was thought to increase carbon storage but the carbon sink previously attributed to woody encroachment of grassland has likely been overestimated because of reduced root biomass when shrubs replace grasses (Jackson et al., 2002). Thus, the location of specific terrestrial carbon sinks in North America and their capacity to sequester atmospheric carbon remains poorly defined.

The glaciated prairie pothole region (PPR) is a major nonforested landform in north-central North America (Fig. 1). The PPR is approximately 900,000 km² (Mann, 1986; Phospahala et al., 1974) and may have contained over 20 million ha of wetlands prior to European settlement (Millar, 1989; Tiner, 1984). Soils in the PPR are fertile and the area has been

extensively developed for agriculture. Consequently, >50% of the wetland area in the PPR of the United States (Tiner, 1984) and 71% in Canada (Environment Canada, 1986) have been drained for agricultural development. Although ground-based sink studies have found cropland and grassland soils to be important for carbon storage (Lal et al., 1999), carbon storage in wetlands has not been evaluated. Wetlands make up 23% of the land area in the PPR, and they are important components of the global ecosystem, performing many important functions, including carbon cycling (Mitsch and Gosselink, 2000). Extensive conversion of PPR wetlands for agricultural production has stimulated considerable interest in restoring previously farmed wetlands for conservation purposes (Knutson and Euliss, 2001). To determine the impact of this practice on the storage of atmospheric carbon, we compared the soil organic carbon content of undisturbed PPR wetlands to those with a previous history of cultivation. We also calculated a regional estimate of the potential carbon storage in wetlands because the PPR is within the zone (i.e., below 51°N) thought to represent a large



Fig. 1. The prairie pothole region of North America.

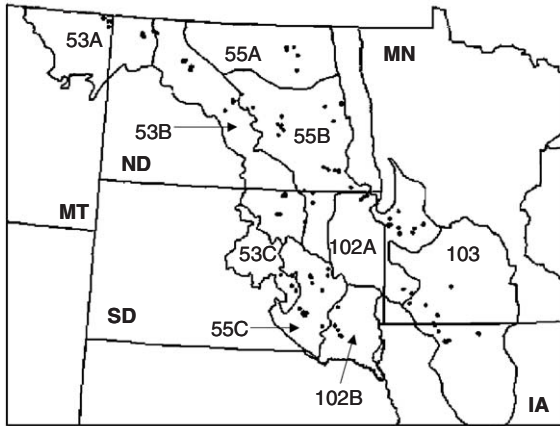


Fig. 2. Locations of wetlands sampled for soil organic carbon within Major Land Resource Areas (MLRAs) within the United States prairie pothole region. Coteau (terminal moraine) regions include MLRAs 53A, 53B, 55C, and 102B and the Glaciated W.B. Plains (ground moraine) includes MLRAs 55A, 55B, 102A, and 103.

terrestrial carbon sink (Fan et al., 1998). We compared this regional estimate of potential carbon storage that would occur by restoring previously farmed wetlands to an estimate of converting all agricultural cropland in the region to no-till management. Since it is unlikely that either land-use change would be fully implemented, we also compared rates of carbon sequestration in restored wetlands to that of no-till cropland.

2. Methods

2.1. Experimental design and soil sampling

Field measurements of soil organic carbon were made in 174 wetlands in the United States PPR that included the glaciated portions of Montana, North and South Dakota, Minnesota, and Iowa (Fig. 2). We used a systematic sampling design stratified by physiographic region (i.e., Coteau and Glaciated Plains; allocation of sampling points was proportional to the linear length of each physiographic region) to select representative random wetlands along the northwest-southeast climatic and land-use gradients in the PPR. Near each sampling point ($n=21$), we selected one seasonal and one semipermanent palustrine emergent depressional wetland (Cowardin et al., 1979) in each of the following land-use categories: (1) restored

wetlands <5 years old in the U.S. Department of Agriculture's (USDA) Conservation Reserve Program (CRP) or similar grasslands (i.e., idled farmed land planted back to perennial grasses), (2) restored wetlands >5 years old in CRP-type habitats, (3) drained wetlands in CRP-type habitats, (4) nondrained wetlands in CRP-type habitats, and (5) reference wetlands in native grassland with no history of cultivation in the wetland basin or the surrounding catchment. We restricted the area of seasonal wetlands to 0.4–0.8 ha and semipermanent wetlands to 2.5–5.5 ha; area range selection criteria represented the mean basin size (± 1 standard deviation) of seasonal and semipermanent wetlands in North and South Dakota (National Wetland Inventory Data; see <http://www.nwi.fws.gov>). Ideally our sampling effort would have resulted in the selection of 210 wetlands (Table 1), but certain wetland categories were not available near every sampling point.

We collected soil samples along four randomly established transects that radiated from the wetland center to the outer edge of the wet-meadow zone (Stewart and Kantrud, 1971). We collected soil samples to a depth of 30 cm from each vegetative zone (i.e., wet-meadow, shallow-marsh, and deep-marsh zones) bisected by transects. Samples from each zone within a wetland were composited by 0 to 15 and 15 to 30 cm depth increments. On a subsample of wetlands ($n=70$), we collected soil cores to a depth of 30 cm for bulk density (total solids per unit volume; g cm^{-3}) determination from the wet-meadow and shallow-marsh zones. We determined total C with a LECO model CN2000 analyzer (LECO Corporation, 1994a,b) and inorganic C using the volumetric meth-

Table 1
Wetlands sampled for soil organic carbon in the United States prairie pothole region in 1997

Wetland class ^a	Treatments ^b			
	Reference	Nondrained	Restored	Drained
Seasonal	21	21	38	19
Semipermanent	19	17	27	12

^a Wetland classification according to Cowardin et al. (1979).

^b Reference wetlands have no history of cultivation in their wetland basin or surrounding catchments; nondrained, restored, and drained wetlands have a history of cultivation and are in Conservation Reserve Program (CRP) or similar grasslands (i.e., prior farmed lands planted back to perennial cover).

Table 2

Regression equations used to predict bulk density from organic carbon concentration

Wetland class	Wetland zone ^a	Depth (cm)	<i>n</i>	Regression equation ^b	<i>r</i> ²	<i>P</i> value
Seasonal	WM	0–15	38	1.3433–0.0713 × OC	0.25	0.0015
Seasonal	WM	15–30	38	1.3652–0.0781 × OC	0.23	0.0021
Seasonal	SM	0–15	38	1.3596–0.0901 × OC	0.64	<0.0001
Seasonal	SM	15–30	38	1.4588–0.0105 × OC	0.42	<0.0001
Semipermanent	WM	0–15	32	1.3030–0.0637 × OC	0.34	0.0005
Semipermanent	WM	15–30	32	1.3932–0.0945 × OC	0.33	0.0006
Semipermanent	SM, DM ^c	0–15	32	1.1747–0.0517 × OC	0.16	0.0243
Semipermanent	SM, DM	15–30	32	1.3926–0.0820 × OC	0.31	0.0009

^a WM=Wet Meadow; SM=Shallow Marsh; DM=Deep Marsh.

^b OC=Organic Carbon concentration.

^c Equation derived from shallow-marsh zone was used to predict bulk density for deep-marsh zone.

od of Wagner et al. (1988). Organic C was determined as the difference between total C and inorganic C. Organic C concentration for each wetland was adjusted for soil bulk density to estimate Mg OC ha⁻¹ for each depth increment and vegetative zone. For wetlands where bulk density was not directly measured, we estimated bulk density from the regression relationship between bulk density and carbon concentration (Table 2). We calculated percent area (ha) of wetland zones from a total station topographic survey of each wetland. Carbon estimates for each zone were weighted (i.e., multiplied) by percent zone area and summed to provide an estimate of Mg OC ha⁻¹ for each depth increment.

Analysis of carbon estimates indicated that carbon content only differed in the surface 15 cm with greater carbon in reference wetlands than in previously farmed wetland categories (i.e., drained, restored, and nondrained wetlands; Table 3). Since carbon did not differ among wetlands with a farmed history, we combined these wetland types into a single category, called ‘farmed’ wetlands (Table 4).

2.2. Estimating wetland carbon stores in the United States and Canada

To estimate regional carbon stores, we applied our estimates of carbon stores in natural and restored wetlands to land use and area estimates for the United States and Canada. We used the 1997 National Resources Inventory (NRI; U.S. Department of Agriculture, 2000) data to estimate wetland and upland areas by broad land-use category within each Major Land Resource Area (MLRA; U.S. Department of Agriculture, 1981) (Fig. 2) in the United States PPR. To estimate wetland and upland areas by land-use category in the PPR of Canada, we overlaid the Canadian wetland inventory (Ducks Unlimited Canada, 1986) and land cover data (Prairie Farm Rehabilitation Administration, 1995). Unlike the NRI database, the Ducks Unlimited Canada (1986) wetland inventory lacks estimates of drained wetlands. Estimates of wetlands altered or lost due to agricultural drainage in the Canadian PPR range from 71% to 75% (Environment Canada, 1986;

Table 3

Comparison of soil organic carbon (Mg ha⁻¹) among treatments by depth for 174 wetlands in the prairie pothole region of the United States, 1997

Treatment	0 to 15 cm ^a		15 to 30 cm ^b		0 to 30 cm ^c	
	Mean	95% C.I.	Mean	95% C.I.	Mean	95% C.I.
Drained	51.9 ¹	(46.4–57.4)	47.2 ¹	(42.2–52.1)	99.1 ¹	(89.4–108.7)
Nondrained	52.6 ¹	(47.6–57.5)	43.7 ¹	(39.2–48.2)	96.3 ¹	(87.5–105.0)
Restored	51.9 ¹	(48.1–55.7)	44.6 ¹	(41.2–48.0)	96.5 ¹	(89.8–103.2)
Reference	62.2 ²	(57.4–67.1)	43.9 ¹	(39.5–48.2)	106.1 ¹	(97.6–114.6)

Means within column followed by a common number are not significantly different ($P < 0.05$).

ANOVA model probabilities for treatment main effects: ^a($F_{3, 170} = 4.41, P = 0.005$), ^b($F_{3, 170} = 0.43, P = 0.729$), ^c($F_{3, 170} = 1.21, P = 0.309$).

Table 4

Comparison of soil organic carbon (Mg ha⁻¹) among farmed and reference wetlands by depth for 174 wetlands in the prairie pothole region of the United States, 1997

Treatment	0 to 15 cm ^a		15 to 30 cm ^b		0 to 30 cm ^c	
	Mean	95% C.I.	Mean	95% C.I.	Mean	95% C.I.
Farmed	52.1 ¹	(49.5–54.7)	44.9 ¹	(42.6–47.3)	97.0 ¹	(92.4–101.7)
Reference	62.2 ²	(57.4–67.1)	43.9 ¹	(39.5–48.2)	106.1 ¹	(97.6–114.6)

Means within column followed by a common number are not significantly different ($P < 0.05$).

ANOVA model probabilities for treatment main effects: ^a($F_{1, 172} = 13.3, P = 0.0003$), ^b($F_{1, 172} = 0.19, P = 0.667$), ^c($F_{1, 172} = 3.44, P = 0.065$).

Young, 1994). To adjust for losses not accounted for in the wetland inventory, we assumed a conservative 71% wetland loss. The wetland area estimate, unadjusted for wetland loss, indicated that 42.5% of the total wetland area in the PPR occurred in Canada; hence, the adjusted estimate of 71.8% is more reasonable because 67.5% of the PPR occurs in Canada (Table 5).

Organic soil carbon estimates for the surface 15 cm of farmed and reference wetlands (i.e., wetlands with no prior history of cultivation) we sampled within each MLRA were averaged and used to estimate carbon stores of palustrine wetland soils (Table 6). For palustrine wetlands in cropland and in restored grassland (e.g., U.S. Department of Agriculture's Conservation Reserve Program), farmed wetland carbon estimates were used to estimate current carbon stores, and reference wetland estimates were used to estimate historic carbon stores. To estimate carbon stores in wetland soil in the Canadian PPR, we applied our estimates of average carbon stores in reference and farmed wetlands in the United States (Table 7) to wetland areas in Canada.

3. Results

3.1. Potential carbon storage in restored wetlands

To estimate the carbon storage potential of wetland restoration activities, we projected, over a 10-year period, the amount of carbon that would be stored if all cropland wetlands were restored in the PPR. We considered three types of carbon storage processes: (a) replenishment of soil carbon in the 0–15 cm soil profile, (b) carbon stored by mass accumulation of sediments, and (c) carbon stored in the living plant community. Analysis of carbon estimates demonstrated that, relative to reference wetlands, farmed wetlands have lost on average 10.1 (95% C.I.=4.7 to 15.6) Mg OC ha⁻¹ in the surface 15 cm of soil (Table 4). Our research also showed that when semi-permanent wetlands (Stewart and Kantrud, 1971) are restored, carbon in the surface 15 cm is replenished at a rate of 3.05 Mg OC ha⁻¹ year⁻¹ (Fig. 3). Based on this rate of increase, it would take an average of 3.3 years (95% C.I.=1.5 to 5.1) for carbon lost in the surface 15 cm due to cultivation to be replenished

Table 5

Terrestrial and aquatic areas in the prairie pothole region of North America

System	U.S. PPR, 10 ³ ha (SE)	Canada PPR, 10 ³ ha	Total PPR, 10 ³ ha	Percent of total area (10 ³ ha)	
				U.S.	Canada
Palustrine	5678 (63.6)	2635	8313	68.3	31.7
Lacustrine	543 (4.3)	1959	2502	21.7	78.3
Subtotal	6221 (64.3)	4594	10,815	57.5	42.5
71% loss ^a	na	11,247	na	na	na
Total aquatic	6221 (64.3)	15,841	22,062	28.2	71.8
Terrestrial	24,239 (101.7)	47,433 ^b	71,672	33.8	66.2
Grand total	30,460 (93.3) ^c	63,274	93,734	32.5	67.5

^a Estimate of drained or altered wetlands in Canada not accounted for by Ducks Unlimited Canada (1986). This estimate assumes that the wetland area accounted for in the inventory (i.e., subtotal=4594) represent only 29% (Environment Canada, 1986) of the original wetlands.

^b Terrestrial area (58,680) minus the estimated drained wetland area (11,247) not accounted for by Ducks Unlimited Canada (1986).

^c Does not include riverine wetlands (132 × 10³ ha) or Federal land (347 × 10³ ha).

Table 6

Estimate of soil organic carbon (Mg ha^{-1}) in the surface 15 cm of farmed and reference wetlands within Major Land Resource Areas (U.S. Department of Agriculture, 1981) within the prairie pothole region of the United States

MLRAs	Farmed wetlands			Reference wetlands		
	Mean	95% C.I.	<i>n</i>	Mean	95% C.I.	<i>n</i>
102A	48.1	(40.9–55.2)	16	73.4	(54.6–92.3)	4
102B	36.4	(32.4–40.4)	5	38.9	(15.5–62.3)	2
103	56.2	(48.6–63.9)	20	76.7	(70.0–83.5)	7
53A	58.2	(47.9–68.6)	15	59.1	(26.3–92.0)	4
53B	58.0	(52.4–63.7)	27	59.1	(42.6–75.6)	10
53C	50.9	(44.4–57.5)	7	54.9	(00.0–132.6)	2
55A	54.3	(46.7–61.9)	8	62.3	(00.0–150.4)	2
55B	48.9	(43.5–54.4)	21	67.5	(61.0–74.0)	5
55C	43.0	(35.7–50.4)	15	45.4	(29.8–61.0)	4

through restoration. Reference seasonal wetlands (Stewart and Kantrud, 1971) had significantly more organic carbon in the upper 15-cm of soil ($63.5 \text{ Mg OC ha}^{-1}$) than farmed seasonal wetlands ($51.9 \text{ Mg OC ha}^{-1}$; $F_{1,97}=10.7$, $P=0.0015$). However, we were unable to detect a temporal increase in soil carbon for restored seasonal wetlands, a wetland class with short hydroperiods that is dry more often than semipermanent wetlands on a seasonal and inter-annual basis. This finding suggests that our sample size or the temporal replication was inadequate to detect the increase in soil carbon in seasonal wetlands. Because NRI lumps wetland water regimes into a single category (i.e., palustrine versus seasonal or semipermanent), we assumed that carbon stocks in

Table 7

Estimate of historic and potential carbon storage (Tg) in cropland and wetland in the prairie pothole region of North America

System/country land cover	Area ^a 10^3 ha (SE)	OC Stores ^b		Historic ^c OC loss (SE)	OC gain ^d (first 5 years)	OC gain ^e (next 5 years)	OC ^f standing crop	OC gain ^g (10 year total)
		Past (SE)	Present (SE)					
Terrestrial/USA								
Cropland	16,129 (112.6)	1037	726	311	36	36	na	72
Other	8,110 (92.3)							
Terrestrial/Canada								
Cropland	17,958	1128	790	338	40	40	na	80
Other	29,475							
Total terrestrial	71,672	2165	1516	649	76	76	na	152
Aquatic/USA								
Cropland wetland	3844 (50.7)	281 (9.0)	209 (10.7)	72 (14.0)	72	16	27	115
Other	2377 (39.3)							
Aquatic/Canada								
Cropland Wetland	12,400	771 (34.3)	646 (15.8)	125 (37.8)	125	51	87	263
Other	3441							
Total aquatic	22,062	1052	855	197	197	67	114	378

^a Area by system (terrestrial and aquatic) and land-use (cropland and other) in the USA based on the National Resources Inventory data (U.S. Department of Agriculture, 2000), and for Canada, the Ducks Unlimited wetland inventory (Ducks Unlimited Canada, 1986) and Prairie Farm Rehabilitation Administration land cover data (Prairie Farm Rehabilitation Administration, 1995). Land cover within “other” includes: native and non-native grasslands, forests, other non-cultivated lands, and 66×10^3 ha of cultivated cropland with incomplete soil records to estimate carbon stores.

^b Baseline estimates (i.e., present stores) of carbon in croplands of the USA and Canada were estimated using 1997 NRI data and the Canadian Soil Organic Database (Lacelle, 1996). Past stores (prior to cultivation) were estimated by assuming a 30% loss of carbon (e.g., present stores/0.7). Present and past stores in cropland wetlands were estimated using information presented in this paper.

^c Loss of carbon due to agricultural cultivation (past stores–present stores).

^d Expected increase in carbon in the surface 15 cm after 5 years if cropland was placed in no-till (assumes a 1% annual increase) and expected increase if cropland wetlands were restored (based on information presented in this paper).

^e Increase in carbon for cropland based on the same relationship as during the first 5 years. For wetlands, increase represents mass accumulation of carbon ($0.830 \text{ Mg OC ha}^{-1} \text{ year}^{-1}$) for 5 years.

^f Carbon stored in developing plant community of restored wetlands (based on 7 Mg OC ha^{-1}). This is not applicable (Na) to cropland in no-till because annual crop produced is harvested.

^g Expected total carbon stored after 10 years if all cropland was placed in no-till and all cropland wetlands were restored.

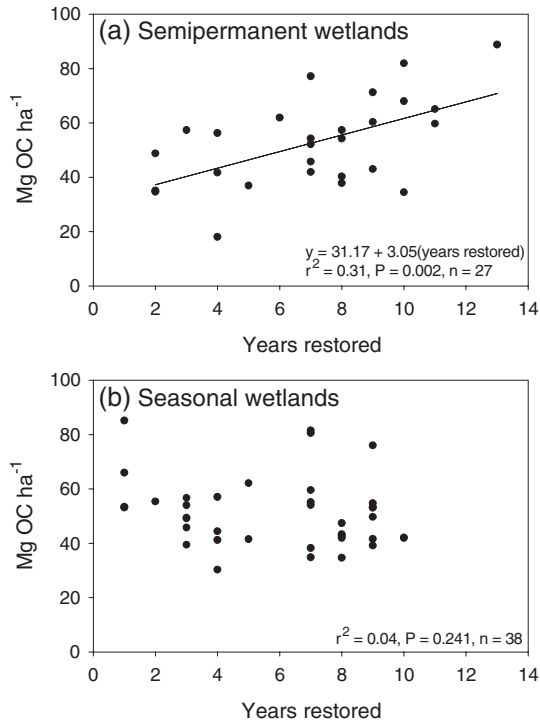


Fig. 3. Relationship between soil organic carbon in the surface 15 cm and years restored for (a) semipermanent and (b) seasonal wetlands.

all palustrine wetlands would return to reference condition within 5 years. Based on this assumption, restoration of all cropland wetlands would result in the storage of 197 Tg of soil organic carbon during the first 5 years following restoration (Table 7).

We estimated mass accumulation of carbon in reference wetland soil at $0.830 \text{ Mg ha}^{-1} \text{ year}^{-1}$ based on an average sedimentation rate of 2 mm year^{-1} (Gleason, 2001) and a carbon content of 62.2 Mg ha^{-1} we found in reference wetlands (Table 4). We assumed that restored farmed wetlands develop mass accumulation rates similar to reference wetlands; hence, in years 5–10 post-restoration, an additional 67 Tg of carbon would be sequestered (Table 7). Finally, the vegetative community that develops in restored wetlands represents an additional pool of carbon. Above and below ground annual biomass of plants in wetlands range from 7 to 17 Mg OC ha^{-1} (McDougal, 2001; Wetzel, 2001); however, much of the below ground biomass likely contributes to the soil organic carbon pool. Hence, we

used the estimate of 7 Mg OC ha^{-1} for above ground biomass and estimated that 114 Tg of carbon would be associated with the standing crop of wetland vegetation (Table 7). In total, 378 Tg of organic carbon can potentially be stored in the soil, sediment accumulation, and in the plant community of restored wetlands over a 10-year period.

3.2. Potential carbon storage in upland

We used the 1997 NRI database and methods described in Lal et al. (1995, 1998) to estimate potential organic carbon storage under no-till management in the upper 15 cm of cropland soil within MLRAs in the United States PPR. We used these baseline estimates of organic carbon in cropland to project past and potential land-management scenarios. The literature indicates that 20% to >50% of the organic carbon of native prairie is lost when converted to cultivated agriculture (Mann, 1986; Cihacek and Ulmer, 1995). Assuming that, on average, cultivation has resulted in a 30% loss of carbon, cropland in the United States PPR has lost 311 Tg of organic carbon (Table 7). Converting conventionally farmed cropland to no-till agriculture would increase soil carbon by about $1\% \text{ year}^{-1}$ or $200 \text{ to } 300 \text{ kg C ha}^{-1} \text{ year}^{-1}$ (Lal et al., 1998). Using this annual rate of increase, we estimated that 72 Tg of carbon would be stored over a 10-year period if all cropland in the United States PPR was converted to no-till management (Table 7).

To make similar calculations for Canadian cropland, we used the Canadian Soil Organic Carbon Database (CSOCD) (Lacelle, 1996), in combination with the Canadian land cover data (Prairie Farm Rehabilitation Administration, 1995), to estimate the carbon storage in cropland by broad land-use category. The CSOCD database includes surface soil carbon estimates to a 30-cm depth. Studies indicate that approximately 55% to 60% of the organic carbon content within the surface 30 cm occurs in the upper 15 cm (Gebhart et al., 1994; Follett et al., 1997). Because the CSOCD database is for the surface 30 cm, we estimated carbon in the surface 15 cm by multiplying carbon estimates within each land-use category by 0.57. We then applied the procedures we used for the United States PPR to estimate carbon loss in cropland (338 Tg) and carbon potentially stored (80 Tg) by conversion to no-till agriculture in

Canada (Table 7). In total, we estimate that 152 Tg of soil organic carbon could be sequestered under no-till management in the PPR of the United States and Canada.

4. Discussion and conclusions

Numerous carbon sinks contribute to overall storage of atmospheric CO₂ in the terrestrial Biosphere (Pacala et al., 2001) and current research is focused on the identification and management potential of a diversity of sinks, both biological and geological. Our study demonstrates that prairie wetlands are an important and previously overlooked biological carbon sink in North America. The finding that prairie wetlands have the potential to sequester more than twice as much carbon as conversion of all cropland to no-till agriculture (378 Tg versus 152 Tg; Table 7) is especially significant because cropland wetlands comprise only 17% of the land area. Wetlands are the most productive terrestrial ecosystem in the Biosphere (Whittaker and Likens, 1973). Although it is highly unlikely that wetland restoration or no-till agriculture will be fully implemented in the PPR, our results show that the rate of carbon recovery in prairie wetlands is especially high relative to projections for no-till cropland. On a global scale, wetlands comprise the largest pool of stored carbon, representing 33% of the soil organic matter on only about 4% of the land surface area (Eswaran et al., 1993). Cropland in the PPR has been depleted of organic matter through conventional agriculture (Mann, 1986; Lal et al., 1995; Cihacek and Ulmer, 1995) and the rate of sequestration is low under no-till agriculture, averaging about 1% annually (Lal et al., 1998). However, no-till is the most effective means of sequestering carbon in cropland while maintaining crop production (Lal et al., 2004).

While prairie wetlands are important for carbon storage, there are valid concerns over the release of greenhouse gases (GHG) such as methane (CH₄) and nitrous oxide (N₂O). Wetlands appear to be the largest source of CH₄, contributing about 20% of the annual global emission to the atmosphere (Wang et al., 1996). Methane is a very potent GHG with a global warming potential (GWP) of 21 (Intergovernmental Panel on Climate Change, 1996), and it is naturally emitted by

wetlands (Mitsch and Gosselink, 2000). Nitrous oxide is an even more potent GHG with a GWP of 310 (Intergovernmental Panel on Climate Change, 1996) and it persists about as long as CO₂ in the atmosphere, approximately 120 years whereas CH₄ only lasts for 12–15 years (U.S. Environmental Protection Agency, 2003). While nitrogen enrichment of ombrotrophic mires apparently has little or no effect on methane emission (Hutchin et al., 1996), data from a glaciated region in northeastern Germany similar to the North American PPR suggest that enrichment from nitrogen fertilizers and accelerated mineralization of soil organic matter elevate the emission of CH₄ and N₂O (Merbach et al., 2002). Findings from Germany are consistent with other studies that demonstrate that nitrogen fertilization enhances emission of N₂O (Thornton and Valente, 1996; Davidson et al., 2000).

Restored wetlands in North America are generally sited within marginal farmlands that have been voluntarily idled by private landowners (e.g., Conservation Reserve Program of the U.S. Farm Bill) and reseeded to perennial grasses; hence, they receive little or no enrichment from agricultural fertilizers. In comparison, most remaining wetlands in the PPR are situated within agricultural fields where they are farmed except during extremely wet periods. Hence, most wetlands in the PPR at the current time receive runoff from surrounding agricultural areas that have been shown to exacerbate CH₄ and N₂O emission rates (Merbach et al., 2002). Consequently, converting cultivated cropland to perennial vegetation within restored wetland catchments should reduce nutrient enrichment in restored wetlands and lower emissions of N₂O, and possibly CH₄ from wetland basins. Further, the conversion of cropland to perennial grassland has been shown to reduce CH₄ emission from upland soils (Keller et al., 1990; Dorr et al., 1993; Dobbie and Smith, 1994; Parashar et al., 2001). Hence, the conversion of cropland to perennial grassland when wetlands are restored should reduce overall CH₄ emission.

Our results are consistent with the literature that cultivation is the most important factor in soil carbon loss (Lal et al., 2004). However, the restoration of wetland hydrology (e.g., plugging artificial drains) also is a critical component of restoration. The fact that carbon storage is enhanced under anoxic conditions is important because flooded wetlands provide

optimal conditions for accretion of organic matter. Our study suggests that the overall organic carbon content of prairie wetlands is comparable to native grasslands (Blank and Fosberg, 1989), but the rate of carbon sequestration is approximately five times higher in restored wetlands than restored grassland (Follett et al., 2001). Upland catchment areas, normally re-vegetated when wetlands are restored, clearly provide additional carbon storage. The rate of carbon sequestration in wetlands will decrease over time as carbon storage in restored wetlands replenishes lost carbon stores. Carbon accretion in restored grassland proceeds slowly, but overall storage should exceed that of wetland soils because it comprises a substantially larger land area. Reduction in GHG emission in restored wetlands should provide long-term GHG reduction benefits; GHG emissions associated with the previous agricultural land-use change should be reduced.

The annual fossil CO₂ emission for North America in 1990 was 1.6 Pg C (Fan et al., 1998). Hence, wetlands in the PPR of North America have potential to offset 2.4% of the annual fossil CO₂ emissions over a 10-year period. This estimate may be low because detailed information on carbon loss by major land areas was not available for Canada. Multiplying our estimate of 10.1 Mg OC ha⁻¹ for mean carbon loss for cropped wetlands in the United States by the area of cropped wetlands in the United States and Canada (Table 4) suggests that 38 Tg and 125 Tg of organic carbon have been lost from wetlands in the United States and Canada, respectively. However, the 72 Tg loss we report for the United States (Table 4) accounted for spatial variation in carbon loss associated with specific MLRAs that ranged from <1 to 25 Mg ha⁻¹ (Table 1). In contrast, our estimate for Canada did not account for differences in climate, precipitation, and land-use that vary spatially and influence soil organic carbon content.

The most recent inventory of GHG emissions and sinks for the United States (U.S. Environmental Protection Agency, 2003) reported restored prairie wetlands as carbon sinks but our study is the first to document the magnitude of this sink. Much additional research is needed to refine our estimates of sink size and its overall importance. Specific information on carbon loss within major land-use areas in prairie Canada is needed to fully assess the overall size of

this carbon sink. Further, research on other wetland types and relationships of grassy buffer to GHG emissions will be required before optimal land-use strategies can be developed to help mitigate global climate change.

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