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Toward improved coefficients for predicting direct N_2O emissions from soil in Canadian agroecosystems

B.L. Helgason[1,*], H.H. Janzen[2], M.H. Chantigny[6], C.F. Drury[3], B.H. Ellert[2], E.G. Gregorich[4], R.L. Lemke[5], E. Pattey[4], P. Rochette[6] and C. Wagner-Riddle[7]
*[1]Agriculture and Agri-Food Canada (AAFC), 107 Science PL S7N OX2 Saskatoon, Sk, Canada; [2]AAFC, Lethbridge, Ab, Canada; [3]AAFC, Harrow, On, Canada; [4]AAFC, Ottawa, On, Canada; [5]AAFC, Swift Current, Sk, Canada; [6]AAFC, Ste-Foy, PQ, Canada; [7]University of Guelph, On, Canada; *Author for correspondence (tel: +1-306-975-10; fax: +1-306-966-4226; e-mail: helgasonb@agr.gc.ca)*

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Abstract

Agricultural soils emit nitrous oxide (N_2O), a potent greenhouse gas. Predicting and mitigating N_2O emissions is not easy. To derive national coefficients for N_2O emissions from soil, we collated over 400 treatment evaluations (measurements) of N_2O fluxes from farming systems in various ecoregions across Canada. A simple linear coefficient for fertilizer-induced emission of N_2O in non-manured soils (1.18% of N applied) was comparable to that used by the Intergovernmental Panel on Climate Change (IPCC) (1.25% of N applied). Emissions were correlated to soil and crop management practices (manure addition, N fertilizer addition and inclusion of legumes in the rotation) as well as to annual precipitation. The effect of tillage on emissions was inconsistent, varying among experiments and even within experiments from year to year. In humid regions (e.g., Eastern Canada) no-tillage tended to enhance N_2O emissions; in arid regions (e.g., Western Prairies) no-tillage sometimes reduced emissions. The variability of N_2O fluxes shows that we cannot yet always distinguish between potential mitigation practices with small (e.g., <10%) differences in emission. Our analysis also emphasizes the need for developing consistent experimental approaches (e.g., 'control' treatments) and methodologies (i.e. measurement period lengths) for estimating N_2O emissions.

Introduction

Nitrous oxide (N_2O) is the fourth most important anthropogenic contributor to atmospheric warming after water vapor, carbon dioxide and methane. Globally, the atmospheric concentration of N_2O has increased by 46 ppb (17%) since 1750 and is now increasing at a rate of 0.25% yr^{-1} (IPCC 2001). Nitrous oxide accounts for about 5% of the total atmospheric greenhouse effect. In addition to its effects on global warming, nitrous oxide reacts with stratospheric ozone and contributes to increased UV-B intensity at the earth's surface (Socolow 1999). Globally, 70% of annual anthropogenic N_2O emissions are attributed to animal and crop production (Mosier 2001).

In Canada, annual N_2O emissions are estimated to be $\sim$ 108 Gg N, of which agriculture accounts for $\sim$ 76 Gg. Agricultural soils alone represent $\sim$ 55% of total Canadian agricultural N_2O emissions. In the period between 1990 and 2000, according to current estimates, N_2O emissions from Canadian soils increased by 23% (Olsen et al. 2002), largely because of increasing

amounts of N additions from inorganic fertilizer and manure.

Currently, Canadian GHG inventories are based on the IPCC method (Houghton et al. 1997) and thus, direct soil N_2O emissions are estimated simply as a function of N input (N_2O–N = 0.0125*N input). Both the N inputs data and the emission factors remain uncertain (> ±30% for agricultural soil emissions) (Olsen et al. 2002). Estimating N_2O emissions from agroecosystems is difficult given the complexity of elements which influence the production of N_2O through both time and space. Although all major N inputs are considered in international accounting methodology, the IPCC Guidelines (Houghton et al. 1997) do not account for the specific impacts of climate, N source, soil type or cropping systems due to the lack of sufficient information (Mosier et al. 1998).

Deriving coefficients that predict emissions as a function of agroecosystem properties and crop and livestock production practices is not easy. Elucidating the effects of individual factors on N_2O emissions from soil will require more research; for now, scientists and policy makers are often forced to rely on rudimentary first approximations. Developing country-specific methodologies is an important next step in global GHG accounting. In this paper, we evaluate Canadian research data as a first step toward deriving simple national coefficients for predicting N_2O emissions from agroecosystems. This collation of Canadian flux measurements will also help us to identify fruitful research areas promoting the development of better methods for estimating N_2O emissions in Canadian agroecosystems.

Approach

N_2O data were collected from research trials conducted in the last 10 years at diverse agricultural areas across Canada (Table 1). The dataset encompassed ca. 400 treatment-years from different cropping systems, tillage practices, N fertilizer types and rate applications (one treatment measured over 3 years equals 3 'treatment-years'). The research sites were situated in a wide range of climatic zones in Alberta, Saskatchewan, Ontario and Québec, and spanned soil textures from loamy sand to clay. The majority of measurements were made using collection chambers placed on the soil surface (both 'open' and 'closed'). More detailed information regarding measurement techniques can be found in the publications and reports listed in Table 1. The

Table 1. Summary of locations, precipitation, soil texture and measurements for experimental sites.

Location	Precipitation (mm yr^{-1})	Texture class	Measurements[1]
Québec City, PQ	1174	Sandy Loam	12[a,b]
		Loamy sand	6[b]
		Silty clay loam	4[a]
		Loam	14[a,c,d]
		Clay	12[a,b]
St-Marc, PQ	1034	Sandy loam	4[e]
		Loam	4[e]
		Clay	4[e]
Montreal, PQ	1020	Sandy loam	12[f]
		Silty clay	9[f]
		Clay	21[f]
St-Jean sur Richelieu, PQ	1034	Clay loam	2[a]
Ottawa, ON	879	Sandy loam	4[a]
		Loam	7[a,g]
		Clay loam	3[h,i]
Elora, ON	867	Silty loam	34[j,k]
Guelph, ON	867	Loam	20[l,m]
Woodslee, ON	876	Clay loam	18[n]
Melfort, SK	411	Clay	30[o]
Indian Head, SK	427	Clay	30[o]
Scott, SK	355	Loam	30[o]
Swift Current, SK	349	Sandy loam	3[p]
		Loam	36[o]
		Silty loam	28[a,p,q]
Three Hills, AB	490	Clay loam	26[a]
Lethbridge, AB	Irrigated	Clay loam	16[a]
Cooking Lake, AB	450	Silty loam	7[r]
Breton, AB	550	Silty loam	11[r]
Ellerslie, AB	450	Clay loam	14[r]

[1]The term 'measurement' was used to define the measured emission (kg N_2O–N ha^{-1}) from one treatment (mean of all experimental replicates) over the course of the associated measurement period. For example:
Measurement: the mean cumulative flux of 36 chamber measurements for one field treatment (e.g., 50 kg ha^{-1} urea) (4 replicate plots), taken from April through October of one year (e.g., 2001).
[a]Unpublished data; [b]Rochette et al. 2004; [c]Rochette et al. 2000; [d]Chantigny et al. 2001; [e]Rochette et al. 2003; [f]MacKenzie et al. 1998; [g]Lessard et al. 1996; [h]Grant and Pattey 1999; [i]Grant and Pattey 2003; [j]Wagner-Riddle et al. 1997; [k]Wagner-Riddle and Thurtell 1998; [l]Maggiotto et al. 2000; [m]Maggiotto and Wagner-Riddle 2001; [n]Drury et al. 2003; [o]Lemke et al. 2003b; [p]Lemke et al. 2003a; [q]Lemke et al. 2001; [r]Lemke et al. 1999.

duration and timing of measurements varied among sites (Figure 1). While this disparity likely influenced the reported N_2O fluxes, we present the values as reported, rather than try to normalize them retroactively. Similarly, we did not extrapolate flux data to estimate annual emissions, but report cumulative fluxes from each measurement interval. The term 'measurement' denotes the cumulative emission (kg N_2O–N ha^{-1}) from one treatment (mean of all experimental replicates) over the measurement interval. For example, in some studies one 'measurement' refers to the mean cumulative flux calculated for flux determinations on 36 days from April through October in a given year for one field treatment (4 replicates); in other studies it might refer to cumulative flux calculated from determinations on 30 sampling days during an entire year.

N_2O emissions are inherently variable (Grant and Pattey 1999; Ball et al. 2000; Yanai et al. 2003; Šimek et al. 2004). As such, we did not exclude 'outliers' in the data set, except in two extreme cases. At one site (Rochette et al. 2003), under barley receiving 60 kg N ha^{-1}, N_2O emissions were as high as 44 kg N_2O–N ha^{-1} yr^{-1}; all treatments were excluded from this site. At another site (Ellert, unpublished), one replicate was excluded because its emissions were consistently 5–30 times higher than those in other replicates for the identical treatment on apparently uniform field plot experiments. The origin of the high emissions from these 'hot-spots' awaits further research.

In paired comparisons of legume and non-legume crops, different rates of N fertilizer were applied to legumes and non-legume crops ('controls'). Legume crops often received no fertilizer N so we were not able to estimate N_2O emission per unit N applied. Where more than one non-legume crop was present within a particular field trial in a given year (e.g., wheat and barley), pea:wheat and pea:barley comparisons were made. Regression analysis and analysis of variance (ANOVA) was performed using Proc REG, Proc STEPWISE and Proc GLM in SAS (SAS Institute 1999).

Deriving simple N_2O emission coefficients

International GHG accounting of direct N_2O emissions from agricultural soils relies on linear relationships between N_2O emitted and N applied (Bouwman 1996). We used a similar analysis to evaluate this approach for N_2O estimation under Canadian conditions. Manured and non-manured soils were considered separately, because the rates and patterns of emission from these soils may differ (Lessard et al. 1996; Velthof et al. 2003).

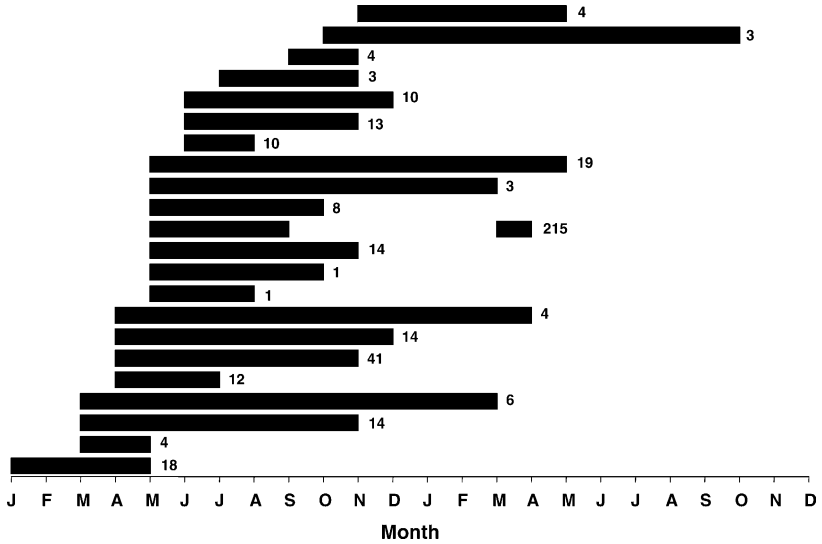

Figure 1. Frequency, length and temporal distribution of N_2O flux measurements. Temporal distributions of the measurements are approximately represented by the length of the bars, and the total number of measurements made for each specified time period is shown next to the bar.

Non-manured soils

N_2O emissions estimates based on measurements in non-manured cropping systems ($n = 383$) were comparable to estimates given using the IPCC methodology (Figure 2a). A linear fit of all measurements yielded a fertilizer-induced emission (FIE) of 1.18% which is in agreement with the current international guidelines which state an FIE of 1.25% (0.25–2.25%) and with a recent analysis by Bouwman et al. (2002b) which found an FIE of 0.9%.

The *y*-intercept, an estimate of the average emission in soils receiving no N fertilizer, was 0.405 kg N_2O–N ha^{-1} yr^{-1} (significantly different from 0 at $P > 0.0001$). This value reflects emissions from sources such as decomposing organic matter, decomposing crop residues, and (in some cases) biological dinitrogen fixation under legumes. Emissions of N_2O that result from the decomposition of crop residue and biological N-fixation are accounted in the IPCC methodology using additional emissions factors. For the IPCC methodology, Bouwman (1996) estimated a non-anthropogenic ('background') emission value of 1 kg N_2O–N ha^{-1} yr^{-1}. Our 'background' may be lower, in part, because the measurement duration at some sites was less than one year, leading to a conservative emission estimate (Bouwman 1996; Bouwman et al. 2002a). When we excluded data

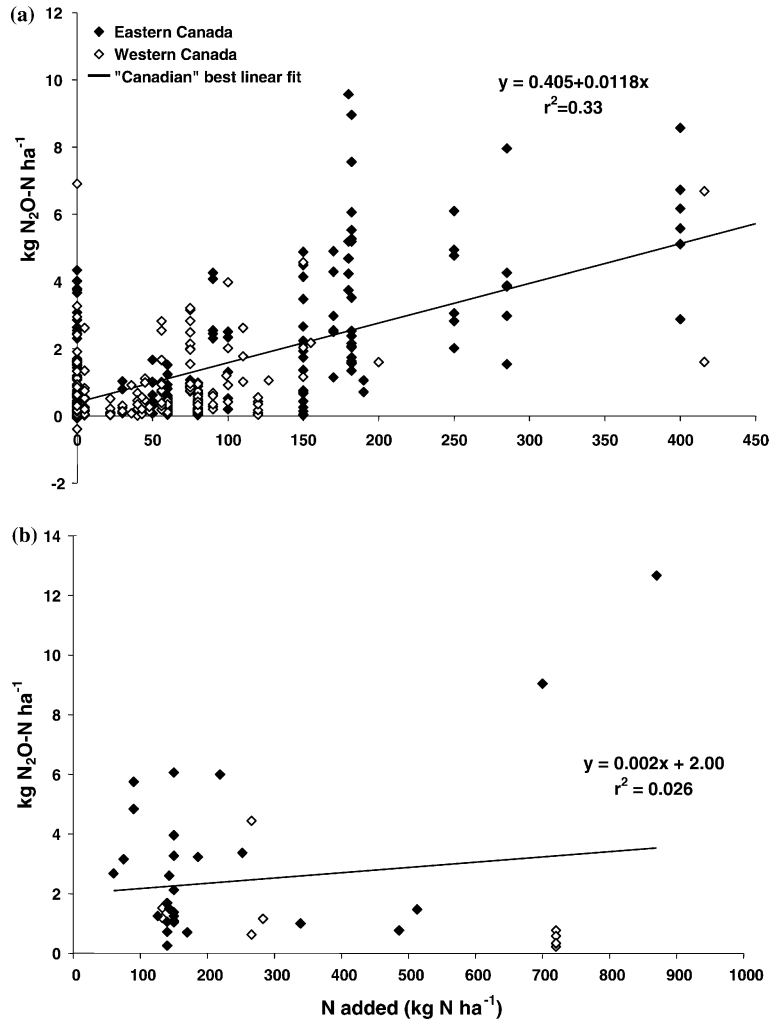

Figure 2. Best linear fit of N_2O emissions measurements where: (a) no manure was applied; (b) manure was applied.

points from very short (less than 5 months) or non-representative (e.g., March–May) measurement periods, and from infrequent measurement intervals ($n = 65$ for removed data), the magnitude of the FIE increased. A best linear fit of this refined data set yielded an equation with slightly lower y-intercept, but higher FIE:

$$N_2O\text{–}N\ (\text{kg ha}^{-1}) = 0.245 + 0.0136^{*}N_a \quad (r^2 = 0.38)$$

where: N_a = N applied (kg ha^{-1})

Manured soils

Linear regression analysis for manured soils demonstrated no significant relationship between N applied as manure and N_2O emissions (Figure 2b). In effect, N_2O emission was independent of the amount of N added in manure (the slope of 0.002 was not statistically significant).

The fit of this equation to the data was poor and did not correspond to emissions derived using the IPCC methodology. Given the disparate experimental conditions and the complicated nature of nutrient cycling in soils receiving organic amendments, this is not surprising. A few data points at high N rates heavily influenced the analysis. In addition, data used in this analysis originated from many different studies and thus there was little or no consistency in experimental conditions. The manures applied ranged from liquid pig slurry to solid beef cattle manure. Annual precipitation values ranged from ca. 450 to 1174 mm yr^{-1}. Differences in the availability of nutrients in the manure, particularly carbon and nitrogen, nutrient availability in the soil as well as moisture conditions have been shown to greatly influence N_2O emissions from manured soils (Lessard et al. 1996; Whalen 2000; Chantigny et al. 2002; Velthof et al. 2003). Differences in site history (consecutive manure applications) can also have a significant influence. Nevertheless, the scatter of data points (Figure 2b) suggests that a single coefficient, as in the IPCC methodology, is unlikely to yield reliable estimates of N_2O emissions, given the diversity of manures and cropping conditions prevalent in Canada.

The lack of relationship between N_2O emissions and manure-N applied confirms the necessity for separate consideration of emissions from manured and non-manured systems. Compared to non-manured soils, the higher N_2O production often seen in manured soils is related to increased organic C and inorganic N in manured soils (Bergstrom et al. 1994), especially in those soils where degradable organic C is limiting before manure application (Granli and Bøckman 1994; Rochette et al. 2000; Velthof et al. 2003). Rapid utilization of available carbon and nitrogen by soil microbial communities may result in increased denitrification. In addition, greater microbial activity resulting from additions of degradable organic C may lead to localized depletion of oxygen at microsites and enhance N_2O production. Higher N_2O emissions in manured soils in Eastern compared to Western Canada implies that denitrification was likely responsible for differences in emissions. Soil moisture levels are often much higher in the humid climatic conditions in Eastern Canada, which can promote denitrification (Sehy et al. 2003).

Predicting N_2O emissions using additional variables

Single linear regression analysis generated very simple coefficients for estimating direct N_2O emissions from soil in Canadian agroecosystems. But application of these simple coefficients under all conditions results in considerable error (Skiba and Smith 2000) and does not provide the ability to compare the effects of management and environmental factors on emissions. A global analysis of N_2O data showed emissions in agroecosystems were dependent on climate, soil factors (texture, drainage, organic carbon content and pH) and management factors (N application rate per fertilizer type and crop type) (Bouwman et al. 2002b). Development of strategies for mitigating N_2O emissions from Canadian soils requires a more specific understanding of the contribution of specific environmental and management factors to N_2O production.

Analysis of variance of all measurements confirmed a significant difference in N_2O emissions between non-manured and manured soils. A second ANOVA also showed a difference in the N_2O emissions from non-legume and legume cropping systems in the non-manured soils. Stepwise regression analysis of average precipitation (mm yr^{-1}) and N applied (kg ha^{-1}) in the three

cropping systems considerably improved the fit of the regression analyses (Table 2), though over half of the variability remained unaccounted for. In non-manured, non-legume systems and in manured systems, N added and precipitation were the most influential predictors of N_2O emission. In legume systems, N added was the only parameter that significantly influenced emissions. Each of these systems are further analysed and discussed separately in the oncoming sections.

Non-manured soils: N_2O from legume vs. non-legume cropping systems

Current international accounting considers N fixed during biological nitrogen fixation as a source of N_2O, in addition to counting N_2O from decomposition of N in legume residues and N applied as fertilizer (Houghton et al. 1997). For the stepwise regression analysis, we compared emissions from *rotations* including legume crops with *rotations* having no legume crops, since biologically-fixed N could influence N_2O emissions in all years of the rotation, not just during the legume phase. A second analysis was performed using paired comparisons of emissions from legume vs. non-legume crops.

The FIE in non-legume systems (0.9%) was less than that in systems that include legume crops (1.2%) (Table 2). Precipitation significantly affected emissions in non-legume systems, accounting for 12% of the variability in emissions data (Table 2), probably through soil moisture effects on denitrification (Sahrawat and Keeney 1986; Bouwman et al. 1993). This effect of soil moisture may be confounded by the fact that wetter systems tend to be more productive and in many cases receive more N inputs. But in legume systems, precipitation had no significant influence on N_2O emission, perhaps because other variables, such as C:N ratio of residues (Huang et al. 2004), superseded the effects of water on denitrification.

A paired treatment comparison of N_2O emissions from legume and non-legume crops demonstrated greater N_2O emissions from non-legumes (Figure 3). This observation likely reflects the confounding effect of fertilization; in most cases, legume crops received no or very little (5 kg ha^{-1}) N fertilizer, while non-legume crops often received significant fertilizer inputs. When emissions were low in non-legume crops (typically in unfertilized Eastern sites), legume crops produced higher emissions. However, in Western Canada N_2O emissions from legumes were almost always less than from fertilized non-legume crops. This is not in agreement with the findings of Bouwman et al. (2002a) who found in a global analysis that N_2O emissions were 37% higher (non-significant difference) in leguminous crops than other upland crops.

Some N_2O may be released by *Rhizobium* during biological N fixation in root nodules (O'Hara and Daniel 1985) but decomposition of residues following the growing season and in subsequent cropping seasons may be a far more important contributor to increased N_2O emissions. For example, Kilian and Werner (1996) found no difference in N_2O emissions between N_2-fixing and

Table 2. Stepwise regression analysis of N_2O emissions (kg N_2O–N ha^{-1}) from multiple farming systems across several regions in Canada.

	Parameter estimate	Standard error	*F*-value	*p*-value	Partial r^2	r^2
Non-manure soils						
Non-legume systems ($n = 316$)						
Intercept	−0.786	0.16	25.40	< 0.0001		0.47
Precipitation (mm yr^{-1})	0.002	0.00025	68.98	< 0.0001	0.12	
Nitrogen added (kg N ha^{-1})	0.009	0.00089	104.61	< 0.0001	0.35	
Legume systems ($n = 67$)						
Intercept	1.04	0.24	18.91	< 0.0001		0.35
Nitrogen added (kg N ha^{-1})	0.012	0.0021	35.10	< 0.0001	0.35	
Manure soils ($n = 38$)						
Intercept	−4.724	2.06	5.28	0.028		0.27
Precipitation (mm yr^{-1})	0.006	0.0018	11.64	0.0016	0.11	
Nitrogen added (kg N ha^{-1})	0.005	0.0019	7.53	0.0095	0.16	

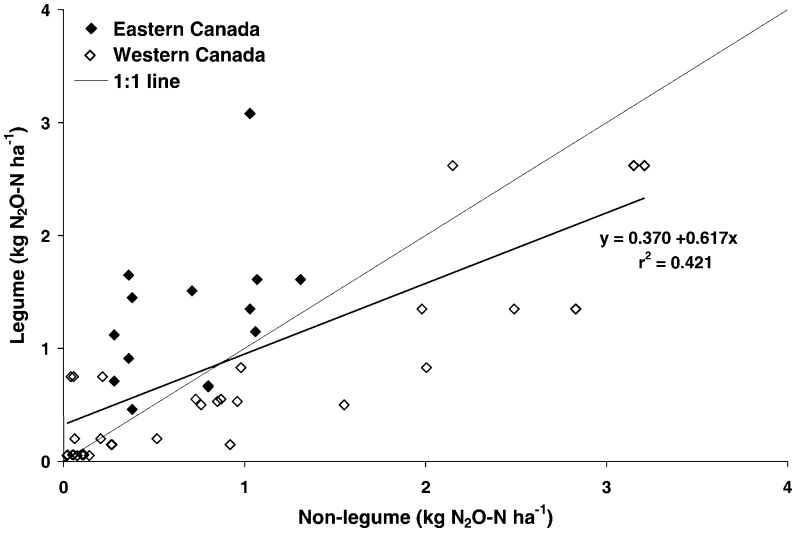

Figure 3. Comparison of N_2O emissions from paired treatments of legume and non-legume crops. Application of N fertilizer was different for legume (0 or 5 kg N ha^{-1} yr^{-1}) compared to non-legume crops (0–190 kg N ha^{-1} yr^{-1}).

non-fixing faba bean varieties during the growing season, but after harvest, higher nitrate levels under the N-fixing variety enhanced N_2O emission. Many studies have demonstrated increased denitrification and/or N_2O emissions as a result of the incorporation of low C:N ratio residues (McKenney et al. 1993; Kaiser et al. 1998; Baggs et al. 2002, 2003). Longer-term effects of legume residue decomposition were observed by Wagner-Riddle and Thurtell (1998) who found elevated N_2O emissions for 2 years after incorporating an alfalfa crop. Similarly, winter season N_2O emissions were greater following N-rich sugar beets than after cereal and oilseed cultivation (Kaiser et al. 1998). Long-term contributions of organic nitrogen from N-rich residues are therefore an important consideration in cropping systems and among different types of legume crops (e.g., annual pulses vs. perennial legume forages). Rather than focusing solely on emissions during legume growth, examining N_2O emissions in all phases of legume systems may provide better understanding of how legumes affect N_2O production.

Manured soils

In a stepwise regression analysis, the amount of N added as manure accounted for 16% of the variability in the measurement data and annual precipitation accounted for an additional 11% (Table 2). A dose response of N_2O emissions to N in manure application has been observed in some instances (Lessard et al. 1996; Chang et al. 1998; Rochette et al. 2000; Whalen 2000). N_2O production following manure application is dependent on many interacting factors (nitrate availability, organic C decomposability and soil moisture) but often there is little correlation between soil physicochemical characteristics (e.g., texture, pH, electrical conductivity) and N_2O production (Lessard et al. 1996; Chang et al. 1998; Whalen et al. 2000).

Rainfall events provoke episodes of prolific N_2O emission in recently manured soils, especially where available N is not limiting (Cates and Keeney 1987; Lessard et al. 1996). Ellis et al. (1998) observed a stimulatory effect of added moisture on N_2O emissions in soils that had received surface-applied pig slurry, but no effect in soils where the slurry was injected. They attributed the response to the physical transport of substrates to the soil microbial biomass and to the creation of anaerobic conditions as the moisture infiltrated the soil. Absence of elevated N_2O flux following recent manure application has been attributed to low soil moisture and water-filled pore space (Chantigny et al. 1998, 2001).

The studies in our data set encompass a wide range of climates, with annual precipitation varying from ca. 350 to 1175 mm yr^{-1} (Table 1). Sehy et al. (2003) and Linn and Doran (1984)

have identified a threshold limit of 60% water filled pore space (WFPS) for elevated N_2O emissions linked to denitrification activity. Reflecting gas diffusivity or aeration as much as moisture availability, WFPS above 60% has been linked to elevated emissions under many conditions (Chantigny et al. 1998; Dobbie and Smith 2001) and is especially significant when soil nitrate is abundant. The coincidence of fertilization and precipitation events was found to be the primary influence on N_2O emissions among multiple sites in Scotland (Dobbie et al. 1999). Our analysis, however, include only annual precipitation values, and will not have captured the specific effects of episodic emissions following rainfall events.

N_2O emissions were not significantly related to soil texture (% clay) in any of these systems, though this factor has been shown to affect emissions elsewhere (Lemke et al. 1998; Bouwman et al. 2002a, b; Kravchenko et al. 2002). For example, Chantigny et al. (2002) measured N_2O production rates that were three times greater in clay soils than in sandy loam soils. The influence of soil texture on the magnitude of N_2O emissions is complex, the most important factors relating to porosity and water content (Dalal et al. 2003).

While including these additional variables in the analyses improved our predictive capacity, less than half of the variability (27–47%) of the N_2O emission data was accounted for. This lack of simple relationships between N_2O emissions and environmental and soil parameters indicates that there are no simple universal driving factors for N_2O production (Beauchamp 1997; Šimek et al. 2004). Bouwman et al. (1993) suggested that up to 60% of soil N_2O production was controlled by five factors (organic matter, soil fertility, soil moisture, temperature and oxygen status). Complex interactions among these factors induces variability in fluxes both temporally and spatially. This inherent variability impedes the practicality of easily predicting N_2O emissions in the field.

Effect of tillage on N_2O emission

Tillage alters both the soil physical environment and the availability of nutrients, including C from crop residues (Campbell et al. 1996; Arshad et al. 1999; Tebrügge and Düring 1999), which can affect N_2O emissions. Tillage influences the amount of soil moisture lost to evaporation and the distribution of soil pore sizes. Similarly, water-stable aggregation in the surface soil may improve under no-tillage resulting in greater organic carbon storage in macroaggregates (Arshad et al. 1999). Residues are placed at different locations as a result of various tillage practices. These differences in moisture status, soil structure and residue placement can affect N_2O production and diffusion in soil and may alter the influences of precipitation and fertilization on N_2O emissions. Some researchers have found marked increases in N_2O emissions under no-till (Staley et al. 1990; Ball et al. 1999; Baggs et al. 2003), while in other studies the effect of tillage on N_2O emissions was less conclusive (Jacinthe and Dick 1997; Lemke et al. 1999; Choudhary et al. 2002; Yamulki and Jarvis 2002; Kaharabata et al. 2003).

The effect of tillage on N_2O emissions from Canadian soils was highly variable. In about half of the observations (56%) emissions from no-till soils were less than those under conventional tillage. The opposite was true in 40% of the comparisons (Table 3), though absolute differences were often quite small. In two cases there was no difference between tillage treatments. This analysis suggests that the effects of tillage may differ among regions across the country. In Eastern Canadian soils (humid climate), soil N_2O emissions under no-till were greater than those from conventional tillage 66% of the time; in Western Canada (semi-arid climate), the converse was often true. In the United States, Six et al. (2004) found that 5 and 10 years after conversion from conventional to no-tillage, N_2O-fluxes had increased and, as in our analysis, the increases were greater in humid climates. However, 20 years after conversion to no-till, N_2O-fluxes had decreased in humid soils (Six et al. 2004). They also observed a decreasing trend in N_2O-fluxes over the 20-year period in both humid and dry climates.

The effect of tillage on N_2O emission also varied among years. In 72% of the instances where emissions were measured for two or more years, the effect of tillage was reversed in subsequent years. A more detailed evaluation of field conditions corresponding to specific emissions measurements may have provided more insight about

Table 3. Influence of tillage practice on N_2O emissions from various cropping systems in Canada.

Location/Meas. period	Precip. (mm y^{-1})	Soil texture	Year	N Applied (kg ha^{-1})	Crop	Absolute difference[a] (kg N_2O–N ha^{-1})	Zero:conv Ratio
Quebec City, PQ	1174	LS	1	60	Barley	0.01	0.99
April–October			2			*0.69*	*1.83*
			3			0.20	0.75
		C	1	60	Barley	*23.62*	*2.15*
			2			*6.0*	*1.98*
			3			*26.76*	*3.20*
Ottawa, ON	879	SL	1	190	Corn	*0.10*	*1.15*
June–October			1	0	Soybean	*0.28*	*1.15*
Woodslee, ON	876	CL	1	182	Corn	1.40	0.84
April–October			2	182		*0.30*	*1.14*
			3	182		1.36	0.61
Swift Current, SK	349	SiL	1	22	Wheat	*0.08*	*4.48*
May–October;			2	22		0.30	0.42
March–April			1	22	Wheat	*0.059*	*2.18*
			2	22		0.021	0.66
			1	43	Wheat	0.056	0.5
			2	43		*0.355*	*2.73*
			1	0	Fallow	0.01	0.29
			1	5	Pulse	*0*	*1.0*
			2	5		*0.55*	*3.75*
Three Hills, AB	490	CL	1	5	Peas	*0.05*	*1.1*
May–October;			2	5		0.30	0.64
March–April			3	5		1.27	0.61
			1	75	Wheat	0.82	0.47
			2	75		1.16	0.42
			3	75		0.17	0.92
			1	75	Wheat	*0.11*	*1.14*
			2	75		0.02	0.97
			3	75		0.38	0.88
			1	0	Fallow	0.31	0.81
			2	0		1.00	0.38
			3	0		3.64	0.47
Breton, AB	550	SiL	1	56	Wheat	0.19	0.49
May–October;			1	720	Wheat	*0.33*	*1.5*
March–April			1	0	Fallow	0.30	0.88
			1	0	Wheat	*0.04*	*1.30*
Ellerslie, AB	450	CL	1	56	Wheat	*0.01*	*1.0*
May–October;			2	56		1.78	0.30
March–April			1	720	Wheat	0.19	0.75
			1	0	Fallow	*0.30*	*1.42*
			2	0		1.81	0.36
			1	0	Wheat	*0.03*	*1.05*
			2	0		2.06	0.15

[a]Absolute difference: the difference in magnitude between zero and conventionally tilled soils (does not indicate a statistically significant difference).

the driving factors behind tillage effects. For example, Choudhary et al. (2002) found strong correlation between WFPS, tillage treatments and N_2O emissions. As soil water content increased under different tillage regimes, N_2O emissions increased also, with greater moisture availability under zero tillage when precipitation events were not frequent. In this way, year-to-year differences in precipitation in combination with conditions of soil nutrient availability and a variety of other factors (e.g., temperature) may influence how tillage affects N_2O emission. Dobbie et al. (1999)

determined that in a variable climate, multiple years of measurement are needed to obtain a robust estimate of mean N_2O fluxes.

Implications

Linear regression of fertilizer N application rate yielded a FIE coefficient that was reasonably close to that of the IPCC, though estimates of 'background' emission (ie., emission from soils receiving no N fertilizer) were lower than other estimates (Bouwman 1996). Elementary stepwise regression analyses to include cropping system and precipitation, as well as N inputs, improved some relationships, but these factors still explained only 27–47% of the variability in the N_2O measurements. Clearly, we need a better fundamental understanding of processes to improve our ability to predict emissions.

Based on our analysis, the use of the IPCC methodology might be improved by adjusting coefficients for broad climate regions in Canada. Though we cannot yet provide definitive coefficients, the best equations for predicting N_2O emissions from humid regions (e.g., Eastern Canada) are likely to be different from those on the semi-arid prairie region.

One puzzling observation arising from our analysis is the occurrence of 'hot spots' – localized areas where N_2O emissions may be several times those from nearby soils under the same management with similar amounts of available N. Others have also observed a similar phenomenon at various scales (Parkin 1987; Christensen et al. 1990; van Kessel et al. 1993; Clemens et al. 1999). These 'hot spots' could account for a large portion of emissions on a landscape, and if we are to produce better emission estimates and find ways of reducing emissions, we will need to better understand the origin of these high emissions. In many cases we do not understand and cannot adequately weight the contributions of nitrification and denitrification responsible for fluxes of N_2O. A first step may be to understand the microbial processes and communities responsible for these emissions; the next step, then, will be to determine how those indigenous microbial communities are affected by different farm management practices.

Though many uncertainties remain, scientists and policy makers are forced to come up with the best available way for estimating N_2O emissions. Simple measurement of N_2O emissions from our soils may provide some insight into the nature of N_2O production. However, to account for the influences of diverse environmental conditions and agronomic practices across the country, the understanding derived from these measurements will need to be merged into models. And further measurements will be needed to test the reliability of these models. One approach, which integrates fluxes over space and time, employs roving tower-based measurement systems to evaluate the effects of management practices on N_2O emissions in diverse agricultural systems (Warland et al. 2001; Grant and Pattey 2003).

At present there is great disparity in the methodology used to evaluate N_2O emissions. Flux measurements are taken using a variety of equipment (flow-through or static chamber vs. tower), over different lengths of time, in different periods during the year. Development of a more common and consistent approach to emissions measurement would greatly enhance our ability to compare regional differences in emissions as a function of system properties and production practices. One obvious improvement would be to insist on annual, year-round measurements wherever possible.

Our analysis indicated that we do not yet have adequate precision to distinguish between potential mitigation practices that result in small differences in emissions (e.g., 10% or less). For example, at many sites, we cannot definitively say how no-till affects N_2O emissions, even though differences among tillage treatments in some years were much greater than 10%. This has crucial implications for mitigation of N_2O emissions from our agroecosystems. If we are unable to measure and verify reductions (or increases) in emissions, how can we make recommendations regarding on-farm management practices? Moreover, without better understanding of the processes that govern N_2O emissions in the field and the way that these processes interact, will more field measurements yield the understanding we need to predict emissions?

A question surfaces from this analysis: if we had twice as many measurements, would our ability to estimate and predict N_2O emissions be much better? Though we cannot be sure, our findings suggest that some of our research efforts might better be expended in improving our understand-

ing of biological processes rather than compiling more measurements. Certainly, a closer examination of processes in conjunction with on-going measurements is required.

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