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Tillage does not increase nitrous oxide emissions under dryland canola (*Brassica napus* L.) in a semiarid environment of south-eastern Australia

Guangdi D. Li[A,C,D], *Mark K. Conyers*[A,C], *Graeme D. Schwenke*[B], *Richard C. Hayes*[A,C], *De Li Liu*[A,C], *Adam J. Lowrie*[A], *Graeme J. Poile*[A], *Albert A. Oates*[A], *and Richard J. Lowrie*[A]

[A]NSW Department of Primary Industries, Wagga Wagga Agricultural Institute, Pine Gully Road, Wagga Wagga, NSW 2650, Australia.
[B]NSW Department of Primary Industries, Tamworth Agricultural Institute, 4 Marsden Park Road, Tamworth, NSW 2340, Australia.
[C]Graham Centre for Agricultural Innovation, Albert Pugsley Place, Wagga Wagga, NSW 2650, Australia.
[D]Corresponding author. Email: guangdi.li@dpi.nsw.gov.au

Abstract. Dryland cereal production systems of south-eastern Australia require viable options for reducing nitrous oxide (N_2O) emissions without compromising productivity and profitability. A 4-year rotational experiment with wheat (*Triticum aestivum* L.)–canola (*Brassica napus* L.)–grain legumes–wheat in sequence was established at Wagga Wagga, NSW, Australia, in a semiarid Mediterranean-type environment where long-term average annual rainfall is 541 mm and the incidence of summer rainfall is episodic and unreliable. The objectives of the experiment were to investigate whether (i) tillage increases N_2O emissions and (ii) nitrogen (N) application can improve productivity without increasing N_2O emissions. The base experimental design for each crop phase was a split-plot design with tillage treatment (tilled versus no-till) as the whole plot, and N fertiliser rate (0, 25, 50 and 100 kg N/ha) as the subplot, replicated three times. This paper reports high resolution N_2O emission data under a canola crop. The daily N_2O emission rate averaged 0.55 g N_2O-N/ha.day, ranging between –0.81 and 6.71 g N_2O-N/ha.day. The annual cumulative N_2O-N emitted was 175.6 and 224.3 g N_2O-N/ha under 0 and 100 kg N/ha treatments respectively. There was no evidence to support the first hypothesis that tillage increases N_2O emissions, a result which may give farmers more confidence to use tillage strategically to manage weeds and diseases where necessary. However, increasing N fertiliser rate tended to increase N_2O emissions, but did not increase crop production at this site.

Additional keywords: conservation tillage, greenhouse gases, nitrogen fertiliser, trace gas emission.

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Introduction

In Australia, the agriculture sector is the dominant source of anthropogenic nitrous oxide (N_2O), accounting for 78.6% of the net national N_2O emissions (Commonwealth of Australia 2014). Globally, 70% of annual anthropogenic N_2O emissions come from animal and crop production (Mosier 2001). There is a clear need for primary producers to implement methods aimed at reducing overall N_2O emissions. Research has attributed the majority of N_2O emitted from soil to the following soil microbial processes: (i) nitrification, in which soil bacteria convert ammonium into nitrate (NO_3^-); (ii) denitrification, in which soil microbes consume soil NO_3^- under anaerobic conditions; (iii) assimilatory NO_3^- reduction, in which NO_3^- is converted to nitrite; and (iv) dissimilatory NO_3^- reduction, in which NO_3^- is converted to ammonium (Mosier *et al*. 1998; Dalal *et al*. 2003; Harris *et al*. 2013). Therefore, activities that affect these soil microbial processes could have a significant impact on N_2O emissions (Dalal *et al*. 2003).

No-till farming has been widely adopted for dryland grain production in Australia due to benefits including improved soil structure with better water infiltration and water retention, increased ground cover with less soil erosion and increased soil fertility with more soil organic carbon sequestered (Chan and Heenan 2005; Hobbs 2007; Thomas *et al*. 2007). In addition, the adoption of no-till farming reduces energy, machinery and labour inputs (Scott *et al*. 2013), hence increasing on-farm profitability (Williams *et al*. 1990). A survey conducted across Australia in 2008 showed that nearly 90% of growers were using no-till technology on their farms (Llewellyn and D'Emden 2010). Nationally, 74% of the 27.8 million ha used for crop and pasture production was sown using no-till methods (ABS 2015). Conversely, no-till farming has also contributed to increased incidences of soil- and stubble-borne diseases, and development of more herbicide-resistant weeds (Chauhan *et al*. 2006). Tillage may be both beneficial and detrimental to many crop diseases, insects, pests and weeds (M. K. Conyers, unpubl.

data). The challenge is to find a balance between the negative impacts of tillage on soil structure by tillage and the maintenance of optimum crop agricultural production. In recent years, several field experiments were conducted across northern and southern grain regions in Australia to investigate how much damage is done to soil by occasional tillage, strategically applied, in an otherwise no-till system (Crawford *et al*. 2015; M. K. Conyers, unpubl. data). It was concluded that strategically timed tillage could be a viable management option to manage diseases and herbicide-resistant weeds in the short-term, but the long-term effect on soil chemical and physical properties requires further investigation. However, there is no published research on the effect of tillage on N_2O emission in dryland cropping systems in south-eastern Australia.

Tillage increases the aeration of the topsoil and mixes the crop residues with the soil, typically enhancing soil microbial activity, and therefore potentially increasing N_2O emissions (Chatskikh and Olesen 2007). However, many studies have indicated increases in N_2O emissions under no-till treatments (Weier *et al*. 1998; Ball *et al*. 1999; Skiba *et al*. 2002). Chatskikh and Olesen (2007) suggested that the greater N_2O emissions under no-till treatments may be due to reduced gas diffusivity and air-filled porosity, often caused by high soil moisture, which favours the activity of denitrifying bacteria. Helgason *et al*. (2005) analysed N_2O emissions in Canada using over 400 datasets from a range of farming management regimes, including tillage practices, and found that no-till treatment increased N_2O emissions in humid climates but decreased it in semiarid climates. In the USA, Six *et al*. (2004) found that the conversion from conventional to no-till practice had a strong time dependency on N_2O emissions in both humid and dry climates, demonstrating that greenhouse gas mitigation by adoption of no-till technology is much more variable and complex than previously considered. Therefore, there is large uncertainty associated with our current understanding of the influence of tillage practice on N_2O emissions.

Increasing application rates of nitrogen (N) fertiliser has been previously shown to increase N_2O emissions in broadacre cropping systems across a range of environments (Dalal *et al*. 2003; Harris *et al*. 2013; Schwenke and Haigh 2015), attributable to higher availability of N vulnerable to gaseous loss and to an imbalance between N availability in the soil and demand by the crop. Tillage is known to elevate levels of plant-available soil N due to mineralisation of soil organic matter. In situations where tillage is used in conjunction with high application rates of fertiliser N, N_2O emissions might be expected to increase further. The objective of this study was to use continuous high resolution N_2O emission data collected over 365 days under dryland canola (*Brassica napus* L.) in a dryland semiarid environment to test (i) whether tillage increases N_2O emission and (ii) whether N application improves productivity without increasing N_2O emission.

Materials and methods

Site description

The field experiment was conducted at the Wagga Wagga Agricultural Institute, Wagga Wagga, NSW (35°01′45″S, 147°20′36″E; 210 m a.s.l.). The soil was a Red Kandosol (Isbell 1996). The baseline soil chemical analysis showed that the site was slightly acidic with a pH of 5.1 in calcium chloride and available phosphorus (P) was 36.1 mg/kg (Colwell P) in the surface 0–0.1 m depth (Table 1). Long-term average rainfall at the site is 541 mm and is relatively evenly distributed throughout the year. Prior to the establishment of the experiment, the site was cropped for at least 5 years with the previous two crops being barley.

Experimental design

A 4-year rotation experiment was conducted during 2012–15 with wheat (*Triticum aestivum* L.)–canola–grain legumes–wheat in sequence. The site was divided into four quadrants with the experimental measurements taken from a different quadrant in each year of measurement. This paper reports the results from a canola crop grown in 2013. A fully automated 12-chamber

Table 1. Baseline soil chemical and physical properties at the experimental site in autumn 2012
–, not measured; ECEC, Effective cation exchange capacity

Soil depth (m)	0–0.1	0.1–0.2	0.2–0.4	0.4–0.6	0.6–0.9	0.9–1.2
pH in $CaCl_2$	5.1	4.9	5.7	6.1	6.2	6.2
Soil total N (%)	0.15	0.06	0.05	0.05	–	–
Soil total C (%)	1.64	0.67	0.46	0.36	–	–
Colwell P (mg/kg)	36.1	8.0	5.2	–	–	–
Bulk density (g/cm^3)	1.41	1.49	1.43	1.35	1.49	1.55
Clay (%)	26.6	40.1	54.3	68.0	67.9	–
Silt (%)	8.8	6.2	2.5	1.2	1.9	–
Sand (%)	64.6	53.7	43.2	30.8	30.2	–
			Exchangeable cations (cmol(+)/kg)			
Al	0.04	0.07	0.01	0.00	0.00	0.00
Ca	6.10	5.27	6.30	6.69	6.20	7.25
K	1.29	0.99	0.93	0.78	0.85	1.08
Na	0.01	0.01	0.03	0.07	0.14	0.27
Mn	0.09	0.06	0.01	0.01	0.00	0.01
Mg	0.68	1.48	3.01	4.56	5.65	7.58
Total ECEC	8.21	7.88	10.29	12.11	12.84	16.19

system was installed and was fully functioning before the crop was sown.

The experiment was a randomised split-plot design with tillage (tilled vs no-till) as the whole plot and N application rates (0, 25, 50 and 100 kg N/ha) as the subplot, replicated three times. Plot size was 5 m × 9 m. The automated chambers were located on 12 plots with two treatment contrasts (tilled vs no-till and 0 vs 100 kg N/ha), replicated three times.

Hyola555 TT canola was sown at 4 kg/ha on 20 May 2013, using an air-seeder with knife points fitted to the front of the tynes, spaced 0.25 m apart. The tilled plots were cultivated at 0.1 m depth with a scarifier in both directions then harrowed twice one day before sowing. At sowing, all plots received 15 kg P/ha as single superphosphate (8.8% P and 11% sulfur) mixed with 5 kg N/ha as urea (46% N) as a base fertiliser. The N treatments were imposed by top-dressing urea at the designed rates before stem elongation on 2 August 2013, 74 days after sowing.

Gas flux measurement

The auto-chamber gas chromatograph (GC) system was installed on the site in March 2013 and tested for 4 weeks before the reporting period (15 April 2013 to 14 April 2014). The system consisted of 12 pneumatically operated static chambers linked to an automated sampling system, an *in situ* GC (SRI GC8610, Torrance, CA, USA) and an LI-820 infrared gas analyser (LI-COR, Lincoln, NE, USA) as described by Rowlings *et al*. (2012). The clear acrylic glass chambers (0.5 m × 0.5 m) with a height of 0.15 m were secured to stainless steel bases inserted permanently into the soil to a depth of 0.1 m. Each chamber covered two crop rows. The chamber height was extended to 0.65 m when crop height exceeded 0.15 m. When the crop height exceeded 0.65 m, the plants in the chambers were periodically trimmed above 0.65 m until the crop was harvested. A sampling manifold with three outlets at low, middle and high positions was fitted in each chamber to ensure the system took representative gas samples when the extended chambers were used. Two bases were installed in each plot to enable the chamber to be swapped between two bases every 1–2 weeks during the growing season and every 3–4 weeks at other times. This was done to minimise the glasshouse effect on plant growth and changes on soil moisture in chambers. A thermocouple sensor was fitted inside one chamber in each replicate to monitor chamber air temperature and a theta probe (Theta-probe MK2K, Delta-T Devices Ltd, Burwell, England) was inserted in each plot to measure soil moisture at 0–0.1 m (every 5 min). Water-filled pore space (WFPS) was calculated as described by Linn and Doran (1984). During the measurement cycle, the chambers were programmed to open automatically during high intensity rainfall events (>0.4 mm/5 min) or when air temperature inside the chambers exceeded 45°C during the growing season and 60°C at other times.

A full measurement cycle (of the four chambers per replicate) for greenhouse gas flux determination commenced with lid closure, and finished when the lids were opened 60 min later. During this time, each chamber was sequentially sampled for 3 min followed by a certified calibration standard: 500 ppb N_2O, 4.0 ppm methane (CH_4) and 800 ppm carbon dioxide (CO_2) (Coregas Pty Ltd Australia). Samples passed through the 3-mL sample loop of two separate eight-port valves before injection into the respective gas detectors: electron capture detector for N_2O and flame ionisation detector for CH_4. The LI-820 was connected to the waste vent of the valve and logged the CO_2 concentration every second as described by Rowlings *et al*. (2012). This process was repeated at 15-min intervals, sampling each chamber four times during the 60-min closure period. The lid then opened for 120 min while the chambers in the other two replicates were sampled. This 3-h cycle allowed eight flux measurements for each chamber to be obtained per 24-h period. The detection limit of the system was approximately –0.43 μg N_2O-N/m^2.h, 0.26 μg CH_4-C/m^2.h and 0.48 mg CO_2-C/m^2.h with 0.15-m head space. Sample dilution via leakage was considered negligible.

N_2O emission calculation

Hourly N_2O fluxes were calculated from the slope of the linear regression of gas concentration vs measurement time during the chamber lid closure, corrected for air temperature, atmospheric pressure and the ratio of chamber volume to surface area as described by Schwenke and Haigh (2016). The raw data were processed using an Auto GHG System Flux Calculator (Flux.net3.3) developed by the Queensland University of Technology (David Rowlings, pers. comm.). The Pearson's correlation coefficient (R^2) for the linear regression was calculated and used as a quality check for each regression. Flux rates were set to zero if the regression coefficient was <0.8 as per Barton *et al*. (2011). Daily N_2O emission for each chamber was calculated by averaging the eight emission measurements for that day. Annual cumulative N_2O emission was then calculated by integrating daily N_2O fluxes over 365 days. The emission factor (EF, percentage of applied N emitted) with background emission correction (Type I) was calculated as per Barton *et al*. (2008).

Agronomic measurements

Seedling establishment was assessed 8 weeks after sowing by counting plants in 1 m of crop row at four random locations in each plot, expressed as plants/m^2. Crop aboveground dry matter (DM) was measured at anthesis and at harvest by cutting two adjacent 1-m crop rows at two locations in each plot, and weighing after drying at 70°C for 72 h. The harvest samples were threshed to separate grain from straw and chaff to calculate grain yield and harvest index. The grain samples were sent to the laboratory for quality measurements, including oil content, crude protein, glucosinolates, test weight and 1000-grain weight, as described by AOF (2014).

Soil mineral N measurement

Deep soil cores were taken pre-sowing (April 2013) and post-harvest (December 2013) using a stainless steel tube (44 mm in diameter) inserted to 1.0 m at four locations in each plot. Each core was segmented into 0.1-m increments over the 0–0.40 m depth and 0.2-m increments thereafter. Samples from each plot were bulked for each depth. Surface soil (0–0.1 m) samples (10 cores per plot, bulked) were taken monthly using a foot corer (25-mm diameter) except for November 2013 and January and March 2014 when soil was too dry (i.e. hard) to insert the coring tube. Soil mineral N concentration (NO_3^--N and ammonium-N) was analysed in all samples as described by

Raymont and Higginson (1992). Soil mineral N was calculated using bulk density estimated from the deep soil coring for each depth at pre-sowing.

N recovery measurement

A micro-plot (1.25 m × 1.0 m, containing five rows of canola) was randomly selected from each plot using a metal frame with 0.1 m inserted into soil and 0.1 m above ground. Each micro-plot received 10% atom enriched urea-^{15}N (prepared from 98 atom% ^{15}N, SerCon, Australia) at one of four N rates (0, 25, 50 and 100 kg N/ha) in both tilled and no-till treatments. The ^{15}N-labelled urea was dissolved in 2000 mL of distilled water and sprayed evenly over the whole area of the micro-plot. The container was then rinsed with 500 mL of distilled water which was also sprayed onto the micro-plot. The total water applied to each micro-plot was equivalent to 2 mm of rainfall.

At crop maturity, aboveground crop biomass, including fallen leaves, was collected from 0.8 m of the middle three crop rows of each micro-plot. The plants in the two outside rows and at 0.1 m each end of the micro-plot were excluded to ensure the plant sample collected had received a uniform rate of ^{15}N. Plant samples were dried at 70°C, and passed through a 2-mm sieve, then finely ground in a puck and ring grinder. A sub-sample (~3.0–3.5 mg) was capsuled into a 1.0 mm × 0.8 mm tin container for total N and ^{15}N isotope analysis (Sercon 20–22 IRMS, UK).

Deep soil samples were taken from all micro-plots after the crop was harvested as described above. The soil samples, including root materials, from 0–0.10, 0.10–0.20 and 0.20–0.3 m, were finely ground through a puck and ring grinder then capsuled for ^{15}N isotope analysis using the same equipment as described for the plant samples.

The N recovery (%) in plant and soil was calculated as follows:

$$\text{N Recovery} = \frac{P \times (c - b)}{f(a - b)} \times 100\%$$

where P is total N in the plant (aboveground biomass) or soil (including roots) in kg/ha, a is ^{15}N atom% in the labelled fertiliser, b is ^{15}N atom% in the plant part or soil receiving no ^{15}N, c is ^{15}N atom% in the plant part or soil receiving ^{15}N and f is the rate of ^{15}N fertiliser applied.

Statistical analysis

Analysis of variance for seedling density, crop biomass, grain yield, grain quality data and soil data at each depth was performed using a split-plot model in Genstat Release 17.1 (Payne *et al*. 2014) with tillage as the main plot and N rate as the subplot. Where tillage × N-rate interactions were not significant at $P = 0.05$, means of main treatment effects are presented and least significant differences (l.s.d.) between treatments presented where appropriate. A multiple linear regression analysis was performed for factors potentially affecting N_2O-N emission. The daily N_2O-N emission rates over 365 days were spline-fitted using a linear mixed model in ASReml-R (Gilmour *et al*. 2009). The fixed effects were tillage, N application rate, the linear component of sampling time and their interactions. Random effects were replicates, the spline component of sampling time and associated interactions. All terms were included in the model initially, but terms that failed to achieve statistical significance ($P < 0.05$) were excluded from the final model. The fixed effects were tested using the Wald statistical test and the random effects were tested using the residual maximum likelihood ratio test when necessary.

Results

Climate and soil moisture

The site received 446 mm of rainfall during the experimental period (15 April 2013 to 14 April 2014), which was 82% of the long-term annual average rainfall (541 mm). There was very little rainfall during late growth in October (16.0 mm) and November (8.2 mm). On five occasions, daily rainfall exceeded 10 mm during December–March (Fig. 1). The minimum average air and soil temperature was 4.2°C and 5.9°C respectively, in July, and the corresponding maxima were 34.6°C and 40.0°C in January. The air and soil temperatures were similar during the growing season, but soil temperatures outside the growing season were much higher than the air temperature (Fig. 1). The soil WFPS in the 0–0.1 m depth closely reflected rainfall patterns. During the winter months (June–August), WFPS exceeded 80% on several occasions when the site received >15 mm of rainfall. Compared with the no-till treatment, WFPS was slightly and non-significantly less in the tilled treatment for the first two months after the tillage and sowing operations. Overall, there was no significant difference in WFPS between tilled and no-till treatments (Fig. 1).

Crop agronomy and grain quality

There were no significant differences in any agronomy or grain quality parameters measured between tilled and no-till treatments, neither for interaction of tillage and N rate treatments ($P > 0.05$). However, there was a significant difference in grain yield between N application rates ($P < 0.01$), with the highest grain yield recorded at the 50 kg N/ha rate. There were also significant differences in oil content ($P < 0.05$), crude protein ($P < 0.001$) and glucosinolates ($P < 0.01$) between N rate treatments. The canola had the highest crude protein and the lowest oil content in the 100 kg N/ha treatment, but the lowest crude protein and the highest oil content in the nil-N treatment (Table 2). The N application rate had no significant effect on seedling density at establishment, DM at anthesis or DM at harvest.

Soil mineral N and N recovery

The soil contained 52 kg/ha mineral N in the 0–0.1 m soil depth before sowing in April 2013. As the season progressed, soil mineral N decreased to 14 kg/ha by July 2013 (Fig. 2). There were significant differences in soil mineral N between N rate treatments after fertiliser N was top-dressed in early August (Fig. 2), but soil mineral N subsequently declined in all treatments during August–October as crop growth progressed. From October onwards, soil mineral N increased greatly (Fig. 2), indicating that mineralisation of N in soil exceeded plant uptake as the soil dried and the crop matured. Soil mineral

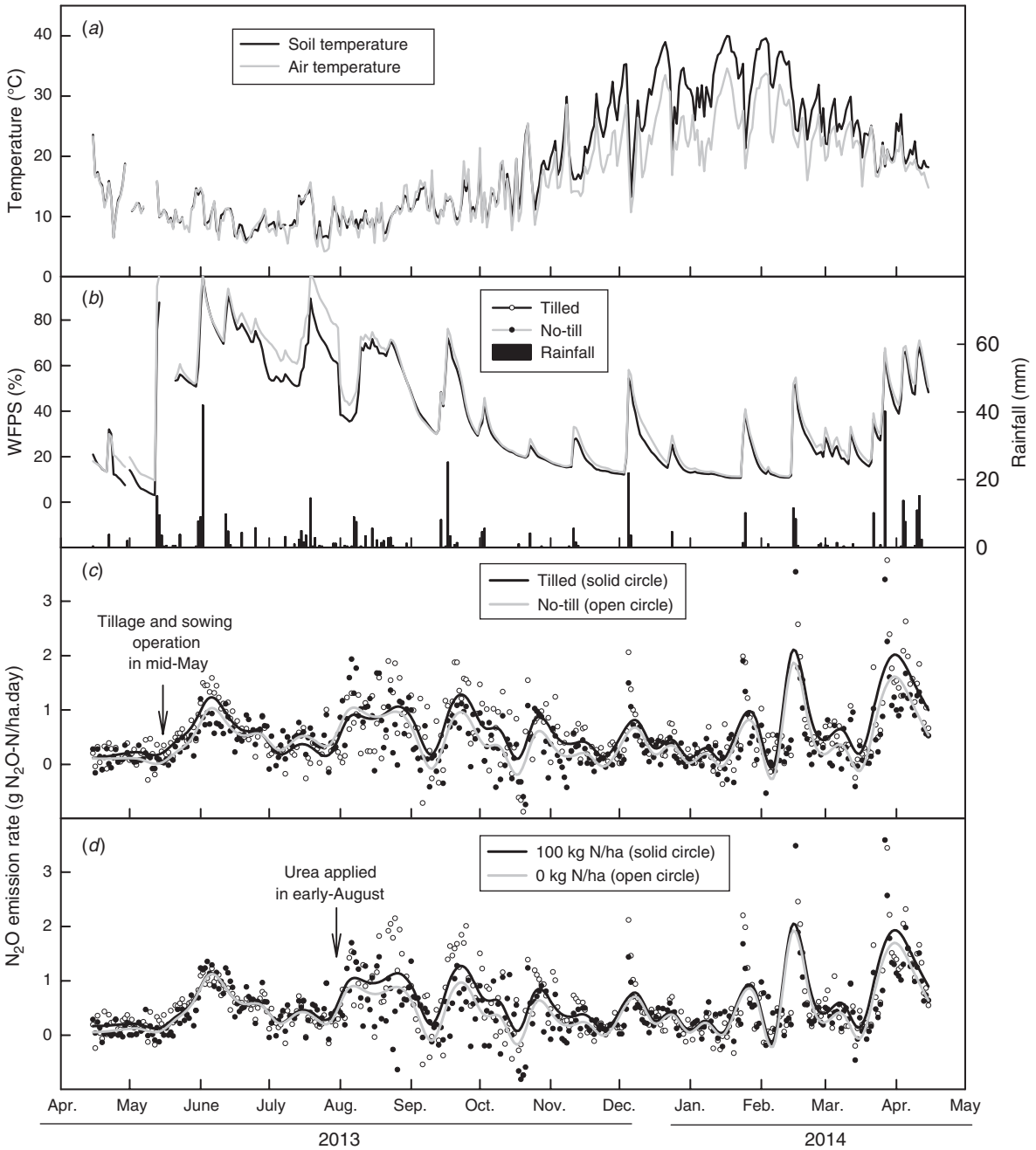

Fig. 1. (*a*) Air and soil temperatures (°C); (*b*) rainfall (mm) and water-filled pore space (WFPS, %) under tilled and no-till treatments; (*c*) daily N_2O emission rate (g N_2O-N/ha.day) under tilled and no-till treatments; (*d*) daily N_2O emission rate (g N_2O-N/ha.day) under 0 and 100 kg N/ha treatments. Both dashed and solid lines on (*c*) and (*d*) are fitted splines for respective treatments.

N peaked in February 2014 and then decreased to 33 kg/ha with no treatment difference by April 2014 (Fig. 2).

Tillage had no significant effect on soil mineral N in any depths in the soil profile (0–1.0 m) at any sampling time (data not shown). However, there were significant differences in soil mineral N within 0–0.4 m due to different N rates between the pre-sowing (April 2013) and post-harvest (December 2013) sampling dates. Below 0.4 m, there was no difference in soil mineral N between the two sampling times (Fig. 3). The soil had 83 kg/ha soil mineral N in the 0–1.0 m profile before sowing with no difference between tillage or N application treatments. After harvest, the soil had only 37 kg/ha soil mineral N in the whole profile with the highest mineral N in the 100 kg N/ha treatment (Fig. 3).

There was no significant difference in ^{15}N recovery percentage between N rate either by plant or soil in 0–0.3 m (Table 2). The majority of N retained in soil (96%) was found in the 0–0.1 m depth, with <1% of N recovered in the

Table 2. Canola agronomic performance and grain quality parameters, and ^{15}N recovery by plant at crop maturity and by soil at 0–0.1, 0.1–0.2 and 0.2–0.3 m under different N rates

DM, Dry matter; *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$; n.s., not significant; –, not applicable

N application rate	0 kg/ha	25 kg/ha	50 kg/ha	100 kg/ha	Mean	Significance	l.s.d.$_{0.05}$
Agronomic performance							
Seedling density (plants/m^2)	44.5	42.6	52.4	51.9	47.9	n.s.	–
DM at anthesis (kg/ha)	4488	4785	4873	5030	4794	n.s.	–
DM at harvest (kg/ha)	7105	7409	7492	7870	7469	n.s.	–
Grain yield (kg/ha)	1584	1714	1858	1753	1727	**	122.2
Harvest index (%)	25.4	26	26.2	26.9	26.1	n.s.	–
Grain quality							
Oil content (%)	43.7	42.9	42.5	41.4	42.6	*	1.24
Crude protein (%)	37.1	38.8	40.0	41.8	39.4	***	0.92
Glucosinolates (%)	8.5	7.8	7.8	8.5	8.2	n.s.	–
Test weight (kg/100 L)	52.1	52.7	52.8	53.2	52.7	n.s.	–
1000-grain weight (g)	7.0	7.3	7.7	8.9	7.7	n.s.	–
^{15}N recovery (% of applied)							
N recovered by plant (%)	–	27.6	28.5	31.0	29.0	n.s.	–
N recovered by soil (%)							
0–0.1 m	–	27.0	24.8	21.8	24.5	n.s.	–
0.1–0.2 m	–	0.7	0.6	0.7	0.7	n.s.	–
0.2–0.3 m	–	0.0	0.3	0.3	0.2	n.s.	–
Total N recovery (%)	–	55.4	54.2	53.8	54.5	n.s.	–
Unaccounted N loss	–	44.6	45.8	46.2	45.5	n.s.	–

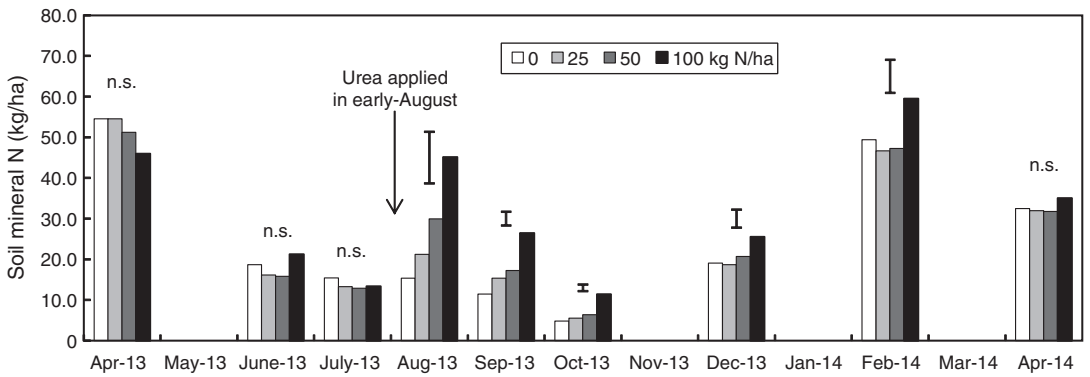

Fig. 2. Monthly soil mineral N (kg/ha) at 0–0.1 m under different N fertiliser application rates. Vertical bars present l.s.d. at $P=0.05$; n.s., not significant.

0.2–0.3 m depth, indicating negligible leaching of ^{15}N beyond 0.3 m. The total N recovery in plant and soil was 54% across N rates, with ~46% of N unaccounted (Table 2), presumably lost to the atmosphere as gases, such as ammonia (NH_3), N_2 and N_2O.

Seasonal N_2O emission

There was a distinct seasonal variation in N_2O emission rates which followed the rainfall pattern, although temperature change may have also contributed. Significant rainfall events stimulated N_2O emissions, particularly summer storms of >10 mm (Fig. 1). Immediately after tillage and sowing operations (tillage on the tilled treatments was carried out one day before sowing), N_2O emission from both tilled and no-till treatments increased, coinciding with two significant rainfall events in late May and June (Fig. 1). The N_2O emissions were slightly higher in the tilled treatment during winter, late spring, early summer and early in the following autumn compared with no-till treatment. However, there was no overall significant difference in daily N_2O emitted between tilled and no-till treatments ($P>0.05$, Table 3).

The N_2O emission rate in the 100 kg N/ha treatment was higher than in the nil-N treatment after N fertiliser was top-dressed and remained higher throughout the growing season (Fig. 1). Over the year of measurement, there was a significant difference in daily N_2O emission rate between 0 and 100 kg N/ha rate treatments ($P<0.001$, Table 3). There was also a significant interaction between tillage treatments and the linear component of time trend ($P<0.01$), but no difference between N rate and the linear component of time trend ($P>0.05$), indicating that the slopes of time trends significantly differed between tilled and no-till treatments, but were similar between N rates in terms of daily N_2O emission rate (Table 3).

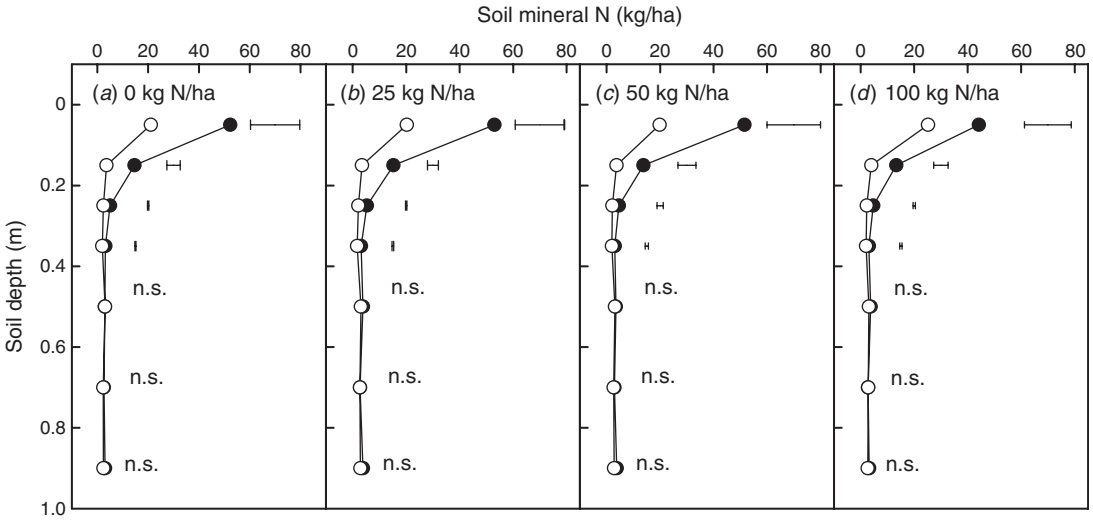

Fig. 3. Soil mineral N (kg/ha) in the soil profile under different N fertiliser application rates at sowing (●) in April 2013 and at harvest (○) in November 2013. Horizontal bars present l.s.d. at $P = 0.05$ between two sampling dates; n.s., not significant.

Table 3. Wald statistics for main effects and their interactions for daily N_2O emission rate over 365 days

F, Fixed effect; R, random effect; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$; n.s., not significant

Strata/decomposition	Effect	N_2O emission rate
Experimental units		
Tillage	F	2.01 n.s.
Residual	R	
N rate	F	26.44***
Tillage × N rate	F	3.41 n.s.
Residual	R	
Experimental units × time		
Linear (time)	F	65.43***
Spline (time)	R	***
Tillage × linear (time)	F	10.19**
N rate × linear (time)	F	1.29 n.s.
Tillage × N rate × linear (time)	F	15.13***
Tillage × spline (time)	R	**
N rate × spline (time)	R	*
Residual	R	

Cumulative N_2O emitted and EF

There was no significant difference in cumulative N_2O-N emitted between tilled and no-till treatments. However, the cumulative N_2O-N emitted under the 100 kg N/ha treatment (224.3 g N_2O-N/ha.year) was higher than that under the nil-N treatment (175.6 g N_2O-N/ha.year) at $P = 0.062$ (Table 4). The average daily N_2O emission rate was 0.48 and 0.61 g N_2O-N/ha.day under 0 and 100 kg N/ha treatments respectively, ranging between –0.81 and 6.71 g N_2O-N/ha.day (Table 4). The EF, corrected with background emission, was 0.05% averaged across tilled and no-till treatments (Table 4).

Relationship between daily N_2O-N emission rate and environmental factors

Multiple linear regression analysis showed that daily N_2O emission rate was closely related to air temperature ($P < 0.01$) and WFPS ($P < 0.001$), with N_2O-N emitted = 0.0273 air temperature + 0.0156 WFPS – 0.499 (Fig. 4). The greatest daily N_2O emission rates (3.8–7.8 g N_2O-N/ha.day) occurred during February and March 2014 when average air temperature was 19–24°C and WFPS was 51–67% (Figs 1 and 4). During the growing season of May–November, N_2O-N was <3.0 g N_2O-N/ha.day with air temperature <20°C, while WFPS varied within 40–100% depending on rainfall intensity and frequency (Figs 1 and 4). No relationship was found between N_2O emission rate and soil mineral N at 0–0.1 m based on data from monthly soil sampling ($P > 0.05$, data not shown).

Discussion

The daily N_2O emission rate was relatively low, ranging from –0.81 to 6.71 g N_2O-N/ha.day. Total N_2O emitted over the 365-day experiment was 176 and 224 g N_2O-N/ha from 0 and 100 kg N/ha treatments respectively, under a canola crop in dryland conditions in southern Australia (Table 4). These losses are comparable to those recorded for rain-fed wheat from a strongly acid soil in a temperate region in south-eastern Australia, where emissions were in the range of 0–12.5 g N_2O-N/ha.day and the annual losses were 200–270 g N_2O-N/ha (Barker-Reid *et al.* 2005), but nearly double those from a canola crop grown in a sandy soil in a semiarid Mediterranean environment in Western Australia (Barton *et al.* 2010) where annual N_2O emission was 80 and 128 g N_2O-N/ha from non-N- and N-fertilised canola respectively. In contrast, the average daily N_2O emission rate measured at the current site (0.48–0.61 g N_2O-N/ha.day) was only a fraction of that measured under a canola crop on raised and flat seedbeds in a high-rainfall environment in south-west Victoria (Harris *et al.* 2013) where the average daily N_2O emission rate was up to 215 g N_2O-N/ha during July–September 2011. Harris *et al.* (2013) concluded that denitrification was the main process leading to high N_2O emissions during the wet period in the high-rainfall cropping system. No obvious periods of waterlogging were observed at the current site, reflecting the

Table 4. Summary of N_2O emission rate and emission factor over 365 days under a canola crop with 0 and 100 kg N/ha applied

n.s., not significant; –, not applicable

N application rate (kg N/ha)	Emission range (g N_2O-N/ha.day)	Average emission rate (g N_2O-N/ha.day)	Cumulative emission (g N_2O-N/ha)	Emission factor (%)
0	–0.81 to 4.63	0.48	175.6	
100	–0.74 to 6.71	0.61	224.3	0.05
		ANOVA (split-plot model)		
Tillage effect	–	–	n.s.	–
N rate effect	–	–	$P = 0.062$	–
Tillage × N rate	–	–	n.s.	–

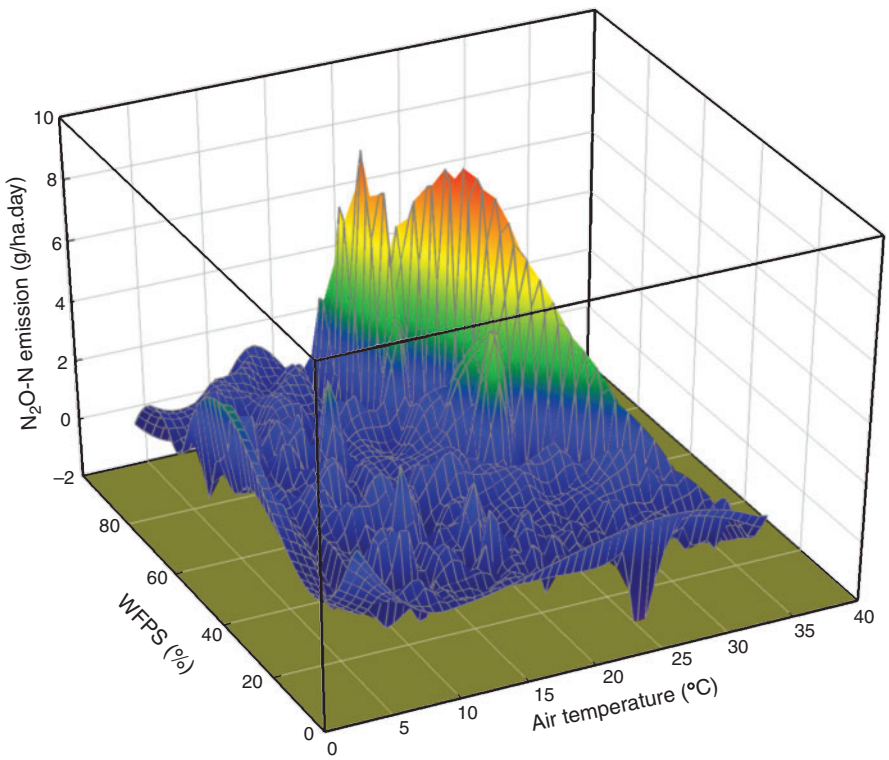

Fig. 4. Relationship of daily N_2O emission rate (g N_2O-N/ha.day) with air temperature (°C) and soil water-filled pore space (WFPS, %). N_2O-N emitted = 0.0273 air temperature + 0.0156 WFPS – 0.499.

low rainfall and a lighter textured soil than that of Harris *et al.* (2013). The EF (0.05%, Table 4) at the current site was nearly identical to that measured from a canola crop grown on a sandy soil (0.06%) in Western Australia (Barton *et al.* 2010), but only one-quarter of the current EF for all non-irrigated N-fertilised crop in Australia (0.2%, ANGA 2015). As a comparison, the EF of 0.15% obtained by Officer *et al.* (2015) on an alkaline Vertosol is consistent with the latter (0.2%, ANGA 2015). We suggest that soil properties such as texture and pH need to be considered when estimating the potential EFs that might be applied in models.

In the current study, the site received 446 mm of rainfall during the experimental period, which was a decile-4 year, slightly drier than the long-term average (541 mm). Although WFPS was over 70% for short periods after significant rainfall events in winter (Fig. 1), soil moisture was not high enough to initiate much denitrification. In addition, active plant growth during the growing season utilised the majority of plant-available soil mineral N and kept the soil mineral N status low (Fig. 2), which reduced the chance for N_2O emission from denitrification. Outside of the growing season, when the air temperature was high, significant rainfall events (>10 mm) stimulated soil microbial activity and resulted in brief spikes in N_2O emission (Fig. 1). The highest N_2O daily emission rate for the year of measurement was in summer at an average air temperature of 19–24°C and WFPS of 51–67% (Fig. 4). Therefore, out-of-season N_2O emissions over the summer fallow period appear to require more attention in this dryland cropping environment, which is consistent with the observations of Barton *et al.* (2008). The current findings highlight the need

to monitor N_2O emissions from rotations which include grain and pasture legume break-crops in this environment, as carry-over of N from one growing season to the next has been previously shown to be substantial (Evans *et al*. 2006), potentially elevating gaseous losses of N_2O, particularly during wet summers.

The inclusion of perennial species into the cropping systems (Robertson and Revell 2014) is one option that could reduce the risk of N_2O emission in this environment. Perennial pasture, with its deep root system, responds quickly to out-of-season rainfall and can use excess water and NO_3^- efficiently (Heng *et al*. 2001; Ridley *et al*. 2001) rather than accumulating in a post-crop fallow period. This also reduces the risk of on-site soil acidification (Scott *et al*. 2000; Dear *et al*. 2009) and off-site soil salinity (Dear and Ewing 2008). An even more progressive option may be to develop perennial-based cropping systems where grain is harvested directly from the perennial crop rather than growing sequences of annual crops in phased rotations with perennial pasture species. Early generation perennial crop germplasm currently exists which, although not yet commercially deployable, holds promise for future cropping systems to further improve resource use efficiencies and reduce losses such as N_2O emissions (Hayes *et al*. 2012; Crews and Dehaan 2015). Further research is warranted to progress perennial plant technologies to reduce N_2O emissions from cropping soils.

No-till technology has been widely adopted in dryland farming systems in south-eastern Australia (Llewellyn and D'Emden 2010; ABS 2015). However, with increased pressure from soil- and stubble-borne diseases and the development of herbicide-resistant weeds under a strict no-till farming system, farmers are tending to use tillage as a strategic tool to manage problematic weeds and break disease life cycles (Crawford *et al*. 2015). In the current study, the site had been cropped using no-till or minimum tillage for at least 5 years before the experiment commenced. The plots under the tilled treatment were cultivated over two years in 2012 and 2013. Results from the current study showed that the tilled treatment did not significantly increase N_2O emissions compared with the no-till treatment ($P > 0.05$, Table 3); although slightly higher N_2O emission flux was recorded for several months after tillage and sowing operations (Fig. 1), probably due to improved soil aeration enhancing soil nitrification rates (Chatskikh and Olesen 2007). No difference in surface soil mineral N was found in any months sampled between tilled and no-till treatments (Fig. 2). Our results, therefore, indicated that short-term tillage (over two years) had a negligible effect on N_2O emission. However, Chatskikh and Olesen (2007) reported that conventional tillage doubled the N_2O emission compared with direct drilling in spring and summer 2004 under spring barley from loamy sand soil at Foulum, Denmark. Chatskikh and Olesen (2007) suggested that it is likely that tillage affected N_2O emissions and crop growth through different processes, where effects of soil compactness reduced root penetration and soil aeration and associated gas diffusivity increased soil organic matter turnover. In contrast, Ball *et al*. (1999) and Skiba *et al*. (2002) reported that direct drilling enhanced N_2O emission by reducing gas diffusivity and air-filled porosity under heavy rainfall. More research is needed to give farmers more confidence to use tillage strategically to manage herbicide-resistant weeds and reduce incidence of soil-borne diseases.

There is ample evidence that increased N application increases N_2O emission in many environments (Wang *et al*. 2011; Harris *et al*. 2013; Schwenke *et al*. 2015). In the current study, the N_2O emission increased significantly immediately after fertiliser N was top-dressed compared with the nil-N treatment (Fig. 1). After crop harvest, soil mineral N increased and peaked in February 2014 (Fig. 2). High soil temperature, coupled with episodic summer rainfall events, stimulated soil microbial activity, resulting in N mineralisation from crop residues and soil organic matter with concomitant increased N_2O emissions. The significantly higher soil mineral N under the 100 kg N/ha treatment, particularly during the fallow period, was either due to over-supply of fertiliser N under that treatment, or rapid breakdown of the N-rich canola residues (Schwenke *et al*. 2015). Over the 365-day measurement period, applying 100 kg N/ha increased the annual N_2O emission by 28% compared with the nil-N treatment, most probably due to increased soil nitrification and denitrification processes resulting from the higher soil N status (Fig. 2). However, there was no relationship between daily N_2O emission rate with soil mineral N, probably because the soil sampling was not frequent enough to document the rapid changes in soil mineral N content occurring due to rainfall events.

Results from ^{15}N recovery measurements showed that the total recovery of applied ^{15}N from plant and soil was ~54% with no difference between N rates. This result is similar to the result on a neutral-alkaline Black Vertosol in the Liverpool Plains where Schwenke and Haigh (2016) reported that 55–57% of the applied N was recovered from the 40 and 120 kg N ha^{-1} treatments, but substantially less than the 76% recovery measured after sorghum crops grown on Vertosols in the northern Australian grains region (Armstrong *et al*. 1996). The unaccounted N (~46%) at the current site is presumably lost to the atmosphere as gases, such as NH_3, N_2 and N_2O as N leaching beyond 0.30 m was negligible (Table 2). With an extremely low EF (0.05%) for N_2O in this dry environment (Table 4), the majority of the gaseous loss was probably from NH_3 and N_2 loss. Volatilisation of NH_3 could be significant (Freney *et al*. 1983; Turner *et al*. 2012) as N was top-dressed and no significant rainfall fell until 5 days after N was applied (17 mm, Fig. 1). Emission of N_2 by denitrification during the rest of the season may be responsible for the remainder of the total N loss. An incubation study with intact soil cores from tropical savanna and grasslands of northern Australia also indicated that N_2 emissions accounted for 82.4–99.3% of total N lost (Werner *et al*. 2014). Therefore, we suggest that measurement of gaseous NH_3 and N_2 losses could provide greater insight into N losses from semiarid Mediterranean cropping environments and ultimately help identify agronomic options for improving crop N recovery to minimise losses to the environment.

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