Assessment of Baseline Nitrous Oxide Emissions in California Cropping Systems

FINAL REPORT

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Abstract
Production of the greenhouse gas (GHG) nitrous oxide (N_2O) results from the activity of soil microorganisms. Nitrogen (N) inputs, soil moisture, and carbon (C) availability stimulate the production of N_2O which contributes about one third of the total GHG emissions from California's agriculture sector. Crop yields and cumulative N_2O emissions were measured for two years in tomato, wheat, lettuce, and rice cropping systems fertilized at various N rates. Alfalfa was also monitored. The N_2O emissions were observed to occur mainly during the growing season from soils with high moisture content after N fertilizer applications. In most cases, N_2O emissions increased with increasing N rates. The levels of N application rates that resulted in the lowest N_2O emissions while not reducing yields were identified. In alfalfa, a recently established one-year old stand released half as much N_2O compared to a 5-year old stand. This study generated a database of field-level N_2O emission measurements and accompanying soil moisture and temperature, and mineral N values that can be used to calculate the percentage of applied N released as N_2O to estimate emissions on a larger scale, calibrate and validate N_2O emission biogeochemical models, and develop recommended best management practices.

Executive Summary

Background
Nitrous oxide (N_2O) production in agricultural soils is a microbial process that utilizes inorganic forms of nitrogen in soil. The processes influencing the release of N_2O are affected by fertilizer N inputs, soil resources such as available carbon, and soil moisture. Nitrous oxide emissions from agricultural soils contribute about one third of California agriculture's total net greenhouse gas (GHG) emissions [California Air Resources Board (CARB), 2011]. Under the statutory authority of the California Global Warming Solutions Act of 2006 (AB32), CARB has identified research on N_2O emissions from agricultural land as a priority. One of the limitations in understanding the factors affecting these N_2O emissions is the dearth of information to predict N_2O emission potential in the large variety of California cropping systems. Only few field level measurements of N_2O emissions have been published so far, and comprehensive datasets that could be used to calibrate and validate biogeochemical models or derive cropping system-specific emission factors (EFs) to calculate the percentage of fertilizer N emitted as N_2O have been missing altogether. There is an urgent need for baseline N_2O emission data to improve GHG inventories. The present study provides complete 2-year datasets of N_2O emissions and related agronomic and environmental variables in important crops (tomato, wheat, alfalfa, lettuce, and rice) that represent substantial acreage and production in California.

Methods
Nitrous oxide emissions and soil variables were regularly measured for two years in the above cropping systems under typical management with varying N fertilizer applications. Annual N_2O emissions were calculated and yields were measured to identify management practices that minimize N_2O without negatively affecting crop yield potential.

The 2-year studies were conducted at UC Davis Russell Ranch Sustainable Agriculture Facility (tomato), the Hartnell East College Ranch in Salinas (lettuce), Biggs Rice Experiment Station (rice), and in grower fields in the Sacramento Valley (wheat, alfalfa, rice) from 2009-2011. Nitrogen fertilizer was applied at 5 rates ranging from 0 to exceeding the highest rate reported for tomato, lettuce, and wheat. For alfalfa, a one-year and a 5-year old stand were selected. Rice was grown at 3 N rates either under non-flooded conditions for the first month and then flooded or sown into flooded soil.

Nitrous oxide emissions were measured several times per week when soil moisture was elevated after irrigation or rainfall events, and less frequently under dry conditions. The measurements were made by placing a vented chamber on the soil surface and sampling headspace at regular, timed intervals. The air samples were analyzed by gas chromatography and the flux of N_2O was calculated from the change in N_2O concentration over time. The annual N_2O emissions were calculated by converting the measured fluxes to daily fluxes and interpolating between daily fluxes. The emission factors were calculated by dividing the amount of annual N_2O-N emissions by the amount of annually applied N (fertilizer) inputs.

Soil moisture and temperature were measured with each chamber deployment, and available soil N was determined at 2-3 week intervals to obtain information on how environmental conditions and management affected N_2O emissions. Yields and crop N removal were measured to determine the maximum economic return to N rate.

Results

The N_2O emissions increased with increasing fertilizer N application rates. In the furrow-irrigated tomato systems, cumulative annual emissions ranged from 0.67 to 4.69 kg N_2O-N ha^{-1} and emission factors (EF) from 0.92 to 2.08%. At fertilizer N rates greater than 162 kg N ha^{-1} (maximum economic return to N rate), much higher N_2O losses occurred. The wheat experiments demonstrated that fertilizer type affected N_2O emissions more than N rates. Using a concentrated form of N (anhydrous ammonia), injected into the soil in bands as starter fertilizer, produced as much N_2O as the 56 kg higher application of a less concentrated N formulation that had been broadcast and incorporated. The EFs in the wheat systems ranged from 0.24 to 0.98% and were below the average EF of 1.21% calculated in 25 other wheat studies (Linquist *et al.*, 2012). In lettuce, which was drip-irrigated/fertigated, annual N_2O emissions ranged from 0.58 to 1.51 kg N_2O-N ha^{-1} and the EFs from 0.41 to 0.84%. An N rate lower than 168 kg N ha^{-1} consistently reduced N_2O emissions. Lettuce yields did not increase at an N supply greater than 84 kg N ha^{-1}. In alfalfa, most of the cumulative annual N_2O emissions were released in short, intense bursts immediately following flood irrigation of the fields during April-November. The annual N_2O emissions in the field of a 5-year old stand were more than twice as large (5.20 kg N_2O-N ha^{-1}) as those in the adjacent field of a 1-year old stand (2.30 kg N_2O-N ha^{-1}). The annual N_2O emissions in rice systems ranged from 0.26 to 0.85 kg N_2O-N ha^{-1}, resulting in EFs of 0.12 to 0.74%. Fertilizer N additions before flooding in a dry-seeded system increased N_2O emissions (1.13 to 1.20 kg N_2O-N ha^{-1}). The EFs in rice were similar to the average EF derived in 17 other studies (Linquist *et al.*, 2012), but the N_2O emissions must be evaluated in conjunction with methane emissions, which will covered in a separate study (Assa et al., in preparation; Adviento-Borbe et al., in preparation), because N_2O contributes only a small portion to rice total GHG emissions. All experimental data of this study will be made available for further analysis by a modeling study.

Conclusions

A comprehensive database of N_2O fluxes, soil moisture and temperature, as well as soil mineral N was compiled, thus filling a gap that has hampered efforts to calibrate and validate biogeochemical models of N_2O emissions. We produced refined emission factors for five major California crops. We also showed the biophysical and management factors that lead to N_2O production. This study confirmed that synthetic fertilizer N is a primary driver of N_2O emissions, and it indicated that, in general, N_2O emissions during the rainy season are less important than those during summer, unless a winter crop such as wheat is grown. Finally, we investigated the relationship among fertilizer N application, yields, and N_2O emissions and identified agronomic practices that potentially reduce N_2O emissions without compromising yield potential.

Future research should be directed towards strategies that increase N use efficiency, defined as the ratio of N in the harvested crop per unit N applied, and/or control soil moisture through alternative irrigation techniques. This study showed that often N_2O is emitted in spikes, or bursts, in response to high soil moisture levels. We recommend to further investigate sub-surface drip irrigation, especially in alfalfa, lettuce, and tomato systems. More data are needed on the fate of N fertilizers delivered through a drip system because leftover N not used by a crop can become the substrate of N_2O production after the growing season. Nitrification inhibitors that block the conversion of ammonium to nitrate also deserve further study because increased uptake of ammonium by crops would lower N_2O formation during the conversion of ammonium to nitrate. Lastly, better knowledge is needed on the fate of N in alfalfa systems since no data exist on the amount of N available to the next crop after an alfalfa stand has been removed.

1. Introduction

The California Global Warming Solutions Act of 2006 (AB 32) mandates that the State develops comprehensive strategies to reduce GHG emissions by 2020. Of particular concern is the emission of N_2O from agricultural fertilizer nitrogen (N) practices. In California, N_2O may contribute about 30% to the total net agricultural greenhouse gas (GHG) emissions (CARB, 2011). Therefore, for agriculture, the reduction in N_2O emission is key to reducing overall GHG emissions. We developed a comprehensive data set of N_2O emission and ancillary environmental and agronomic variables for dominant cropping practices in California. The cropping systems include tomato, wheat, alfalfa, rice and lettuce systems under typical management practiced in California. California agriculture is highly productive and accounts for 1 in every 4 items on the daily dinner tables of Americans. The high productivity is a result of exceptionally long growing seasons (both summer and winter), consistent water availability and intensive agronomic production practices. However, little information exists to assess the potential for N_2O production from California agriculture. Other agricultural regions are sufficiently different in climate and crops so that comparisons and extrapolation of existing data sets is not possible to estimate California agricultural emissions. This dearth of information on gaseous N losses in California agriculture is the subject of this report.

Among plant mineral nutrients, the availability of nitrogen is most frequently limiting growth and crop productivity. An adequate supply of nitrogen is therefore of the utmost importance to successful farming. Worldwide fertilizer N consumption is estimated at 96.8 million metric tons (Tg) (IFA, 2008). In California alone, about 0.6 Tg of N is applied annually (CDFA, 2009). Meta-analyses based on over 1000 studies found that increasing fertilizer N application rates significantly increase N_2O emissions, (Eichner, 1990; Bouwman *et al.*, 2002; Stehfest and Bouwman, 2006), and this trend is more pronounced at the high end of N application rates (>200 kg ha^{-1}). Several studies have shown that N_2O emissions increased sharply in response to N inputs that exceeded crop N requirements or economic N yield (McSwiney and Robertson, 2005; Edis *et al.*, 2008). Thus, fertilizer N is a primary determinant of N_2O emissions. However, crop N uptake, the timing and placement of the N application, as well as fertilizer type, influence the magnitude of the N_2O emissions (Mikkelsen, 2009; Burger and Venterea, 2011). On one hand, reducing fertilizer N inputs is probably the best strategy to mitigate N_2O emissions, but on the other hand, compromising yield potential for lack of N would be detrimental to food security and also overall greenhouse gas emission mitigation. For example, a crop that is under fertilized will not reach yield potential and leave residual N in soil leading to the potential release of N_2O. Therefore, to meet the challenge of greenhouse gas reduction, N_2O and yield responses to different fertilizer levels must be studied concurrently to establish optimum application rates. The key outcome of an optimum N fertilization rate is to mitigate N_2O emissions, not eliminate them.

Many factors control the release of N_2O from soils. In addition to fertilizer N, soil mineralization, soil moisture and C substrate availability control N_2O emissions. Denitrification occurs under oxygen (O_2) limitation, typically when diffusion of O_2 from the atmosphere into the soil is limited at high soil water content, for most soils at a water-filled pore space (WFPS) >60% (Linn and Doran, 1984). The highest N_2O fluxes occur at WFPS 60-90% (Linn and Doran, 1984; Davidson, 1992; Dobbie *et al.*, 1999; Simojoki and Jaakkola, 2000). Nitrous oxide is also produced during nitrification although the exact mechanisms are not as well understood as those of denitrification (Bremner and Blackmer, 1978; Wrage *et al.*, 2001). The main driver of nitrification and production of N_2O during nitrification is NH_4^+ availability. Irrigation and rainfall events stimulate microbial activity, including nitrification, denitrification, and N_2O production, and were, therefore, a focal point of our N_2O emission measurements. Due to California's mild winter temperatures and seasonal rainfall patterns, substantial N_2O losses may also sporadically occur during the winter rainy season (Kallenbach *et al.*, 2010), but to-date N_2O emissions during the rainy season have not been comprehensively assessed in any of California's cropping systems.

Few N_2O emission data from California cropping systems have been published (Ryden and Lund, 1980; Burger *et al.*, 2005; Steenwerth and Belina, 2008; Kong *et al.*, 2009; Lee *et al.*, 2009; Kallenbach *et al.*, 2010; Garland *et al.*, 2011) in comparison to other agricultural regions, and this lack of field measurements has hampered efforts to calibrate and validate existing models (Li *et al.*, 2004) and to develop California-specific emission factors (EF, the fraction of the applied fertilizer N lost as N_2O to the atmosphere). Currently, the CARB uses the IPCC (tier 1) approach with a default EF of 1% , a statistically derived value based on a meta-analysis of available data (Bouwman *et al.*, 2002), for regional and statewide estimates of N_2O emissions.

The objective of this study was to determine annual N_2O emissions in cropping systems representative of a large acreage of California's crop land, i.e. in tomato, lettuce, wheat, alfalfa and rice cropping systems, under typical management practices for a period of two years. In addition, in tomato, lettuce, wheat, and rice, N_2O emissions in response to varying amounts and types of fertilizer nitrogen applications were measured. Furthermore, we calculated emission factors of N_2O-N as a fraction of the fertilizer N applied. We also measured yields and crop N uptake in order to start developing best management practices that keep N_2O emissions as low as possible without negatively affecting yield potential. Lastly, we identified management events and practices, and seasonal periods that potentially produce substantial N_2O emissions and discuss management options that show potential to mitigate N_2O emissions.

2. Materials and Methods

2.1. *Selecting representative fields of tomato, wheat, alfalfa, rice, and lettuce cropping systems in the Sacramento Valley and Coastal region.*

The tomato system experiments were conducted at the UC Davis Russell Ranch Sustainable Agriculture research site (38°32’30”N, 121°52’30”W). The soils are classified as Yolo silt loam, a fine-silty, mixed, non-acid, thermic Typic Xerorthent and Rincon silty clay loam, a fine monmorillonitic, thermic Typic Haploxeralf (Table 1). The rotation was tomato/wheat, but in the rainy season following tomato, fields were left fallow.

Table 1. Soil characteristics (0-30 cm) tomato cropping system.

Sand (%)	21.83
Silt (%)	47.00
Clay (%)	31.17
pH (H_2O 1:1)	6.8
Bulk density beds 5-15 cm (Mg m^{-3})	1.37
furrows	1.52
Organic C (g kg^{-1})	10.3
Organic N (g kg^{-1})	1.0

For the wheat study, grower fields near Dixon (approx. 38°26’N, 121°52’W and 38°30’N, 121°50’W) were selected. The field monitored in the first year (2009-10), which was planted in tomato in the preceding summer, contained both a Capay silty clay loam that has a fine, montmorillonitic, thermic Typic Chromoxerert characterization and a Yolo silty clay loam soil type (Table 2). The field used in the second year (2010-11), also a Yolo silty clay loam, had previously been planted with alfalfa for 4 years.

Table 2. Soil characteristics (0-30 cm) wheat cropping systems.

	2009/10	2010/11
Sand (%)	31	21.3
Silt (%)	44	43.7
Clay (%)	25	35
pH (H_2O 1:1)	7.2	7.4
Bulk density beds 5-15 cm (Mg m^{-3})	1.27	1.29
furrows	1.42	
Total C (g kg^{-1})	12.8	14.9
Total N (g kg^{-1})	1.1	1.3

Two adjacent grower fields in the vicinity of Winters (approx. 38°35’N, 121°50’W) were used for the alfalfa study. The soil is a Myers clay, which is a fine, montmorillonitic, thermic Entic Chromoxerert (Table 3). One of the fields was freshly planted to alfalfa in fall 2009, whereas the other field was in its 4th and 5th year during the experiment. Nitrous oxide monitoring started in March 2010.

Table 3 Soil characteristics (0-30 cm) of the alfalfa systems.

Sand (%)	23
Silt (%)	43
Clay (%)	34
pH (H_2O 1:1)	7.7
Bulk density 5-15 cm (Mg m^{-3})	1.43
Total C (g kg^{-1})	12.6
Total N (g kg^{-1})	1.1

The rice study was initially conducted at the California Rice Experiment Station (RES) near Biggs (39°27’31”N, 121°44’23”W), starting in spring 2009. Soils at this site are classified as an Esquon-Neerdobe Complex (fine, smectitic, thermic Xeric Epiaquerts and Duraquerts) (Table 4). In the fall of 2009, all fields were leveled and the experiment station was re-designed in a completely different configuration. Our study then continued on grower fields near Arbuckle (approx. 38°50’N, 121°55’W). The soil type was Clear Lake clay.

Table 4. Soil characteristics (0-30 cm) of the rice systems.

	RES Biggs	Arbuckle
Sand (%)	29	10
Silt (%)	26	32
Clay (%)	45	58
CEC (meq/100 g)	34.2	
pH (H_2O 1:1)	5.0	
Bulk density 5-15 cm (Mg m^{-3})	1.25	
Organic C (g kg^{-1})	10.6	15.2
Total N (g kg^{-1})	0.8	1.4
Olsen-P (mg kg^{-1} soil)	13.4	
Electrical conductivity (dS m^{-1})	0.36	
Extractable potassium (cmolc kg^{-1})	0.44	

The lettuce cropping system monitored was located at the Hartnell College East Campus Farm in Salinas CA (36°40’14.39”N, 121°36’22.50”W). The experimental field site soil was a Chualar loam, which is a fine-loamy, mixed, thermic Typic Argixeroll (Table 5).

Table 5. Soil characteristics (0-30 cm) at the lettuce field site.

Sand (%)	54
Silt (%)	29
Clay (%)	17
CEC (meq/100 g)	15.2
pH (H_2O 1:1)	7.57
Bulk density beds 5-15 cm (g m^{-3})	1.46
furrows	1.69
Total C (g kg^{-1})	12.7
Total N (g kg^{-1})	1.0
Olsen-P (mg kg^{-1} soil)	41.03

2.2a. *Nitrogen fertilization*

In the **tomato** cropping systems, in spring 2009 before planting, NPK-15-15-15 starter fertilizer (8.7% NH_4^+, 6.3% NO_3^-) in granular form at the rate of 50 kg N ha^{-1} was banded at a depth of about 16 cm, except in a 4.6 mx 4.6 m area (4.6 m long section of 3 beds) designated as 'zero-N' treatment. Side dress N in the form of ammonium sulfate ($(NH_4)_2SO_4$) was banded six weeks after planting at a depth of 17 cm, 20 cm from the plant line, in 4.6 m x 4.6 m microplots to bring the total N application (starter N and side dress N) to 75, 162, 225, and 300kg N ha^{-1}. The experimental set-up was a randomized complete block design with three replicates. All microplots were in the same three beds. Monitoring of N_2O flux started in the fall 2009. In spring 2010, a new set of plots within the same rotation was fertilized in the same manner (NPK application April 10), but urea ammonium nitrate (UAN32) was used as side dress (applied May 30) instead of $(NH_4)_2SO_4$. Nitrous oxide flux was monitored in these plots through spring 2011, when a new set of previously fallow plots was chosen. The fertilization regime was the same as in the previous year (NPK applied April 12; UAN32 application May 13), but the size of the microplots had been increased to 4.6 m x 9.1 m.

In the **wheat** systems, in the 2009-10 growing season, 5 experimental treatments were set up in 3.7 m x 4.6 m microplots in each of three corners of a 32-ha grower field. A total of 91, 151, 203, and 254 kg N ha^{-1}, in addition to a zero-N treatment, were applied during the growing season in the following manner: Before planting (Nov 2), 60, 112, and 163 kg N ha^{-1} were applied as $(NH_4)_2SO_4$, anhydrous ammonia (AA; grower practice), and $(NH_4)_2SO_4$, respectively. The $(NH_4)_2SO_4$ was applied by hand at a depth of 13 cm in bands, carved out with a pick-axe, spaced 30 cm apart of each other. On Feb 12, 2010, there was a 91 kg N ha^{-1} aerial application of urea on the entire field except the three zero-N plots, which had been covered with tarps. The 2010-11 experiment was conducted in a different field. The experimental plots received 0, 154, 210, or 266 kg N ha^{-1}. At planting (Nov 4), 56, 112, and 168 kg N ha^{-1} as $(NH_4)_2SO_4$ was surface-applied by hand and then disked in. Additionally, as a separate treatment, the grower injected 112 kg anhydrous ammonia-N ha^{-1}. There was an aerial application of 98 kg N ha^{-1} in the form of urea on the entire field (Feb 22) except the zero-N plots. So there were two treatments that received a total of 210 kg N ha^{-1}, one as $(NH_4)_2SO_4$ (starter) plus urea (aerial side dress), and the other as AA plus urea. The experiment was set up as randomized complete block design with 3 replications. Each block was in a different area of the 23-ha field. For practical reasons, the microplots and the AA-injected area (most of the field) were separate of each other, but all the experimental plots of each block were in the same general area of the field. Each microplot plot was 6 m x 6 m.

The **alfalfa** fields received 17 kg N ha^{-1} as ammonium phosphate fertilizer once during the 2-year monitoring period in fall 2010. The purpose of this fertilizer application was to supply phosphorus.

The N treatments in the **lettuce** system were as follows: 11, 84, 168, 252, and 336 kg N ha^{-1}. In both years, all plots received 11 kg N ha^{-1} as UAN32 10 d after planting. The remainder of the N fertilizer was applied, also as UAN32, via three fertigations, one week apart of each other, starting 5 weeks after planting. The drip tape was 6 cm below the surface. The study was conducted in a 0.25 ha field in a randomized complete block design with four replications. Two 50 m-long beds (bed width 0.68 m) and furrows (furrow width 0.34 m) represented an experimental unit (50 m x 2.04 m). Planting was on June 24, 2009, and June 23, 2010, and harvest took place on August 27, 2009, and on August 23, 2010.

At the Biggs **Rice** Experiment Station, we monitored N_2O emissions during the pre-flooding period and after fall drainage preceding harvest. In the rice experiments, different management systems were studied. The management systems were water-seeded conventional (WSCT), water-seeded no-till stale seedbed (WSNT) and drill-seeded no-till stale seedbed (DSNT). The last two treatments were developed to control weeds. Stale seedbed refers a period of time were the rice paddys are frequently flushed with water to promote weed seed germination. Herbicides are applied during this period to control weed growth for the remainder of the season (see appendix for more details). In the 'water-seeded conventional' treatment, urea was applied to the soil surface incorporated with a harrow just before the permanent flooding for seeding. In the 'water-seeded no-till stale seedbed' treatment, urea was applied to the soil surface the day before the permanent flooding for seeding. In the 'drill-seeded no-till stale seedbed,' 28 kg urea-N ha^{-1} was applied three weeks after planting and the remainder was applied to the soil surface immediately before the permanent flooding four weeks later. Applying a small amount of N fertilizer before flooding was a one-time experiment that is not typically practiced. The N fertilizer treatments in each of the three management practices were 0, 168, and 224 kg N ha^{-1}. There were four replicates per management practice and fertilizer N level. At the sites near Arbuckle, the wet-seeded field received 140 kg N ha^{-1} as Aqua ammonia; the dry-seeded field was fertilized with 100 kg urea-N ha^{-1}. There were three replicates per field.

2.2b. *Nitrous oxide flux measurements*.

In each system, data were collected for two years. We measured N_2O flux intensively immediately before and after N fertilization and irrigation or rainfall events to capture the extent of elevated N_2O fluxes until the fluxes subsided to background levels. After N_2O flux receded and soils were relatively dry, measurements were taken less frequently.

Nitrous oxide flux was measured, using a static chamber technique (Hutchinson and Livingston, 1993). Round PVC chambers (25.4 cm diameter) were used in the wheat and rice systems. In tomato and lettuce, rectangular thin-wall stainless steel chambers, 50 cm x 30 cm, were used on beds, and 10 cm-diameter PVC chambers were used in the furrows. The chamber dimensions in the alfalfa systems were 15 cm by 14 cm. The height of all chambers was 10 cm. In tomato and lettuce systems, the chamber bases had a 2 cm-wide horizontal flange at the top end and were inserted 8 cm deep into the soil, so that the flange was resting on the soil surface. The bases for the round PVC chambers were also inserted 8 cm deep into the soil, but these rings extended 5 cm above the soil surface. The chamber bases were left in place unless field operations required their temporary removal. During sampling, the vented (4.8mm dia., 10 cm long tubes) and insulated chambers were fitted onto the bases and headspace air was removed from a sampling port containing a butyl rubber septa via syringe and needle 0, 20 and 40 min after deploying the chamber tops onto the bases. When N_2O fluxes were expected to be high, samples were taken from the chamber at shorter intervals (0, 15, and 30 min). To collect a gas sample from the chamber, headspace air was removed by inserting the needle of a polypropylene syringe (Monoject) through the septum of the sampling port and by slowly withdrawing 20 mL gas. The gas in the syringes was immediately transferred into evacuated 12-mL glass vials with grey butyl rubber septa (Exetainer, Labco Ltd., Buckinghamsire, UK).

The gas samples were analyzed on a Shimadzu gas chromatograph (Model GC-2014) with a ^{63}Ni electron capture detector (ECD) linked to a Shimadzu auto sampler (Model AOC-5000). The autosampler uses a gas-tight syringe to remove 2 mL gas from the sample vials and injects it into the GC port. The GC uses as carrier gas a mixture of helium and P5 (mixture of 95% argon and 5% methane. The carbon dioxide (CO_2) and N_2O are separated by a Haysep Q column at 80° C. The ECD is set at 320° C and the pressure of the carrier gas flowing into the ECD is 60 kPa. The minimum quantity of N_2O detected by this GC system is 0.1 pg s^{-1}. After the acquisition of the sample, the autosampler's syringe and the GC's sample loop are purged with helium to back flush water and other slow chromatically resolved analytes.

The GC system was calibrated daily using analytical grade N_2O standards (Airgas Inc., Sacramento CA). Quality assurance of the N_2O values generated by the GC and its software was obtained by processing N_2O standards in exetainers after taking them to the field and treating them the same way as field samples. The two standard preparation approaches ensured quality assurance of the lab and field protocols used in this study. Samples were analyzed within two weeks of collection and their quality was insured by ascertaining that the field N_2O standards were not compromised as a result of extended storage. During one period in fall 2010, samples were stored up to five weeks because of a bottleneck due to autosampler problems. However, during this time, the most critical samples (the ones with presumably high N_2O concentrations) were processed on another GC within the usual two-week time frame.

Gas fluxes were calculated from the rate of change in chamber concentration, chamber volume, and soil surface area (Hutchinson and Mosier, 1981). Chamber gas concentrations determined by GC (volumetric parts per million) were converted to mass per volume units assuming ideal gas relations using chamber air temperature values, which were measured by a thermocouple thermometer during each sampling event. Two separate flux calculations were made. The first used an algorithm appropriate for curvilinear concentration data with time when N_2O concentration in the chamber increased at a decreasing rate (Hutchinson and Mosier, 1981; (Hutchinson and Livingston, 1993) and linear regression at all other times. All the emission estimates presented in this report are based on this widely accepted method to calculate soil-to-atmosphere gas flux when at least 3 data points (N_2O concentration at 3 time intervals) are available. The calculation compensates for the diffusion constraints imposed by the rapid increase in the partial pressure of certain gas species (e.g. N_2O) within the chamber when the flux is high. The second method used assumed a linear increase in N_2O concentration in the chamber at all times. A comparison of the results calculated by the two methods is presented in the appendix.

2.3. Annual N_2O emissions.

The annual N_2O emissions were calculated by trapezoidal integration of daily fluxes under the assumption that the measured fluxes represented mean daily fluxes, and that mean daily fluxes changed linearly between measurements. Differences in time-integrated annual N_2O emissions between N fertilization treatments were assessed using analysis of variance (ANOVA) and standard mean separation procedures. Appropriate transformation of the N_2O emission data was carried out for the statistical analysis whenever the data were not normally distributed, and specifics of statistical analyses are given in the Results section.

The emission factors (EF) were calculated as the percentage of N_2O-N divided by the amount of fertilizer N applied. Emission factors at each N fertilizer level were calculated for the different cropping systems. For alfalfa, the annual biomass N (BN_{alf}) inputs to derive the EFs for direct N_2O emissions were calculated according to IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006),

$$BN_{alf} = [Alfalfa_{DM} * N\ content * (.1)^{a} + Alfalfa_{DM} * (.4)^{b} * (.019)^{c}] * (1/4)^{d} \qquad \text{Eq. 1}$$

where, F_{N2O} = annual N_2O-N emissions; $_{DM}$ = dry matter; [a]the fraction of aboveground biomass not harvested; [b] default factor to estimate root dry matter as a function of annual aboveground biomass; [c]default N content of alfalfa roots; [d]the residue for perennial forage crops is only accounted for during pasture removal, which in alfalfa occurs about every 4 years, but to calculate BN_{alf} for the field with the 5-year old stand in the present study, the factor 1/5 was used.

2.4. Measuring environmental variables.

During each sampling event, in addition to chamber air temperatures, soil and ambient air temperatures were measured and gravimetric soil moisture in the 0-15 cm layer was determined. In addition, gravimetric soil moisture was calculated from field-moist and oven-dry (105ºC) mass of soil collected in the 0-15cm layer using a 1.83-cm steel corer. The gravimetric soil moisture values were converted to water-filled pore space values by using measured bulk density values in the 5-15 cm layer. The bulk density was measured twice per growing and rainy season by collecting 10 cm dia. x 7 cm long cores in the 5-15 cm layer of soil, followed by drying the cores to 105ºC. Inorganic N (NO_3^- and NH_4^+) in the 0-15 cm layer was measured approximately bi-weekly during the cropping season and monthly during the rainy season by extracting 15 g of well-mixed soil with 80 mL of 2M potassium chloride solution, and by analyzing the extracts colorimetrically for ammonium (NH_4^+) and nitrate (NO_3^-) by a Shimadzu spectrophotometer (Model UV-Mini 1240). For determining NH_4^+, the phenate (indophenol blue) method was employed (Forster, 1995). Nitrate in the extracts was reduced to nitrite (NO_2^-) with vanadium chloride, and the NO_2^- was analyzed by diazotizing with sulfanilamide followed by coupling with N-(1-naphthyl)ethylenediamine-dihydrochloride (Doane and Horwáth, 2003). The total C and N in soil of the 0-30 cm layer was measured by a C and N analyzer (Costech Analytical Technologies Inc., Valencia, CA) by the dry combustion method (Dumas, 1848) after grinding air-dried representative soil samples to a fine powder. The pH in the 0-30 cm layer of soil was measured in supernatant of soil slurries (soil/H_2O ratio 1:1) by a pH meter (Model 220, Denver Instrument Co., Arvada, CO). Soil texture was determined by a modified pipet method

(USDA,1992).

2.5. Yield Measurements, nitrogen use efficiency and N export.

Yields and N content of the harvested plant parts were measured in all systems. For tomatoes, fresh weights of tomatoes in two sub-samples (2m x 1.52 m) per microplot were recorded. A subset of the tomatoes was ground into a slurry. Dry weight of the fruit was assessed by lyophilizing a portion of the tomato slurry samples. The freeze-dried samples were then analyzed for total N content by the C and N analyzer. Wheat grain yields were assessed by harvesting plants in a 1 m x 0.91 m area within the microplots and separating straw and grain. The grain was oven dried at 60°C, ground to a fine powder and analyzed for total N as above. Sub-samples of the grain were dried at 105°C as the basis for calculating total harvested N. Wheat yields are reported at 13.5% moisture. Alfalfa yields were accounted for by the grower based on the number of bales produced in a field and an average mass of a bale. Alfalfa yields are reported at a moisture content of 18%. Total N content of alfalfa hay was measured by dry combustion as above after drying subsamples of plant biomass at 60°C. For lettuce, both trimmed lettuce biomass (marketable yield) and total, untrimmed biomass are reported. Dry biomass, total N content and N removal by the harvested crop were calculated after drying subsamples at 60°C and measuring total N content. Rice plants were harvested in 1-m^2 areas within each experimental unit (management practice and N level), straw and grain were separated, and N content in plant parts was determined by dry combustion.

The maximum economic return to N rate, i.e. the economic N yield, was inferred from the N application rates and yield data. The economic N yield and N_2O flux data in each system were used to evaluate the hypothesis that N_2O emissions increase non-linearly at N fertilizer levels exceeding crop N demand. Crop N removal and excess N left in the soil or lost from the system as a fraction of the N applied was calculated for each treatment in the different cropping systems.

3. Results

3.1. Tomato

We started monitoring N_2O flux in the fall 2009 and present results from November 2009 to October 2010 as year 1 and from November 2010 to October 2011 as year 2. In year 2, irrigation water inputs were greater than in year 1 (Figure 1). Soil and ambient air temperatures are shown in Figure 2. The water-filled pore space tended to be higher during the irrigation season (June-August) 2011 than in 2010. Figures 4 and 5 show daily N_2O fluxes and the timing of fertilizer applications. The total annual N_2O emissions in the 225 and 300 kg N ha^{-1} treatments were significantly higher than in the other treatments, as indicated by Tukey's mean separation procedure (Table 6, Figures 6 and 7). Total annual N_2O emissions increased with increasing N fertilizer applications in non-linear fashion. By including N fertilizer application rate both as a categorical effect and as a covariate (N rate) in a linear model (proc glm; SAS version 9.2), the linear and the non-linear effect of fertilizer N level on N_2O emissions was separated. A weighted least squares algorithm that uses the reciprocal of the residual variances as weighting factor was employed because residual errors increased with increasing N fertilizer levels (i.e. Levene test indicated lack of constant variance). The weighted ANOVA showed that the nonlinearity was highly significant ($P = .0012$).

Table 6. Results of ANOVA, using the reciprocal of the residual variances of mean N_2O emissions as weighting factor. Values designated with the same letters are not significantly different ($P < 0.05$). n =

Source	**DF**	**P > F**
Block	2	.0524
Year	1	.1014
N rate (covariate)	3	.0012
N rate*year	3	.2511

Means separation (Tukey's)

kg N ha^{-1}	**kg N_2O-N ha^{-1}**
0	1.02[b]
75	1.22[b]
162	1.80[b]
225	4.06[a]
300	4.34[a]

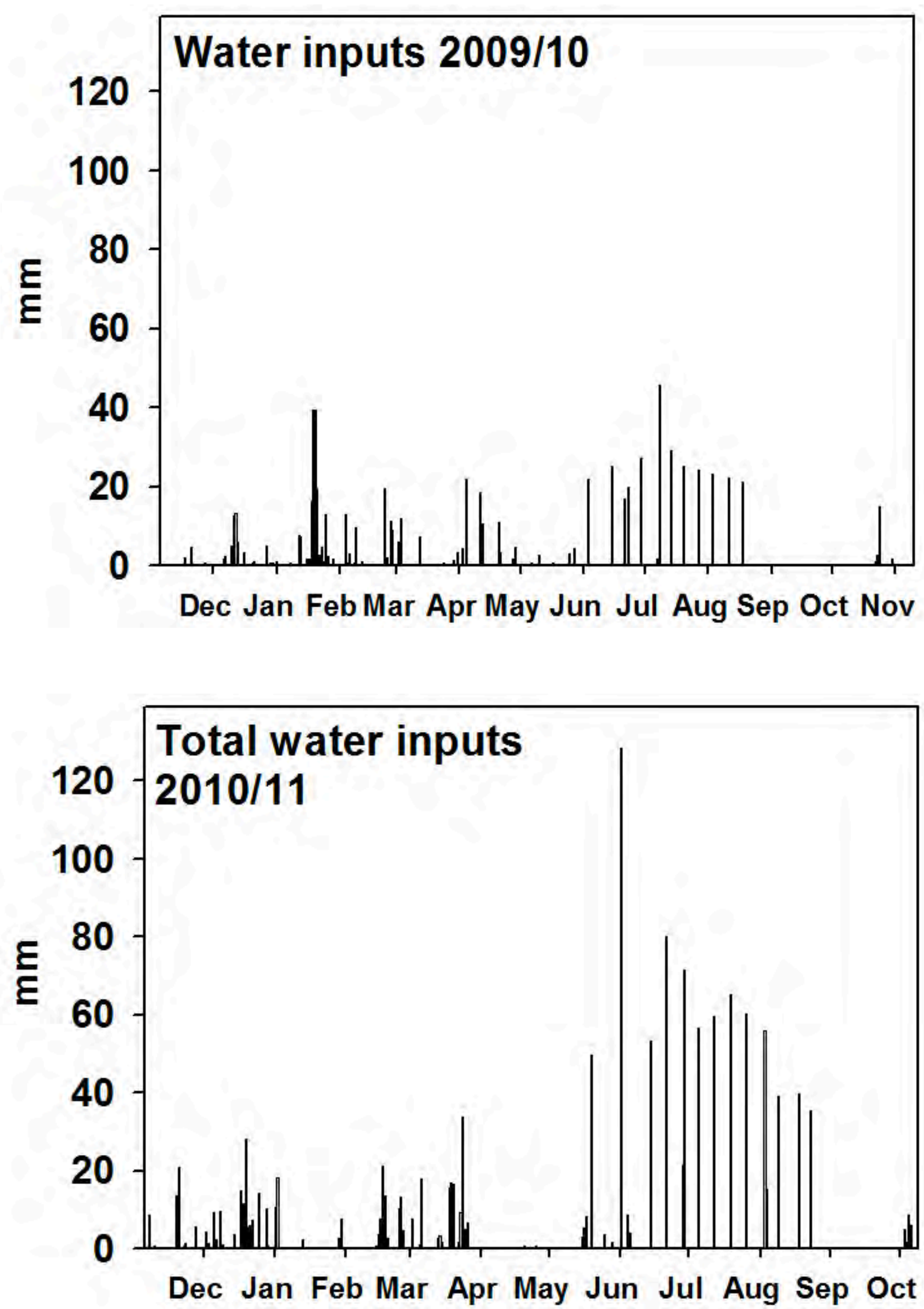


Figure 1. Daily rainfall and irrigation inputs in 2009/10 and in 2010/11 tomato systems at the Russell Ranch.

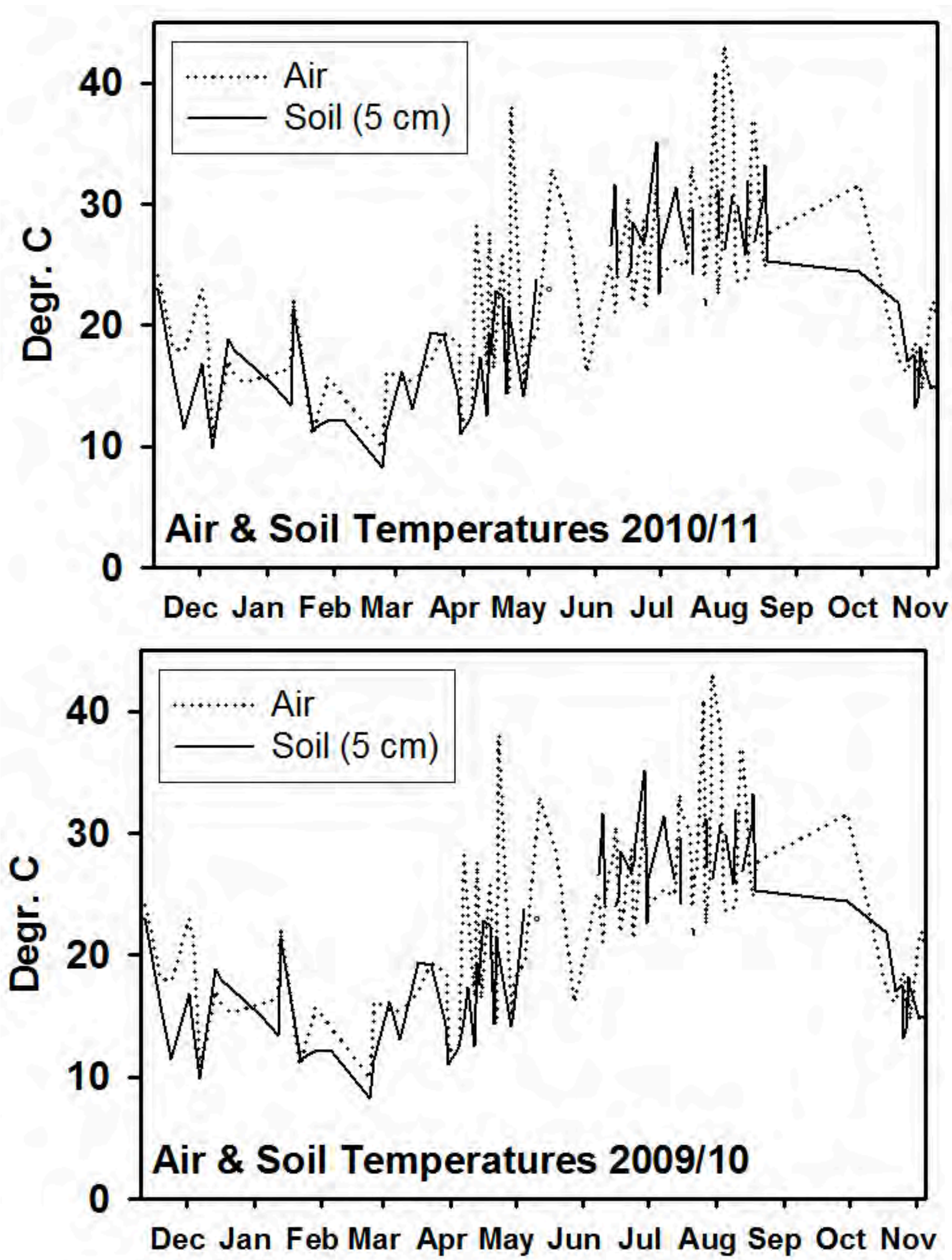


Figure 2. Ambient air and soil temperatures in 2009/10 and 2010/11 in the tomato systems at the Russell Ranch.

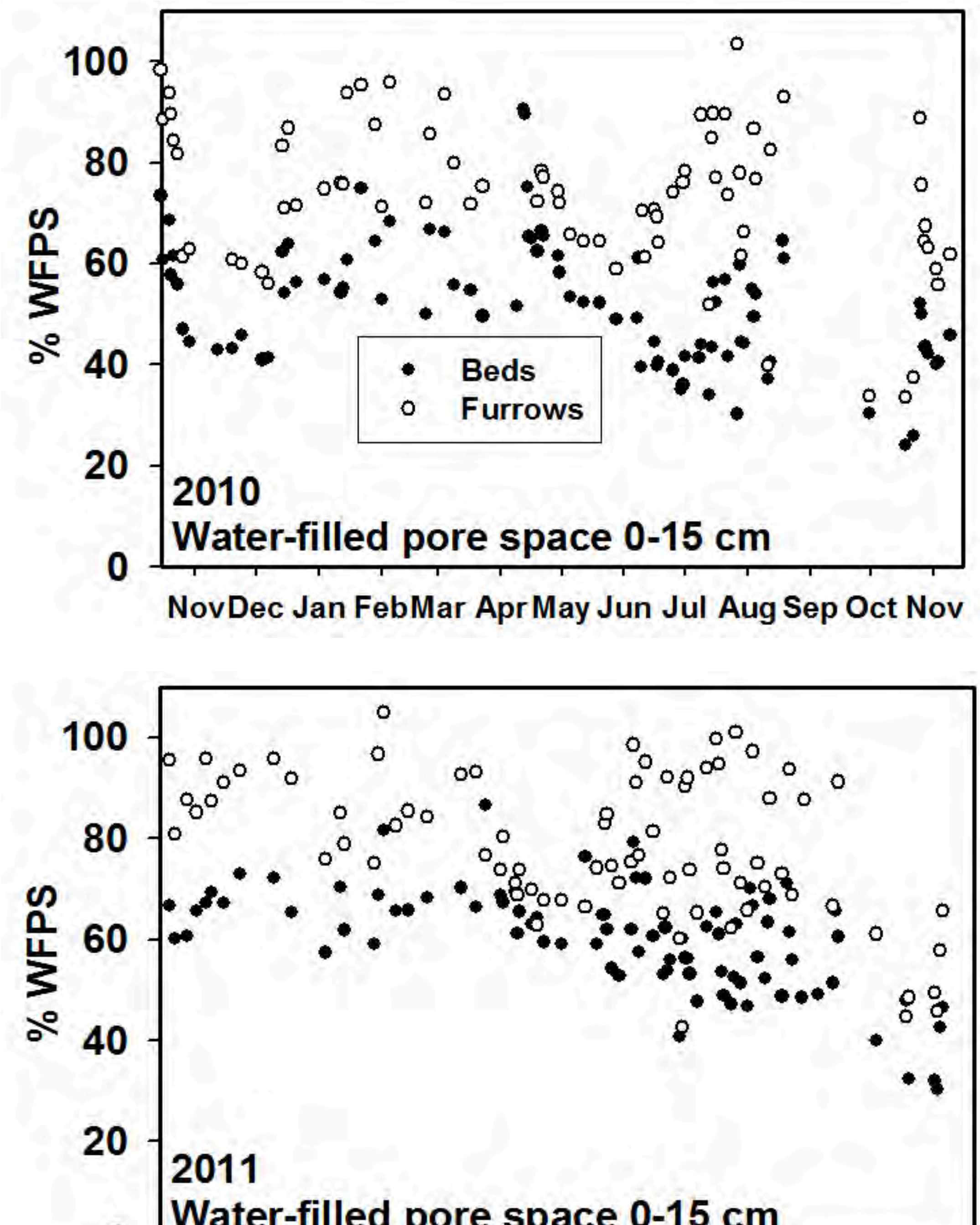


Figure 3. Water-filled pore space in the 0-15 cm layer in beds and furrows of tomato systems in 2009/10 and 2010/11.

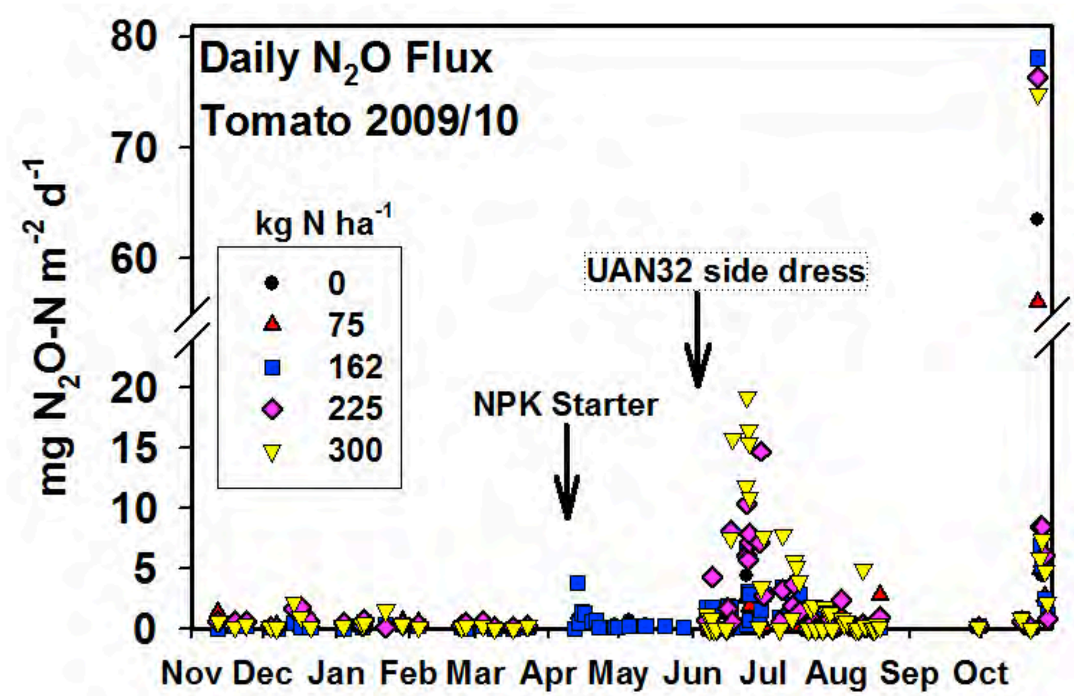


Figure 4. Daily N_2O fluxes in 2009/10 in tomato systems fertilized at five N levels.

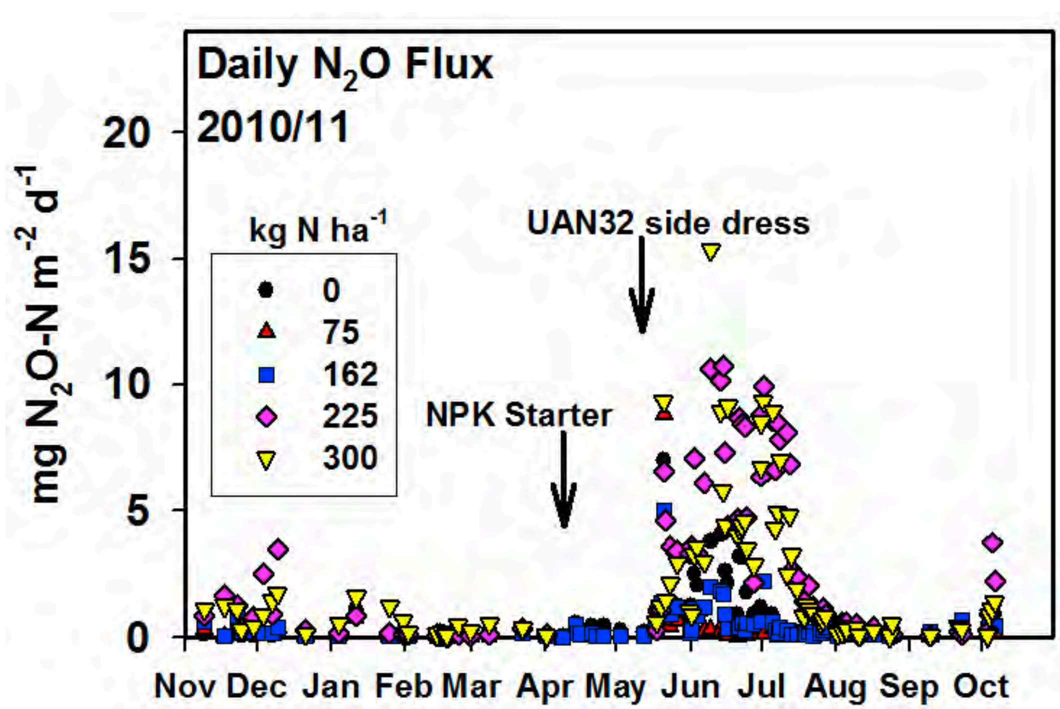


Figure 5. Daily N_2O fluxes in 2010/11 in tomato systems fertilized at five N levels.

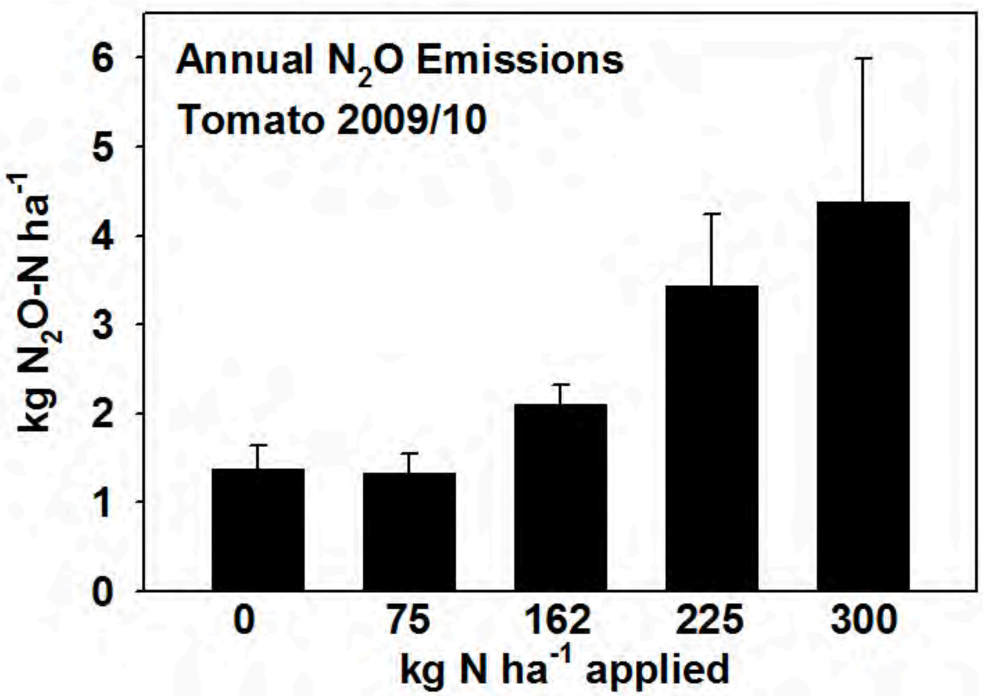


Figure 6. Total annual N_2O emissions in 2009/10 in furrow-irrigated tomato systems fertilized at five different N rates.

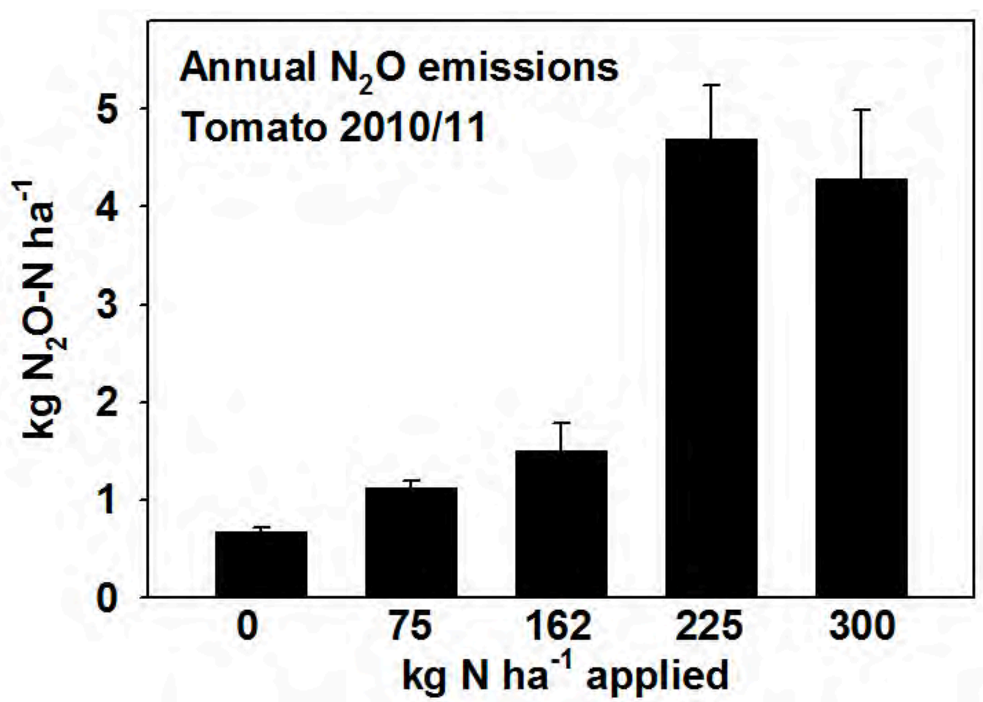


Figure 7. Total annual N_2O emissions in 2010/11 in furrow-irrigated tomato systems fertilized at five different rates.

Table 7. Annual N_2O emissions and standard errors, and emission factors (EF) in tomato, lettuce, wheat, alfalfa, and rice systems. Data collection was from 2009-2012.

Crop & practice	Annual N input	Emissions Year 1	Emissions Year 2	EF Year 1	EF Year 2
	kg N ha^{-1}	kg N_2O-N ha^{-1} yr^{-1}		%	
Tomato	0	1.37 ±.27	0.67 ±.04		
	75	1.33 ±.22	1.12 ±.08	1.77	1.49
	162	2.11 ±.21	1.50 ±.29	1.30	0.92
	225	3.43 ±.80	4.69 ±.55	1.52	2.08
	300	4.38 ±1.61	4.29 ±.70	1.46	1.43
Lettuce	11	0.58 ±.05	0.59 ±.13		
	84	0.71 ±.07	0.56 ±.03	0.84	0.67
	168	0.69 ±.07	1.13 ±.20	0.41	0.67
	252	1.09 ±.08	1.14 ±.14	0.43	0.45
	336	1.51 ±.27	1.42 ±.22	0.45	0.42
Wheat	0	0.24 ±.07			
	91	0.31 ±.08		0.34	
	151	0.57 ±.12		0.38	
	203 (AA)	1.31 ±.35		0.64	
	254	0.50 ±.13		0.20	
	0		0.72 ±.22		
	154		0.88 ±.18		0.57
	210		1.42 ±.10		0.67
	210 (AA)		2.05 ±.17		0.98
	266		2.15 ±.23		0.81
Alfalfa	43.1 (5-yr)		5.20 ±.79		12.06
	55.4 (1-yr)		2.30 ±.26		4.15
Rice*					
WSCT	168	0.26		0.15	
DSNT	168	1.13		0.67	
WSNT	168	0.48		0.29	
WSCT	224	0.27		0.12	
DSNT	224	1.20		0.54	
WSNT	224	0.56		0.25	
WSCT	140		0.85		0.61
DSCT	100		0.74		0.74

***Abbreviations of rice treatments: WSCT = wet-seeded conventional tillage; DSNT = drill-seeded no-till; WSNT =wet-seeded no-till; DSCT = drill-seeded conventional tillage.

In the tomato systems, the emission factors (EFs) ranged from 0.92% (162 kg N applied ha^{-1} in 2011) to 2.08% (225 kg N applied ha^{-1} in 2011)(Table 7). On average (2-year average), the EF in the 162 kg N ha^{-1} treatment was the lowest (1.11%), followed by the 300 (1.44%) and the 75 (1.62%) and 225 kg N ha^{-1} (1.80%) treatments. Overall, the tomato systems had the highest EFs among the fertilized cropping systems in this study (average of four N rates in tomato 1.50% compared to 0.42% in rice, 0.54% in lettuce, and 0.57% in wheat).

In both years, the major portion of N_2O emissions occurred during the tomato growing season, following the side dress application of UAN32 (Figures 8 & 9). In 2009/10, for the treatments with N rates higher than 75 kg N ha^{-1}, 49 to 68% of the total emissions occurred during this period; in 2010/11, 53 to 77% of the total N_2O emissions took place following the UAN32 side dress application. The N_2O emissions following the 50 kg N starter fertilizer application resulted in <10% of all the emissions occurring during the period between starter and side dress application. In 2009/10, the N_2O emissions with the first rainfall after summer accounted for 18 to 40% of the total annual N_2O emissions, but in 2010/11, this event caused <3% of the total N_2O emissions.

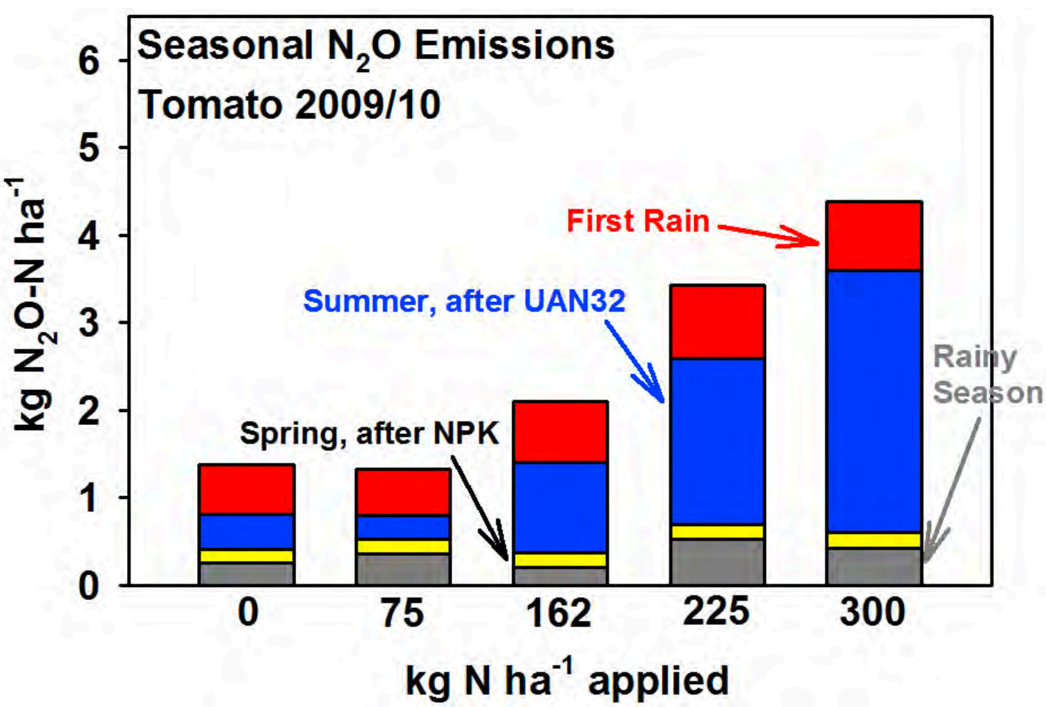


Figure 8. Nitrous oxide emissions in tomato systems in 2009/10 during the rainy season, between starter and side dress application of fertilizers, during the growing season and after the first rainfall after harvest.

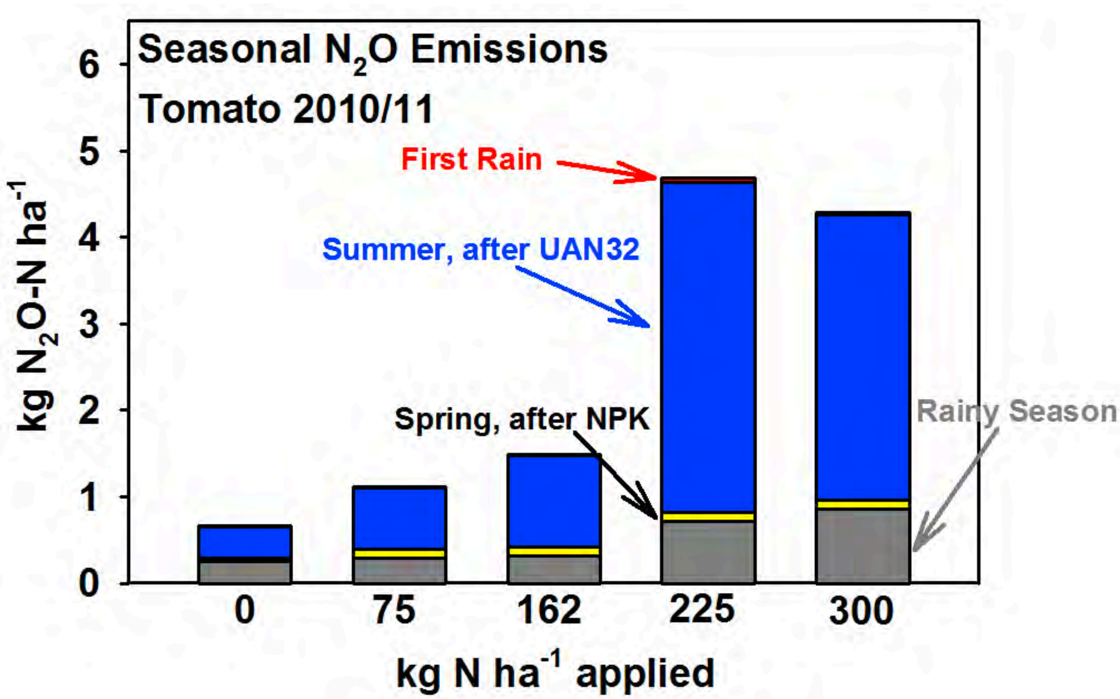
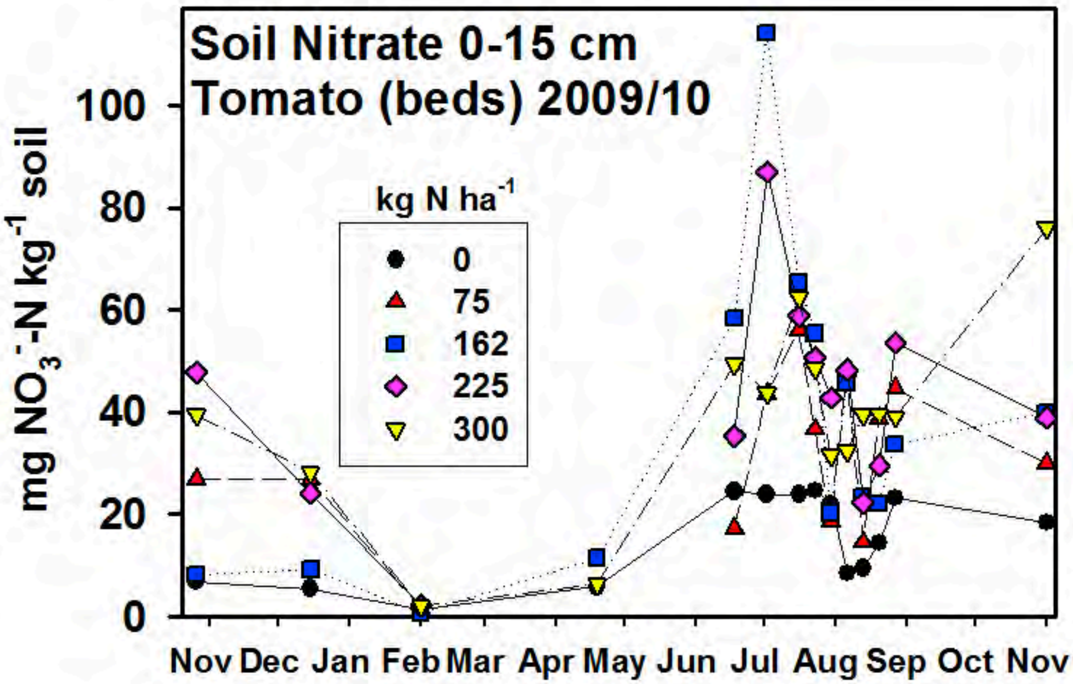
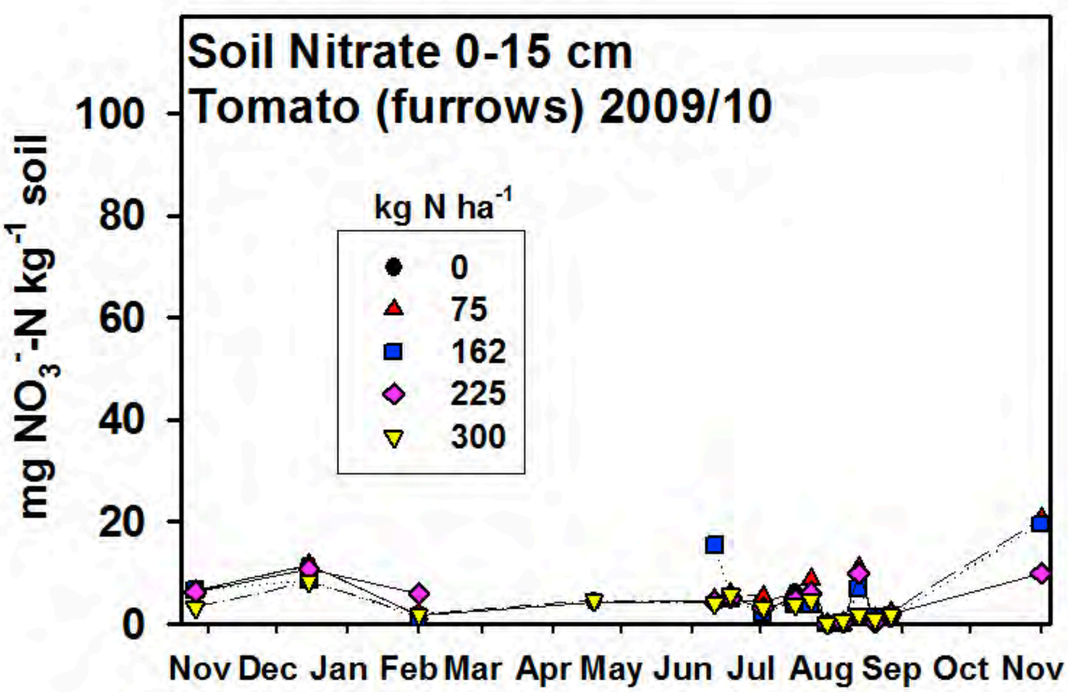


Figure 9. Nitrous oxide emissions in tomato systems in 2010/11 during the rainy season, between starter and sidedress application of fertilizers, during the growing season and after the first rainfall after harvest.

In 2009 and 2010, soil NO_3^- levels were relatively high in late fall, ranging from 17 to over 70 mg NO_3^- -N kg soil, which corresponds to 33 to 148 kg N ha^{-1} in the 0-15 cm layer alone (Figures 10 & 11). Nitrate levels declined throughout the rainy season, and increased to substantial levels again in the following growing season. In fall 2011, NO_3^- concentrations were lower than in the two preceding years. Nitrate concentrations in the furrows, in general, were much lower than in beds. Ammonium concentrations were somewhat elevated during the growing season after the UAN32 side dress application, but relatively low during all other times.


Soil Nitrate 0-15 cm
Tomato (beds) 2009/10
kg N ha-1
0
75
162
225
300
mg NO3--N kg-1 soil
0
20
40
60
80
100
Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov


Soil Nitrate 0-15 cm
Tomato (furrows) 2009/10
kg N ha-1
0
75
162
225
300
mg NO3--N kg-1 soil
0
20
40
60
80
100
Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov

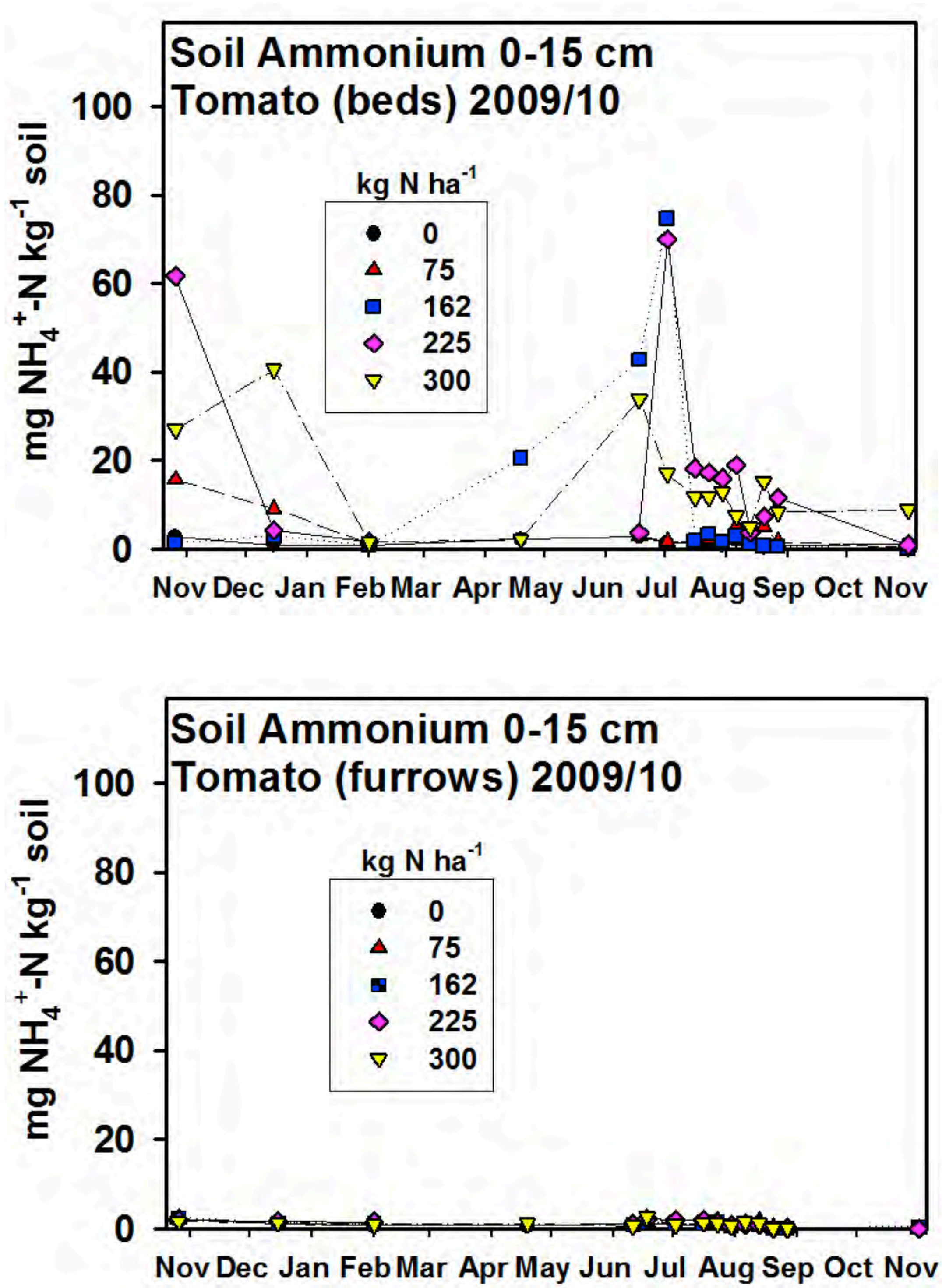
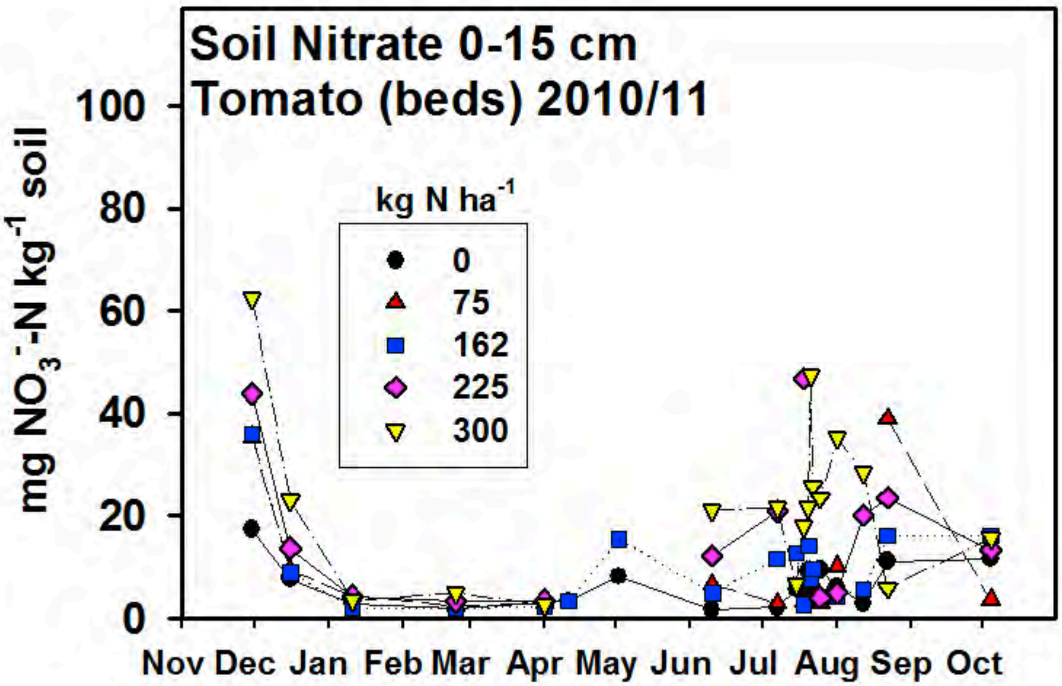
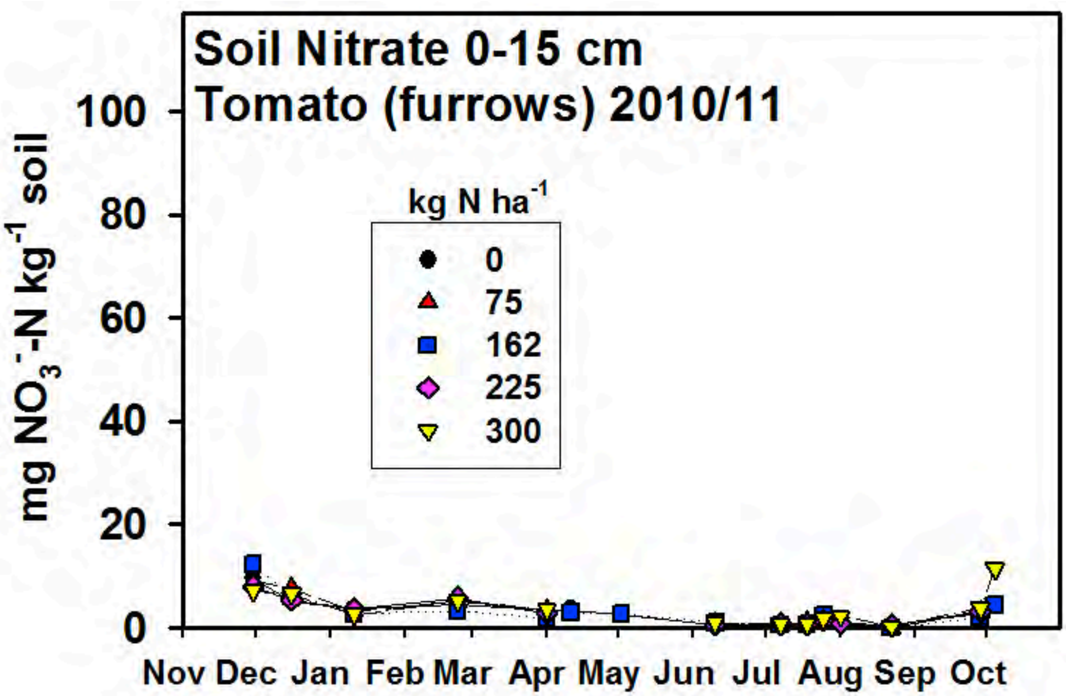
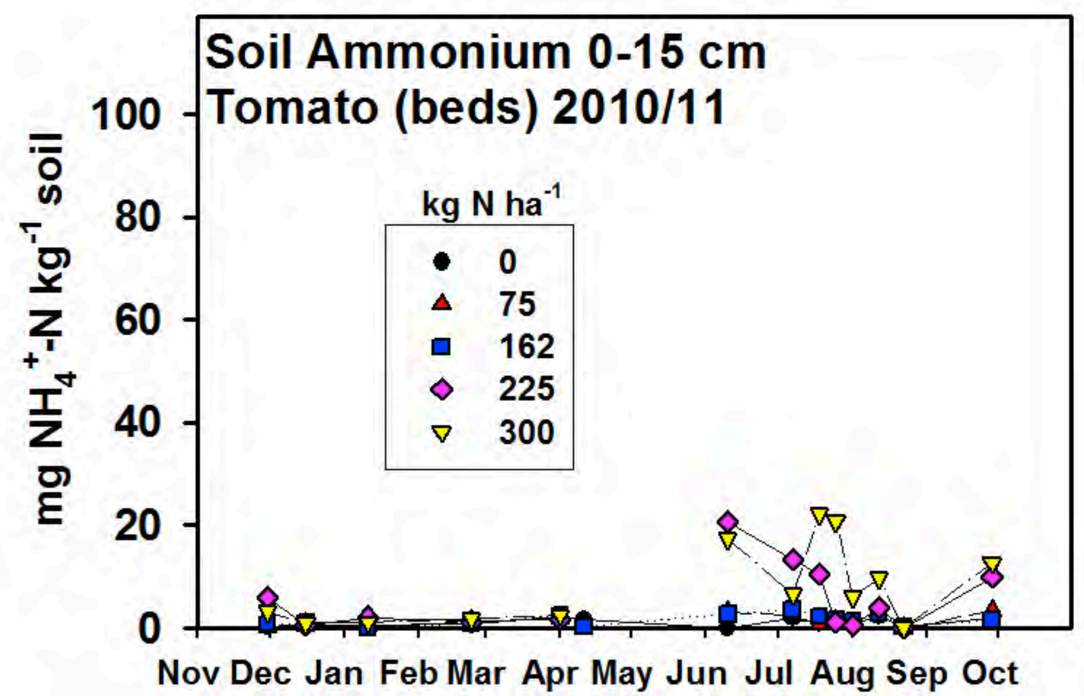


Figure 10. The above four panels show inorganic N (NO_3^- and NH_4^+) concentrations in the 0-15 cm soil layer in beds and furrows of tomato systems fertilized at five different N rates in 2009/10.


Soil Nitrate 0-15 cm
Tomato (beds) 2010/11
mg NO3--N kg-1 soil
kg N ha-1
0
75
162
225
300
0
20
40
60
80
100
Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep Oct


Soil Nitrate 0-15 cm
Tomato (furrows) 2010/11
mg NO3--N kg-1 soil
kg N ha-1
0
75
162
225
300
0
20
40
60
80
100
Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep Oct



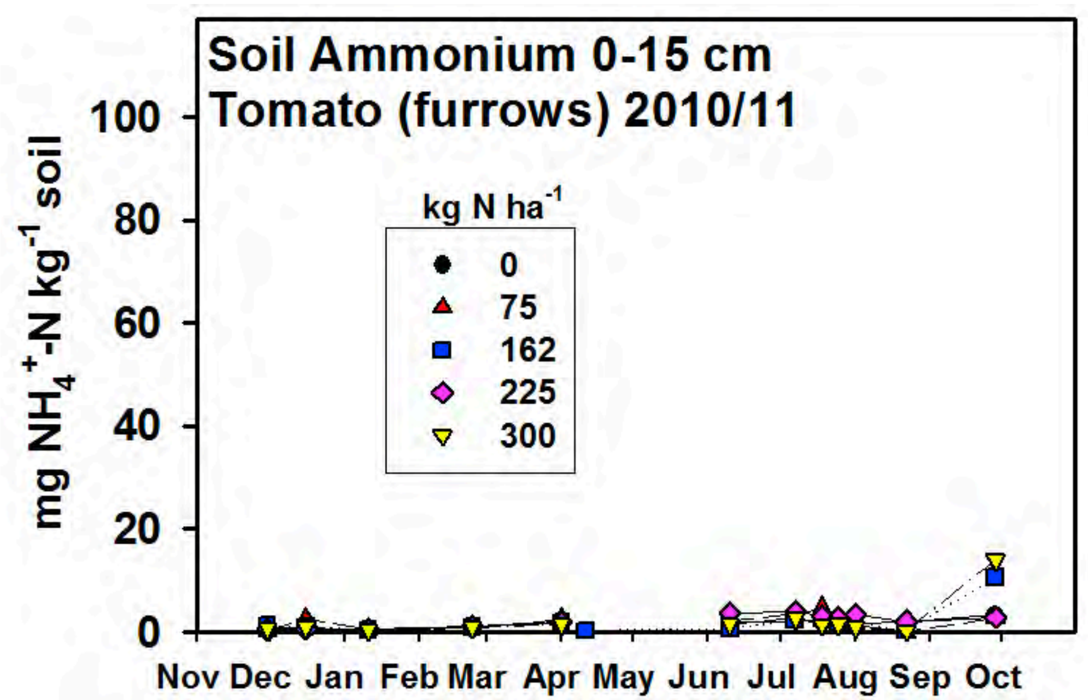


Figure 11. The above four panels show inorganic N (NO_3^- and NH_4^+) concentrations in the 0-15 cm soil layer in beds and furrows of tomato systems fertilized at five different N rates during 2010/11.

In 2010, tomato yields and the N removed by the harvested crop did not differ among the N fertilizer treatments (Table 8). Fruit N content was similar among all but the zero N treatment.

Table 8. Mean yields, tomato (red) fruit N content, and crop N removal in 2010 and standard errors. Values designated by the same letters are not significantly different. NS = non significant. n=3.

N application rate (kg N ha^{-1})	**Yield ($Mg\ ha^{-1}$)**	**Fruit N content (%)**	**Crop N removal (kg N ha^{-1})**	**N application - crop N removal (kg N ha^{-1})**
0	47 ±11	1.7 ±0.1 ^{b}	66 ±17	-66 ^{e}
75	42 ±10	2.0 ±.04 ^{a}	67 ±17	8 ^{d}
162	68 ±7	2.0 ±0.1 ^{a}	111 ±14	51 ^{c}
225	45 ±8	2.0 ±0.1 ^{a}	74 ±16	151 ^{b}
300	56 ±7	2.0 ±.04 ^{a}	90 ±13	210 ^{a}
ANOVA	NS	P<0.05	NS	P<0.05

In 2011, yields in the 162, 225, and 300 kg N ha^{-1} treatments did not differ, but they were greater than in the zero-N and 75 kg N ha^{-1} treatments (Table 9). Fruit N content and crop N removal was higher in the 300 kg N ha^{-1} treatment than in any of the other treatments. Fruit N content in the 75 and 162 kg N ha^{-1} treatments were similar. Crop N removal increased with N fertilization levels.

Table 9. Mean yields, tomato (red) fruit N content, and crop N removal in 2011 and standard errors. Values designated by the same letters are not significantly different (P<.05). n=3.

N application rate (kg N ha^{-1})	**Yield ($Mg\ ha^{-1}$)**	**Fruit N content (%)**	**Crop N removal (kg ha^{-1})**	**N application – crop N removal (kg N ha^{-1})**
0	47.7 ^{b}	1.4 ^{c}	58 ^{d}	-58 ^{e}
75	60.9 ^{b}	1.5 ^{bc}	83 ^{d}	-8 ^{d}
162	77.5 ^{a}	1.7 ^{b}	119 ^{c}	43 ^{c}
225	82.3 ^{a}	2.0 ^{a}	149 ^{b}	76 ^{b}
300	92.1 ^{a}	2.1 ^{a}	174 ^{a}	126 ^{a}

3.2.Wheat

Nitrous oxide fluxes were measured from planting through harvest (Figure 12). We also measured N_2O flux several times after harvest, but did not detect any significant emissions. In the 2009/10 growing season, the total annual N_2O emissions in the treatment that received 112 and 91 kg N ha^{-1} as anhydrous ammonia and urea, respectively, were >twice as large and significantly greater than in any of the other treatments (1.31 ± 0.35 kg N_2O-N ha^{-1})(Figure 13). The N_2O emissions during the period between starter and side dress N applications were at least 5 times higher in the AA than in any of the AS treatments (0.78 ±0.30 vs. 0.15 ±0.03 kg N_2O-N ha^{-1}), whereas emissions following the urea application until harvest were similar among the three highest N application treatments, ranging from 0.34 ±0.10 to 0.53 ±0.05 kg N_2O-N ha^{-1} (Figure 14).

Both NH_4^+ and NO_3^- levels increased in response to the fertilizer applications and subsequently declined to near zero levels (Figures 15 & 16). The WFPS was below 60% until mid-December, and stayed above 60% until mid-March when it declined to 40% (Figure 17). The rainfall pattern is shown in Figure 18. On April 4, 2010, the field was flood-irrigated (not shown in Figure 18) and subsequently, the WFPS was >60% for about two weeks before declining for the remainder of the season. Yields did not differ among the treatments (Table 10). However, N content was significantly greater in the grain of the three highest N application plots than that of the 91 kg N ha^{-1} and zero-N plots. Nitrogen removal by the harvested crop was significantly greater in the 203 kg N ha^{-1} (AA + urea) than in any of the other treatments.

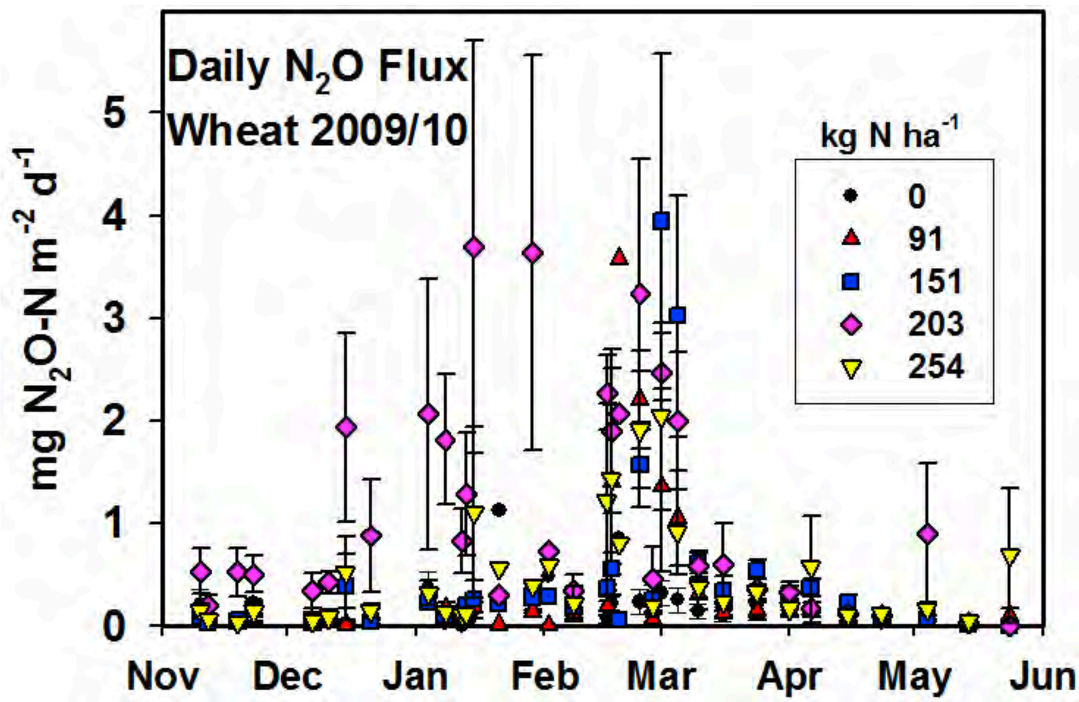


Figure 12. Mean daily N_2O fluxes and standard errors in wheat system fertilized at five different N rates in 2009/10. The fertilizer was applied at planting and as a side dress in early February. More details about fertilizer application is given in text and Figure 14.

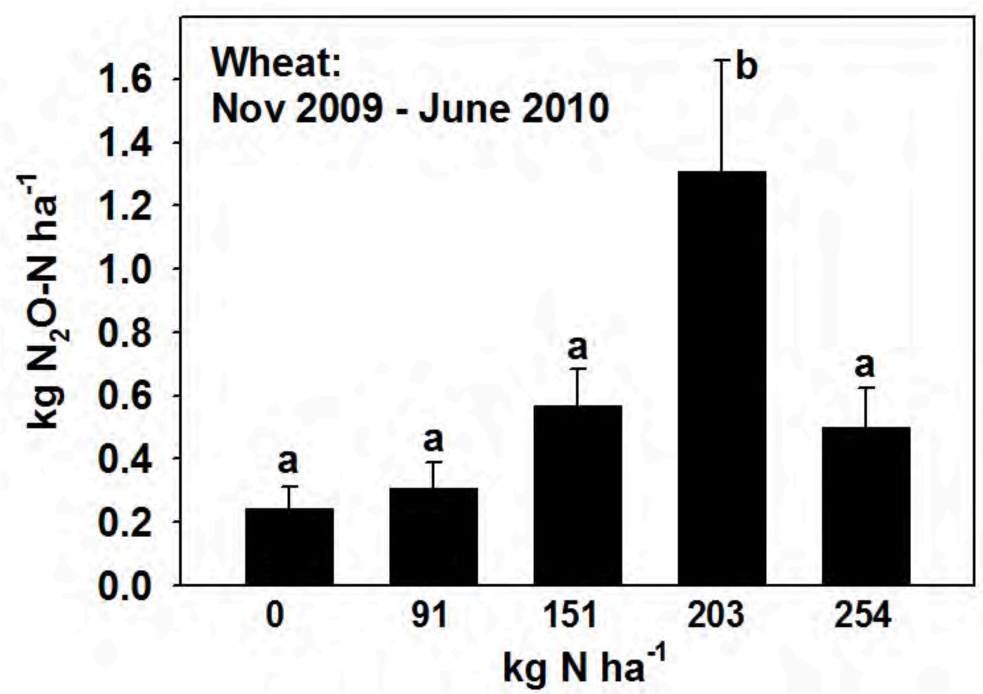


Figure 13. Mean annual N_2O emissions and standard errors in wheat systems fertilized at five different N rates in 2009/10. Bars designated with the same letters are not different from each other.

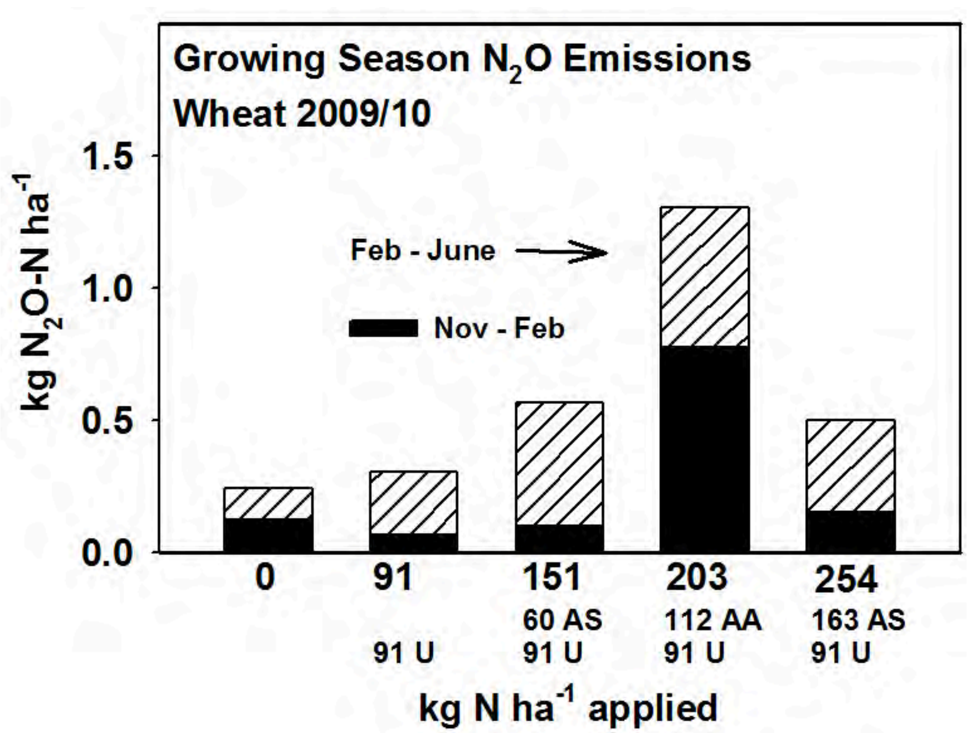


Figure 14. Nitrous oxide emissions from planting to side dress application (Nov – Feb), and from side dress application until harvest (Feb – June). The side dress N was aerially applied as urea (U). At planting, either ammonium sulfate (AS) or anhydrous ammonia (AA) was applied, whereas two treatments did not receive any fertilizer.

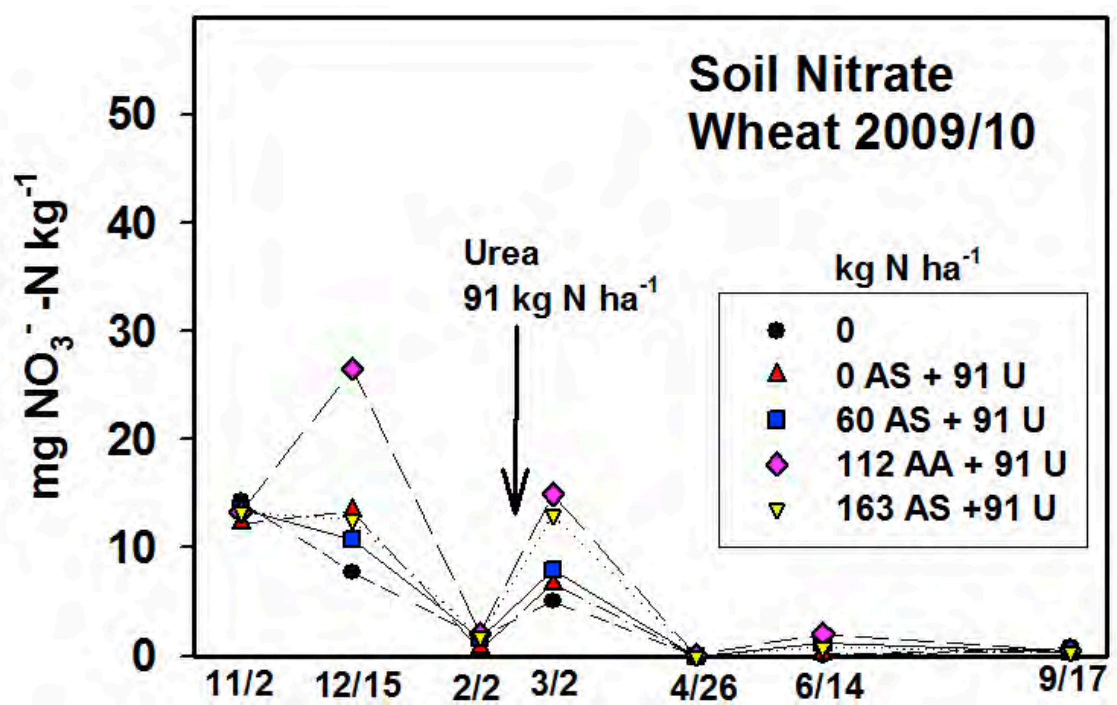


Figure 15. Soil NO_3^- concentrations in the 0-15 cm layer during the 2009/10 wheat growing season.

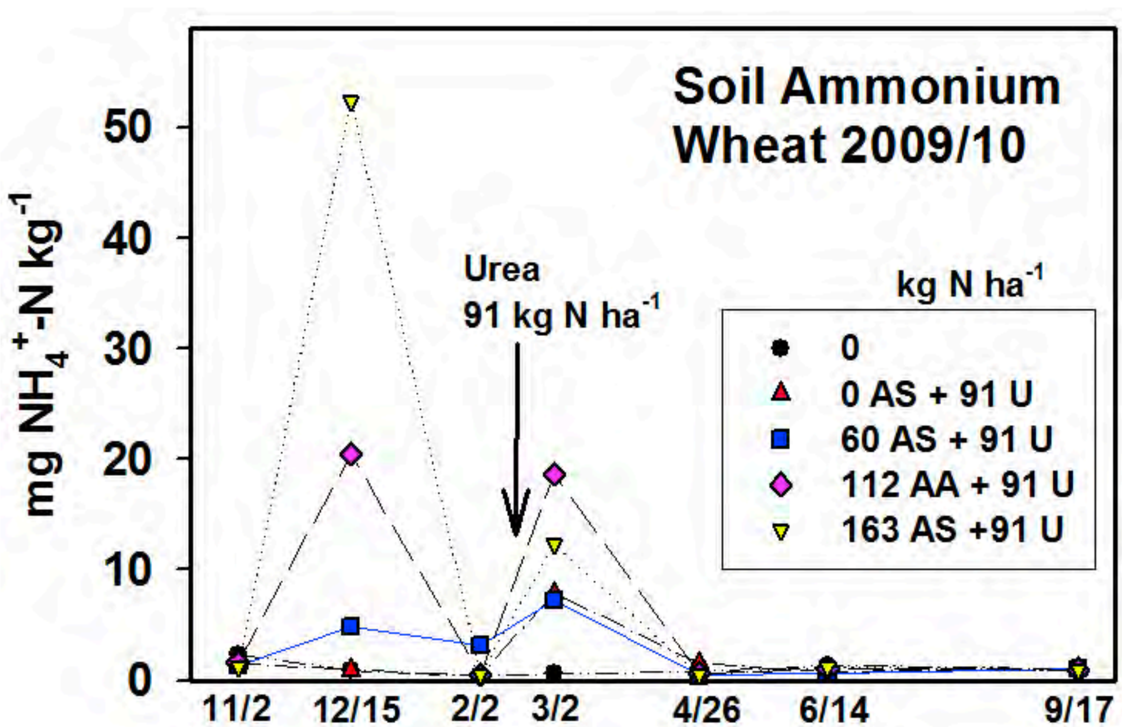


Figure 16. Soil NH_4^+ concentrations in the 0-15 cm layer during the 2009/10 wheat growing season.

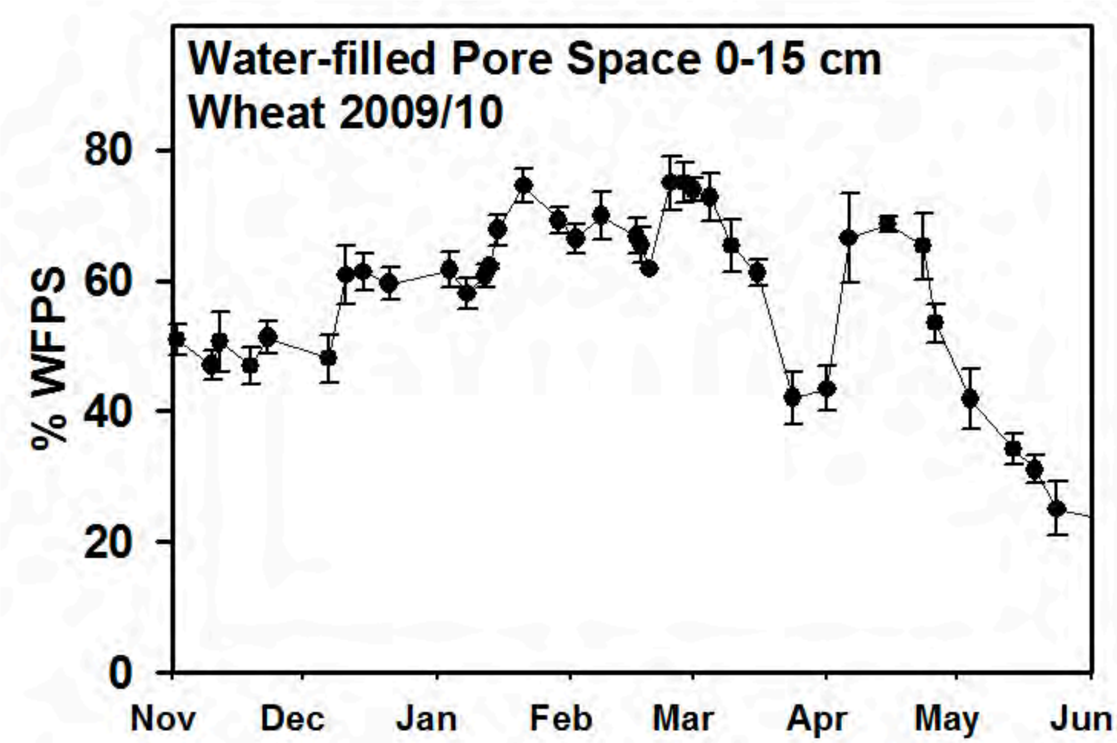


Figure 17. Water-filled pore space in the 0-15 cm layer during the 2009/10 wheat growing season.

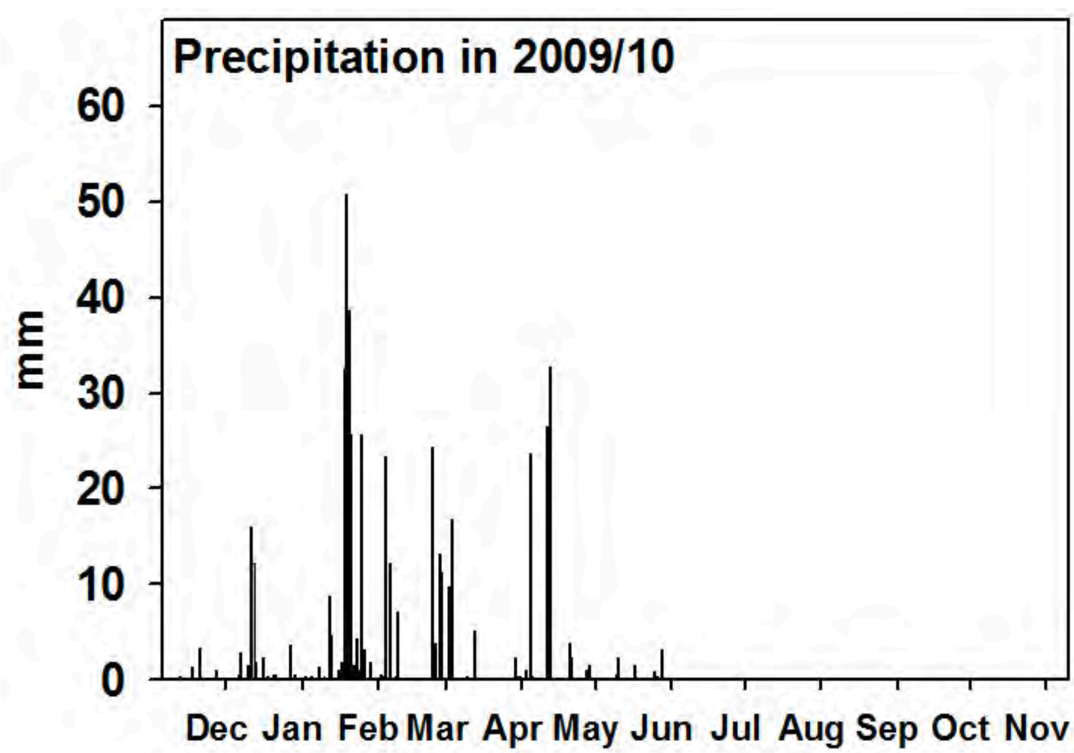


Figure 18. Daily rainfall during the 2009/10 wheat growing season.

Table 10. Mean wheat yield, grain N content, and crop N removal and standard error in 2010. Values designated by the same letters are not significantly different ($P<0.05$). NS = non significant. n=3.

N application (kg N ha^{-1})	Yield (Mg ha^{-1})	Grain N content (%)	Crop N removal (kg N ha^{-1})
0	6.0 ±0.7	1.5 ±.04 ^{b}	91 ± 1 ^{b}
91	6.0 ±0.7	1.5 ±0.1 ^{b}	89 ±15 ^{b}
151	6.6 ±0.7	1.9 ±0.1 ^{a}	122 ±14 ^{b}
203	8.6 ±0.8	2.0 ±0.1 ^{a}	167 ±12 ^{a}
254	6.0 ±0.8	2.0 ±.02 ^{a}	120 ±14 ^{b}
ANOVA	NS	P<0.05	P<0.05

In the 2010/11 growing season, two of the treatments had the same side dress (98 kg urea N ha^{-1}) and total N application rates (210 kg N ha^{-1}), but the starter N fertilizer (112 kg N ha^{-1}) was applied as either broadcast AS or injected AA. Daily N_2O fluxes are shown in Figure 19.

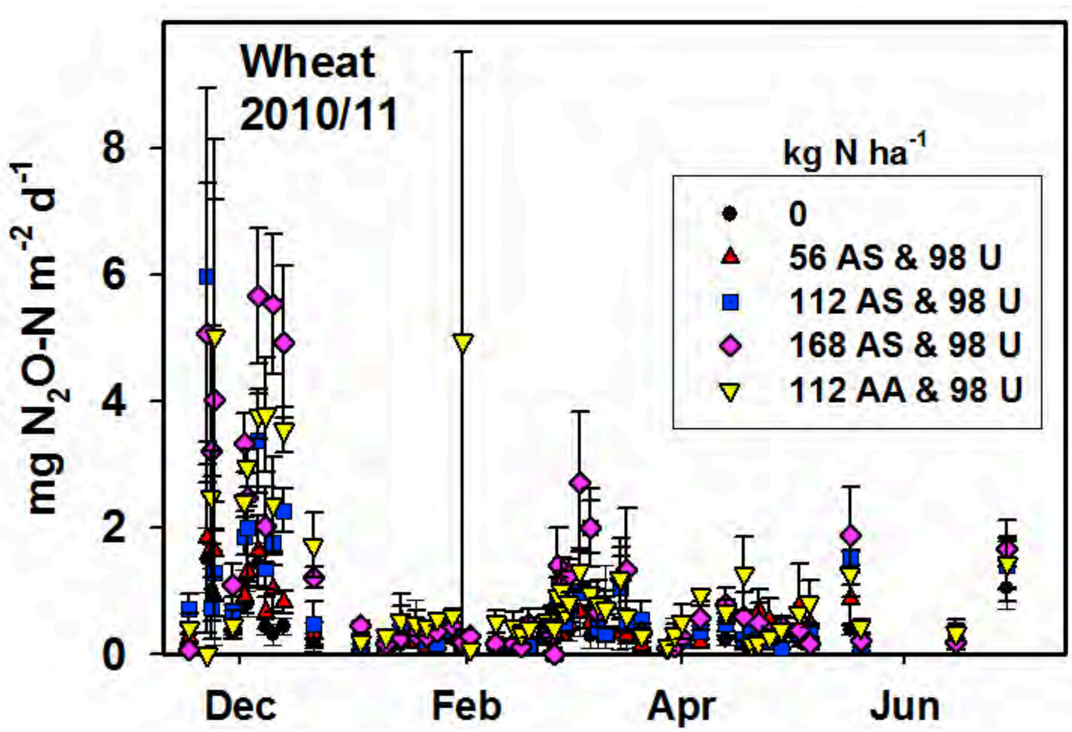


Figure 19. Mean daily N_2O emissions and standard errors in wheat systems fertilized at five different N rates. The starter fertilizer was applied as either broadcast ammonium sulfate (AS) or anhydrous ammonia (AA). The aerial side dress application was in the form of urea (U).

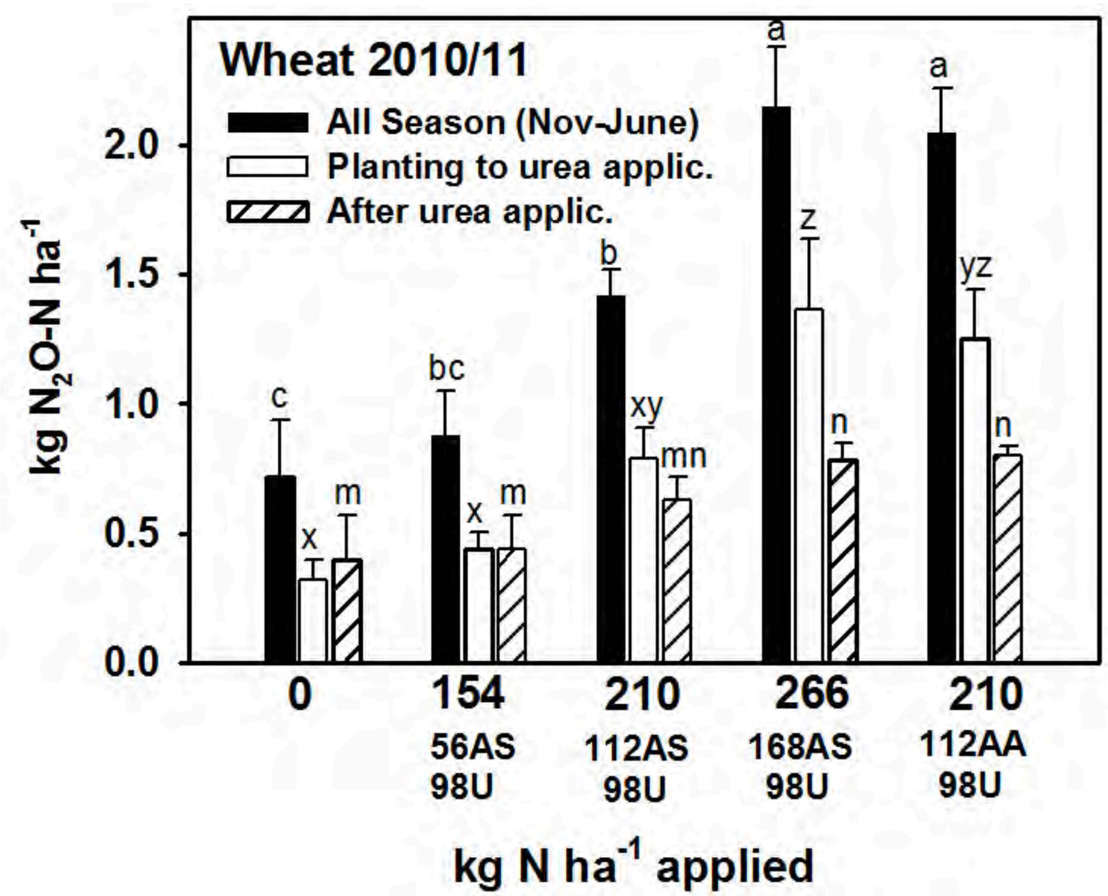


Figure 20. Mean N_2O emissions and standard errors during the entire wheat growing season and during the planting to side dress and the side dress to harvest periods in 2010/11. The starter fertilizer was applied as either broadcast ammonium sulfate (AS) or anhydrous ammonia (AA). The aerial side dress application was in the form of urea. Bars designated with the same letter are not different from each other. (P<.05). n=3.

During the entire wheat growing season, N_2O emissions were highest in the 266 kg N ha^{-1} treatment (starter 168 kg N ha^{-1} as AS and 98 kg as urea N) (2.15 ±0.23 kg N_2O-N ha^{-1}) and in the 210 kg N treatment that had received AA as starter N fertilizer (2.05 ±0.17 kg N_2O-N ha^{-1}) (Figure 20). During the period from starter to side dress N applications, N_2O emissions did not statistically differ between the two treatments that differed with regard to fertilizer type (112 kg N ha^{-1}). Mean N_2O emissions in the 210 kg N ha^{-1} treatment in which the starter N was applied as AS were 30% lower (1.42 ±0.10 kg N_2O-N ha^{-1}) and not different from those of the 154 kg N ha^{-1} treatment (0.88 ±0.18 kg N_2O-N ha^{-1}). The emissions after the urea application were similar among the three highest N application treatments (0.63 ±0.09 to 0.80 ±0.40 kg N_2O-N ha^{-1}) and also among the two lower N rate treatments that used broadcast AS as starter (154 and 210 kg N ha^{-1}) and the zero-N treatment (0.40 ±0.17 to 0.63 ±0.09 kg N_2O-N ha^{-1}).

Soil nitrate and ammonium reached high levels after the starter application (>250 kg N ha^{-1} in the 0-15 cm alone) (Figures 21 & 22). After the urea application, NO_3^- levels were elevated for more than a month in the 210 kg N ha^{-1} (AA starter) treatment. The precipitation pattern is shown in Figure 23. In the beds, the WFPS was >60% on most sampling days from December until March, and in the furrows the WFPS was about 10% greater during this time (Figure 24). The irrigation on April 18-19 (not shown in Figure 23) >briefly raised the WFPS before it declined to 40% for the remainder of the season.

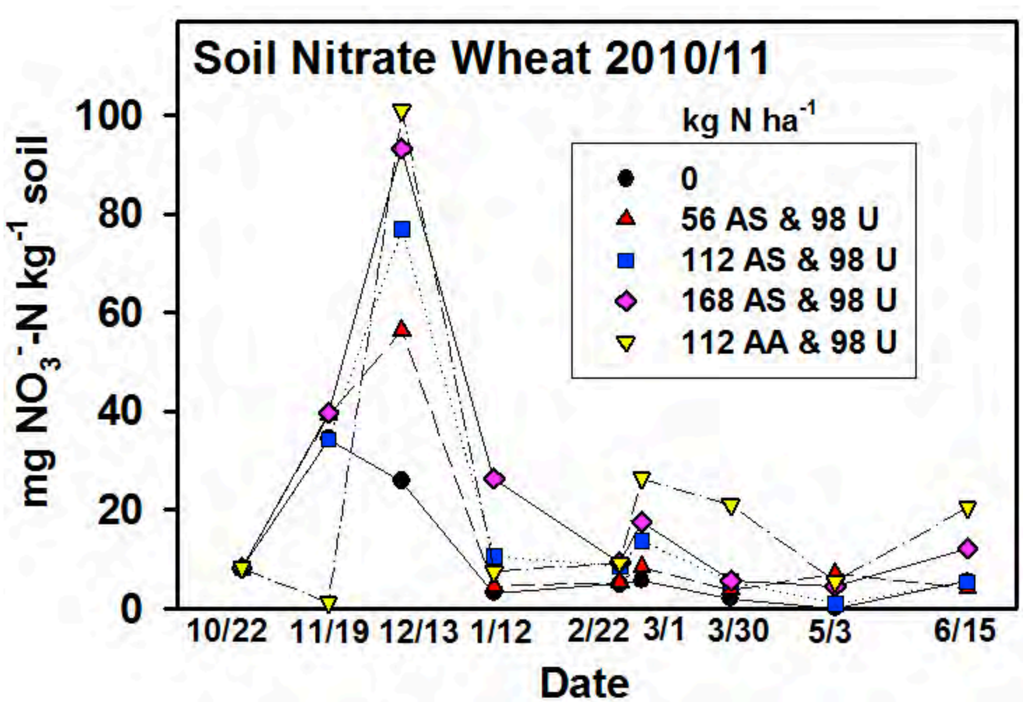


Figure 21. Soil NO_3^- concentrations in the 0-15 cm layer during the 2010/11 wheat growing season.

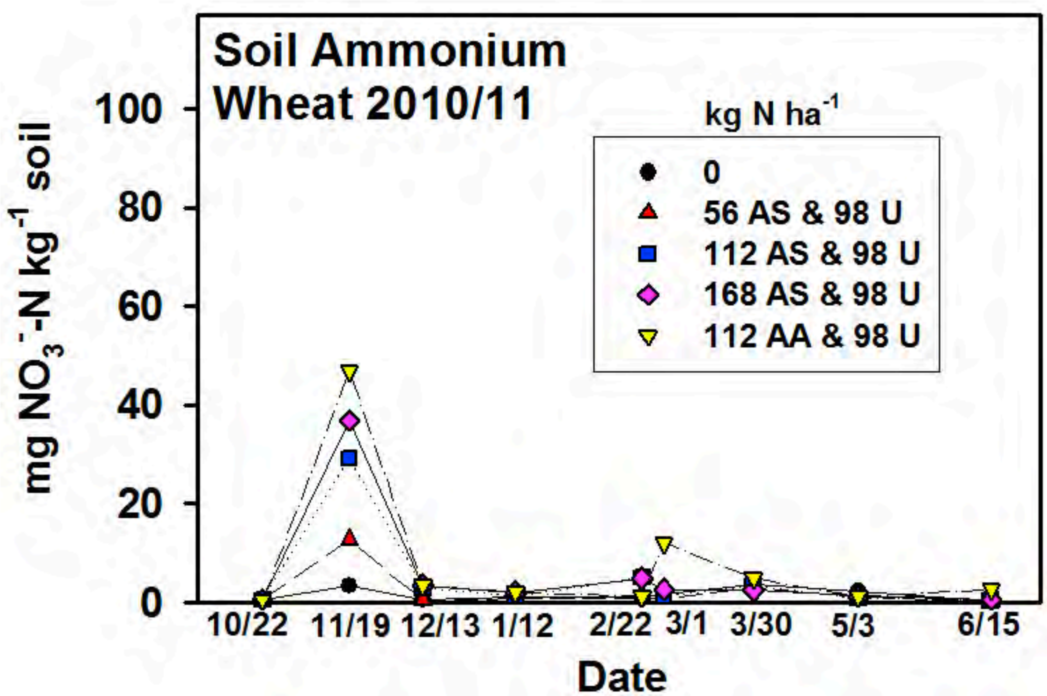


Figure 22. Soil NH_4^+ concentrations in the 0-15 cm layer during the 2010/11 wheat growing season.

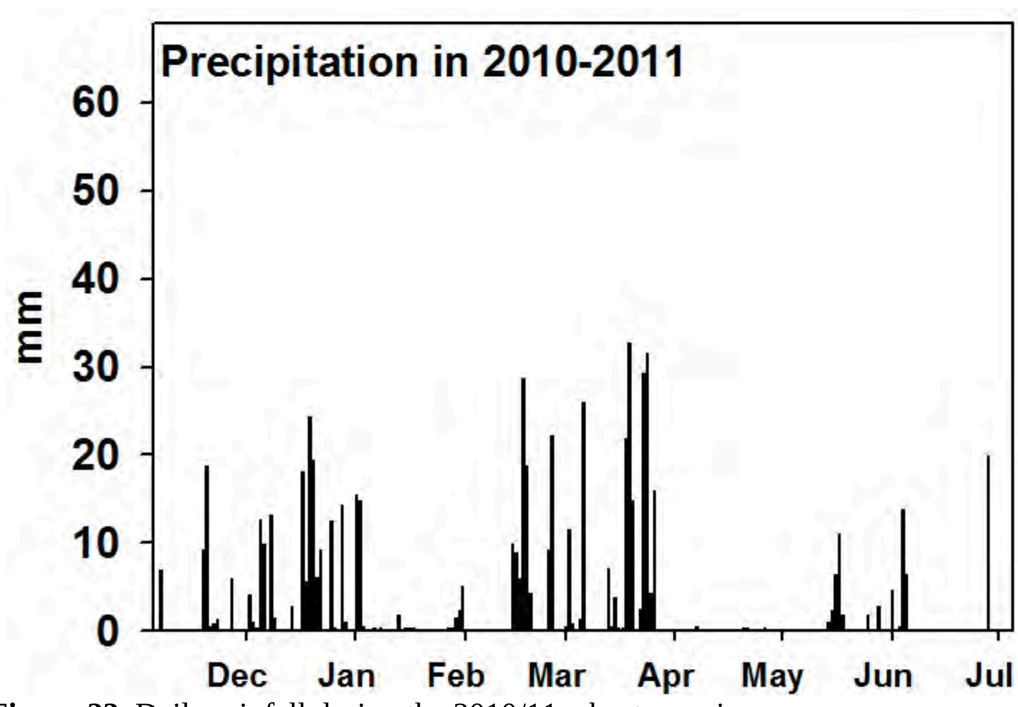


Figure 23. Daily rainfall during the 2010/11 wheat growing season.

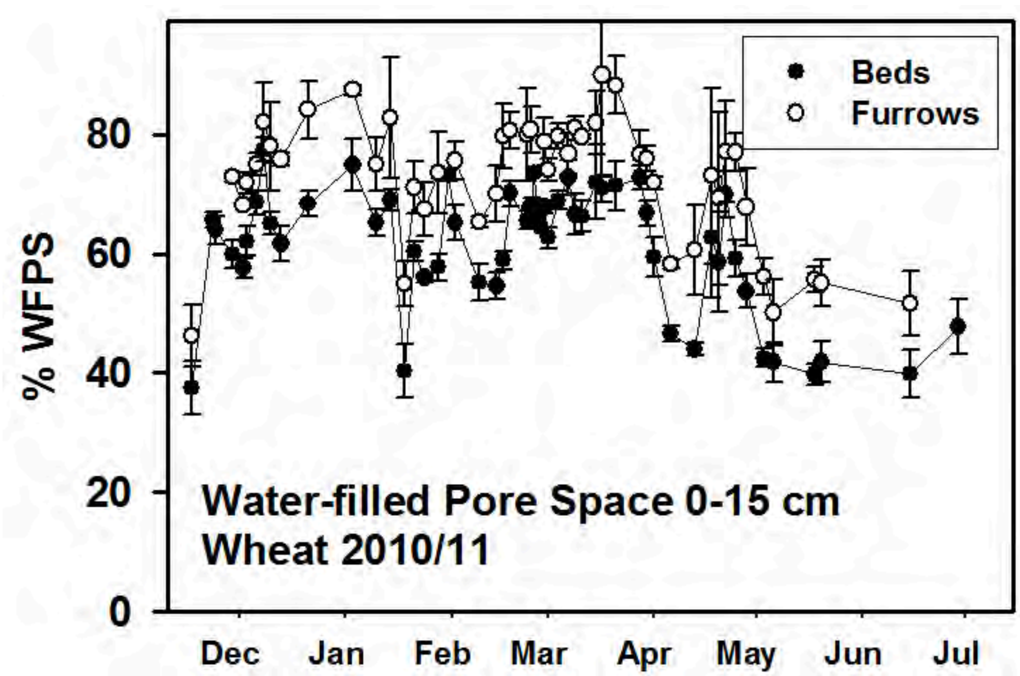


Figure 24. Water-filled pore space in the 0-15 cm layer of beds and furrows during the 2010/11 wheat growing season.

Yields did not differ among N treatments in 2011 and the grain N content was similar among all the fertilized treatments (Table 11). Crop N export was nearly as high or greater than the N application rates. Crop N removal did not differ among the three highest N application treatments.

Table 11. Mean wheat yield, grain N content, and crop N removal and standard error in 2011. Values designated by the same letters are not significantly different (p<.05). NS = non significant. n=3.

N application (kg N ha^{-1})	**Yield (Mg ha^{-1})**	**Grain N content (%)**	**Crop N removal (kg N ha^{-1})**
0	9.3 ±0.4	1.6 ±0.1 ^{b}	147 ± 4 ^{c}
154	10.3 ±0.7	1.9 ±0.03 ^{a}	194 ±10 ^{b}
210 (AS+U)	9.6 ±0.4	2.1 ±0.1 ^{a}	202 ±4 ^{ab}
266	10.4 ±0.7	2.1 ±0.1 ^{a}	220 ±22 ^{ab}
210 (AA+U)	11.1 ±0.1	2.1 ±.04 ^{a}	233 ±4 ^{a}
ANOVA	NS	P<0.05	P<0.05

Following the trend of the annual N_2O emissions, the EFs tended to be higher in the second than the first year, ranging from 0.20% (254 kg applied N ha^{-1}) to 0.64% (203 kg N ha^{-1}) in 2009/10 and from 0.57% (154 kg N ha^{-1}) to 0.98% (210 kg AA-N ha^{-1}) in 2010/11 (Table 7).

3.3. Alfalfa

According to the WFPS values, the soils reached saturation (100% WFPS) with each irrigation event (Figure 25). Nitrous oxide fluxes in the alfalfa systems were characterized by generally low daily fluxes, punctuated by very high N_2O fluxes immediately following each check flood irrigation event (Figure 26). These high fluxes lasted only for one day following each irrigation event. In the second growing season (April – October 2011), we captured the high N_2O emissions with every irrigation event by entering the fields while the tail end of the fields was still submerged.

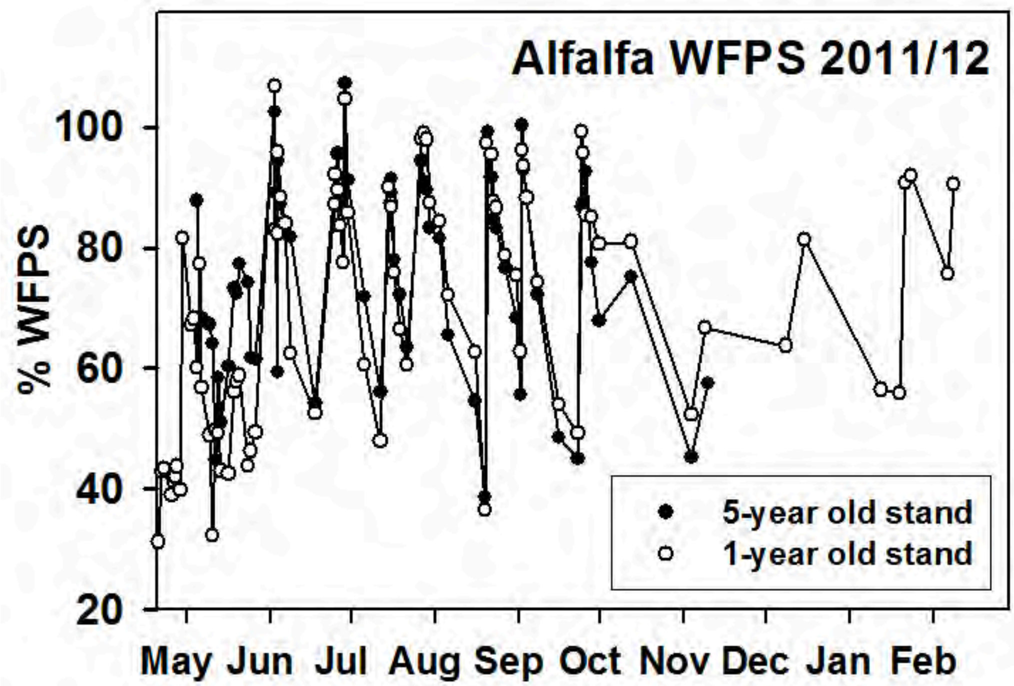


Figure 25. Water-filled pore space in the 0-15 cm soil layer in the 5-year and 1-year old alfalfa system during 2011/12.

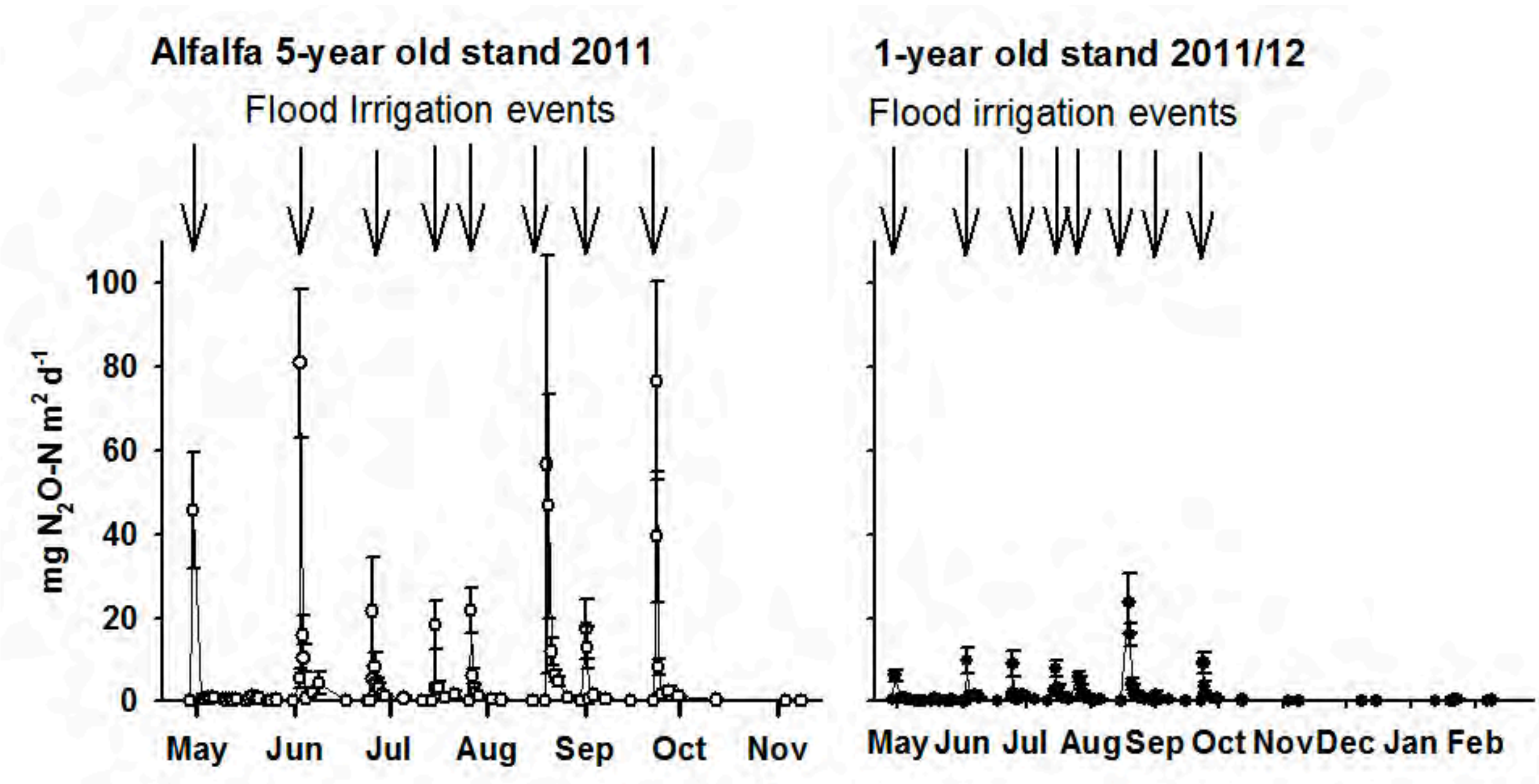


Figure 26. Mean daily N_2O emissions and standard error (n=6) in a 5-year and a 1-year old alfalfa system during the 2011 growing season. The spikes of high N_2O emissions occurred with every one of the 8 irrigation events.

We did not detect the high fluxes after irrigation during the first alfalfa growing season (April – October 2010) because in summer 2010 we did not enter the fields when a part of the field was submerged (Figure 27).

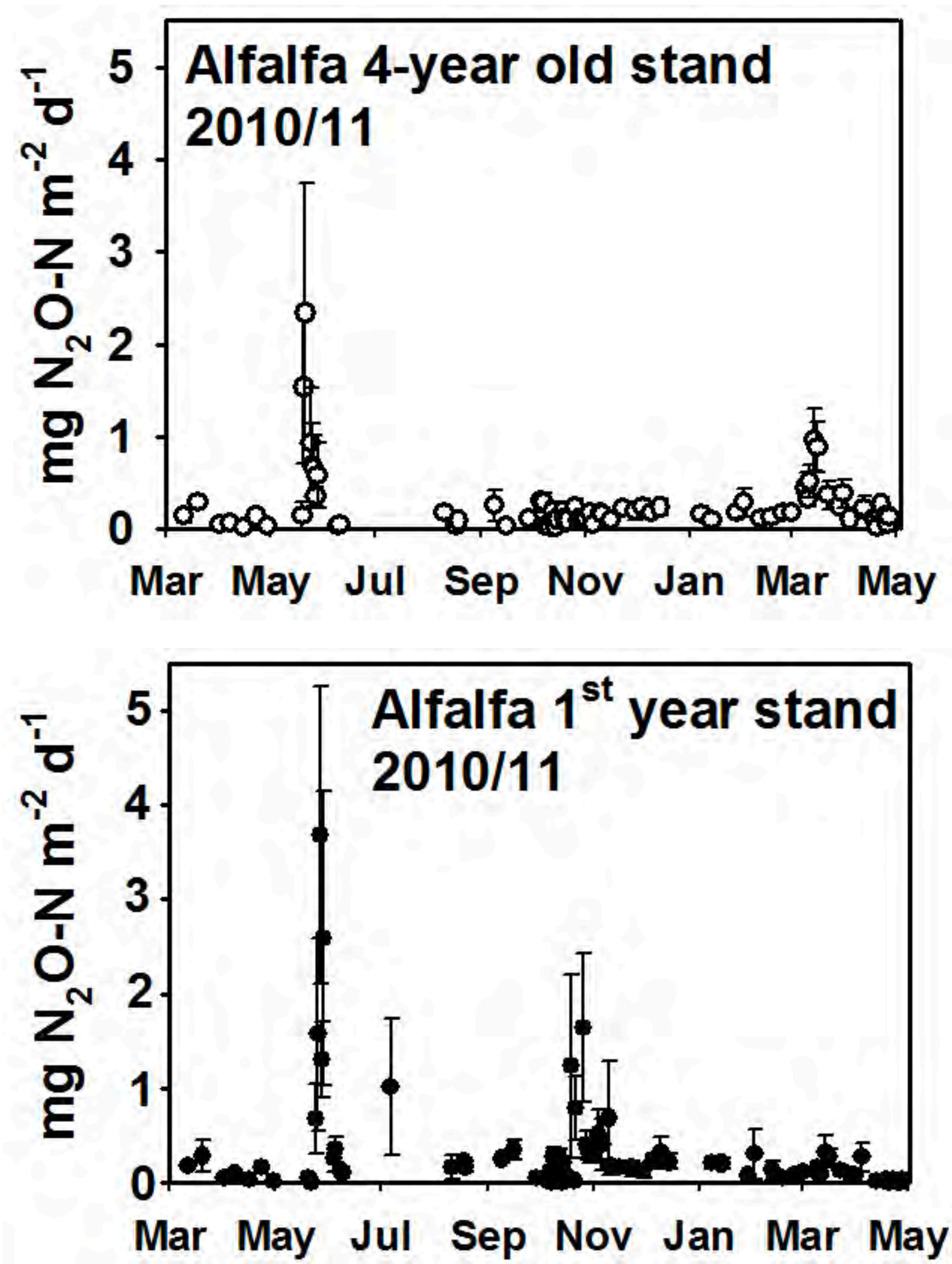


Figure 27. Mean daily N_2O emissions and standard error (n=6) in the 4-year old and the newly established (1st year) alfalfa system during the 2010/11 growing season. During this period, N_2O fluxes were measured frequently during the winter rainy season (Nov-April).

During the winter 2010/11, the N_2O fluxes did not reach the levels of those after check flood irrigation (Figure 27). The WFPS during winter 2010/11 was around 80% for extended periods (Figure 28).

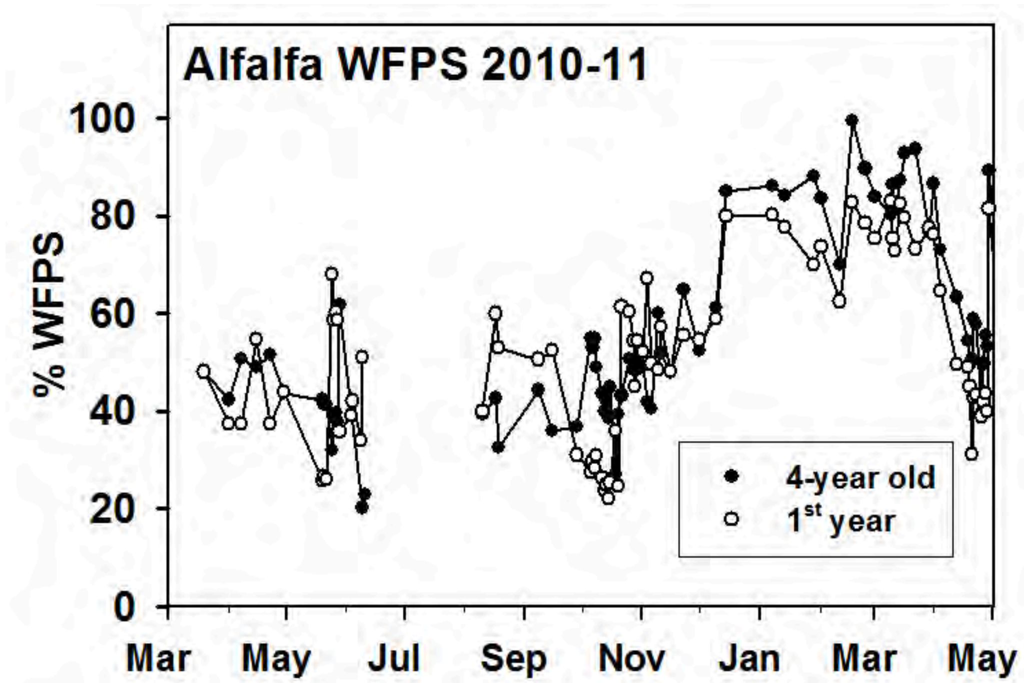


Figure 28. Water-filled pore space in the 4-year old and the newly established alfalfa system during 2010/11.

The annual N_2O emission estimate is based on the 2011 growing season emissions and the time-integrated emissions of the Nov 2010 to April 2011 winter season (Table 12). Additionally, Table 12 shows the total N_2O emissions from Nov 2011 to Feb 2012. The annual N_2O emissions were significantly greater and >twice as high in the 5-year old than in the 1-year old field.

Table 12. Time-integrated N_2O emissions in a 5-year and a 1-year old alfalfa system. The annual N_2O emissions estimate includes the time-integrated N_2O emissions from November 2010 to November 2011. * The annual N_2O emissions were significantly different in the two fields ($p<.05$). n=6.

Age of stand	Nov 2010-April 2011	April 2011 – Nov 2011	Nov 2011 – Feb 2012	Annual emissions *
	kg N_2O-N ha^{-1}			
5 years	0.3 ±0.04	4.9 ±0.8	n.d.[1]	5.2 ±0.8
1 year	0.2 ±0.05	1.8 ±0.2	0.16 ±0.02	2.0 ±0.2

[1] The 5-year old alfalfa field was plowed in November 2011 and planted to another crop.

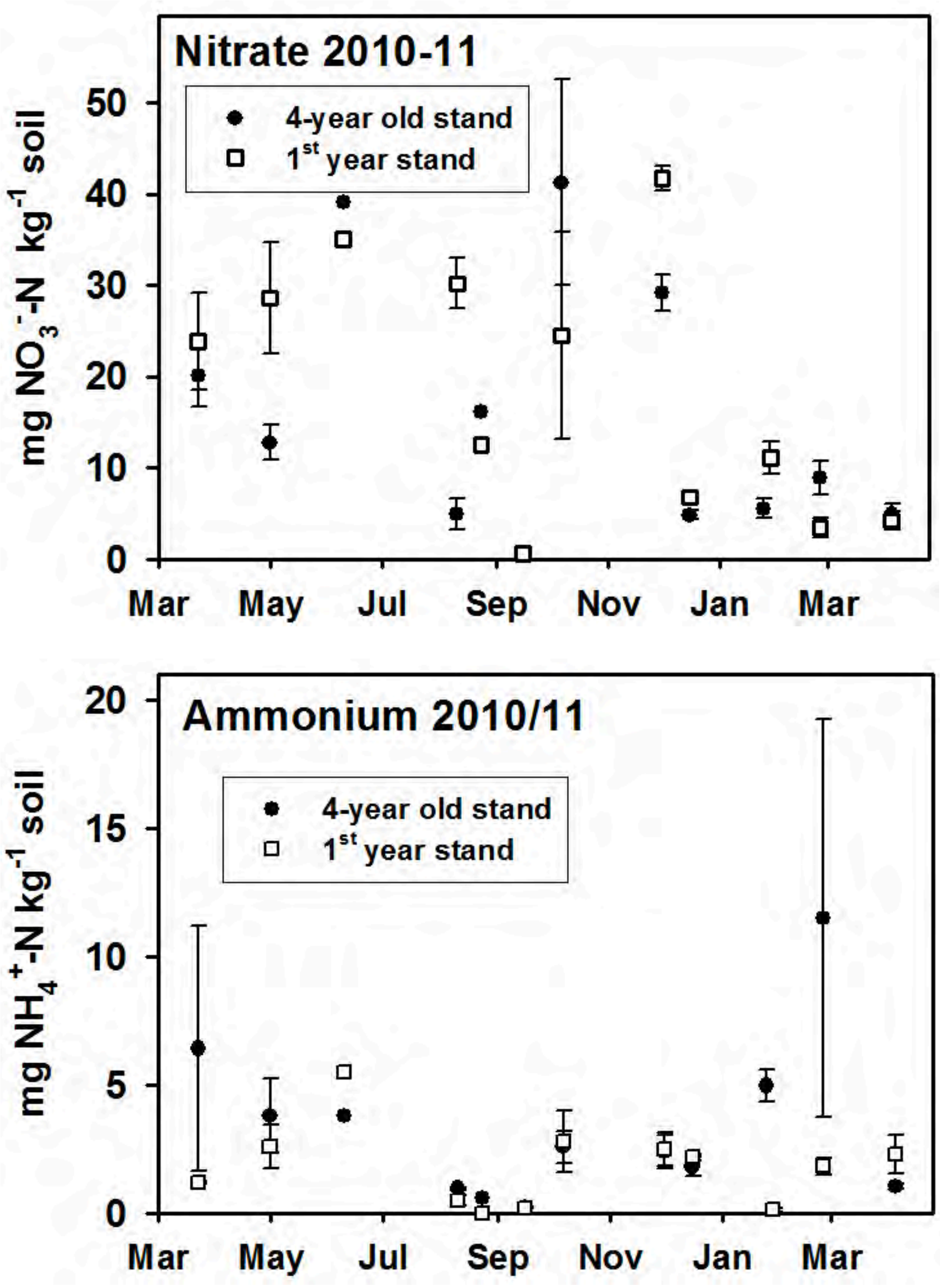
The yields and alfalfa N removal by the harvested crop were similar in the two fields (Table 13). We report yields for both years and mean N content of all cuttings and crop N removal for 2011. Alfalfa N content in 2010 was not measured. In 2010, there were 5 cuttings in the 1-year old and 6 cuttings in the 4-year old system; in 2011, there were six cuttings in both fields.

Table 13. Alfalfa yields, mean N content in dry alfalfa biomass in 2011, and crop N removal in 2011.

Stand age	Yield 2010 (Mg ha^{-1})	Yield 2011 (Mg ha^{-1})	N content (%) 2011	N removal (kg N ha^{-1}) 2011
5 years	19.3	14.7	3.9	473
1 year	15.5	17.3	3.7	525

The yield data and the measured N content of the hay were used as the basis to calculate the N input from the N-fixing alfalfa plants, in addition to the one-time N fertilizer application, in order to calculate the EFs according to the 2006 IPCC guidelines. The EFs were 12.06% and 4.15% for the 5-year and 1-year old fields, respectively (Table 7).

Nitrate and NH_4^+ concentrations were, in general, below 10 mg N kg^{-1} soil, except for the NO_3^- concentrations in summer 2010 (Figure 29). The concentrations of inorganic N were similar between the two fields. Dissolved organic C was generally between 40 and 90 mg C kg^{-1} soil, except for September when mean DOC levels were >100 mg C kg^{-1} (Figure 30). The DOC concentrations were similar in the two fields.



Nitrate 2010-11
4-year old stand
1st year stand
mg NO3- -N kg-1 soil
0
10
20
30
40
50
Mar
May
Jul
Sep
Nov
Jan
Mar
Ammonium 2010/11
4-year old stand
1st year stand
mg NH4+ -N kg-1 soil
0
5
10
15
20
Mar
May
Jul
Sep
Nov
Jan
Mar

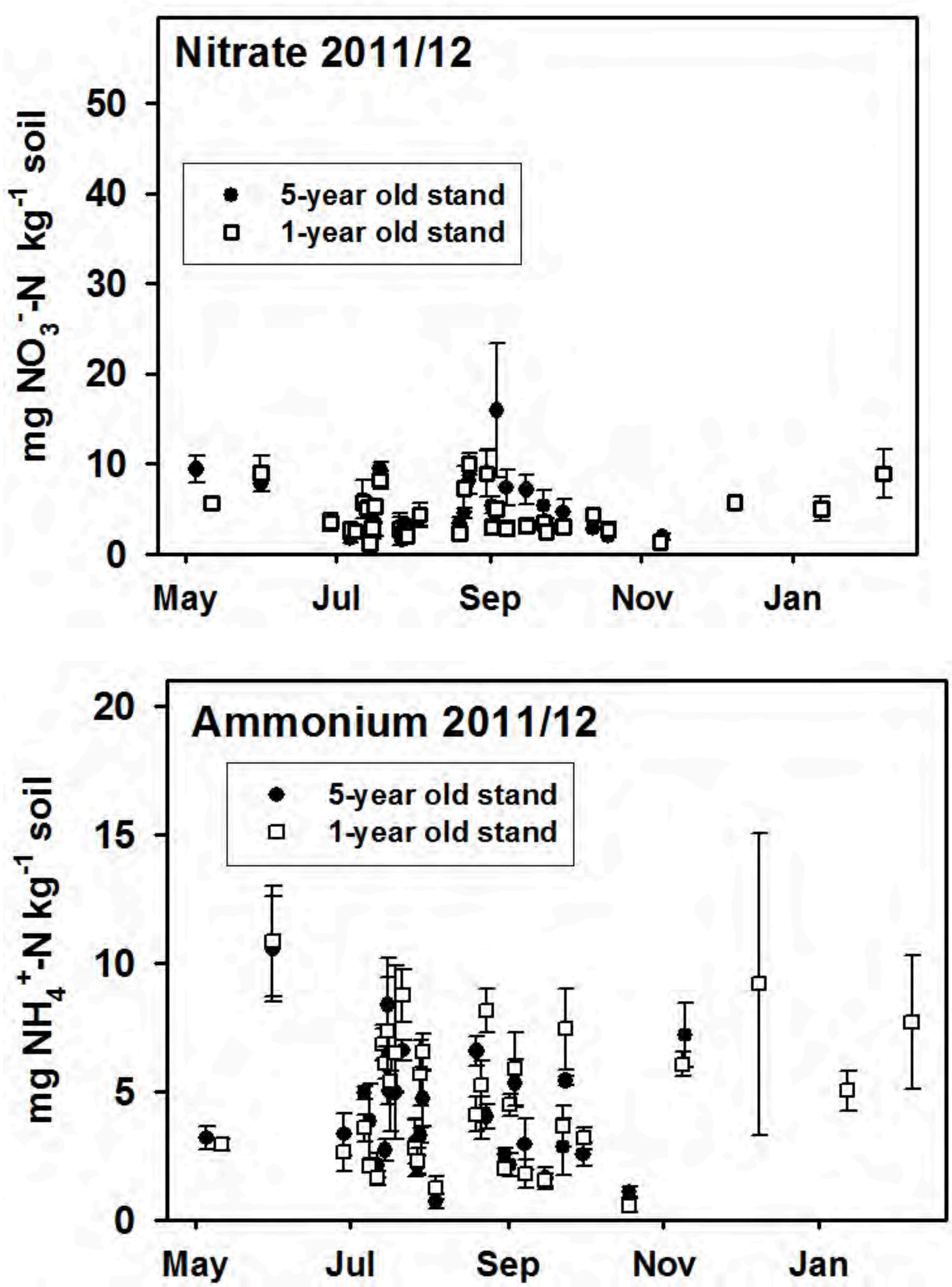


Figure 29. The above four panels show mean soil NO_3^- and NH_4^+ concentrations and standard errors measured during two years (March 2010 Feb 2012) in the 0-15 cm layer in two alfalfa stands of different age.

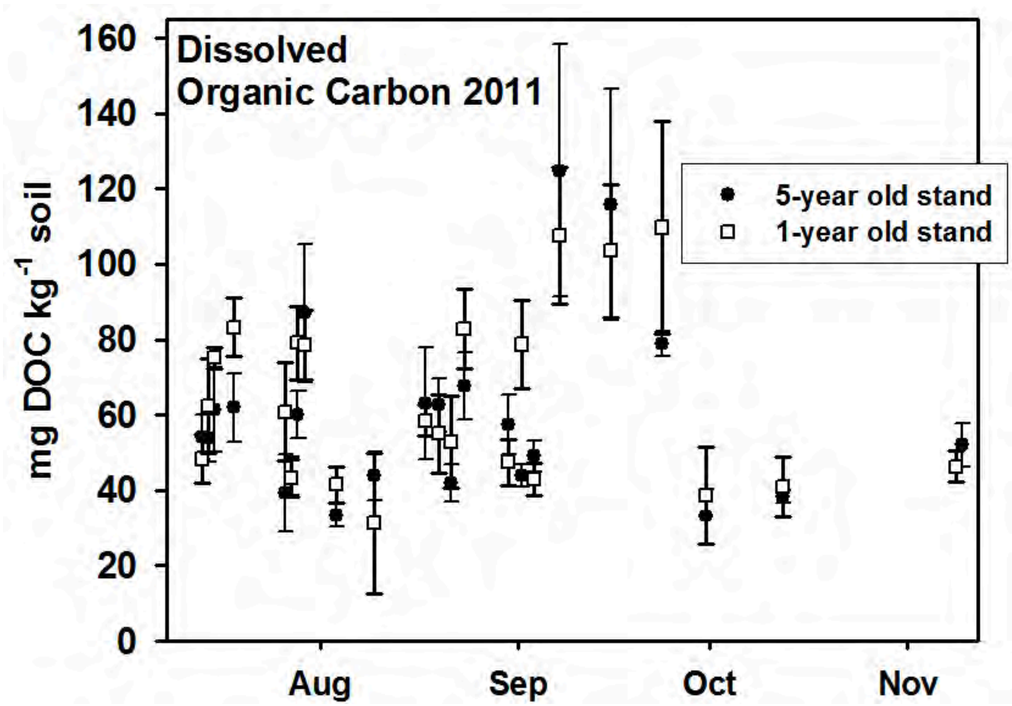


Figure 30. Mean concentrations and standard error (n=6) of dissolved organic carbon, measured in 2011 in the 0-15 cm layer during and in between irrigation events in the 5- and 1-year old alfalfa stand.

<u>*3.4. Lettuce*</u>

In each year, N_2O flux was measured during one cropping cycle in experimental plots fertilized at five different N rates, an irrigation event in fall, and during the winter rainy season and spring (Figure 31). The irrigation event in the fall was comparable to a rainfall event or to an irrigation to germinate the seeds of the next crop. The water inputs as irrigation and precipitation and the resulting soil moisture (WFPS) are shown in Figures 32 & 33. Figure 34 informs about soil and air temperatures during the measurement events.

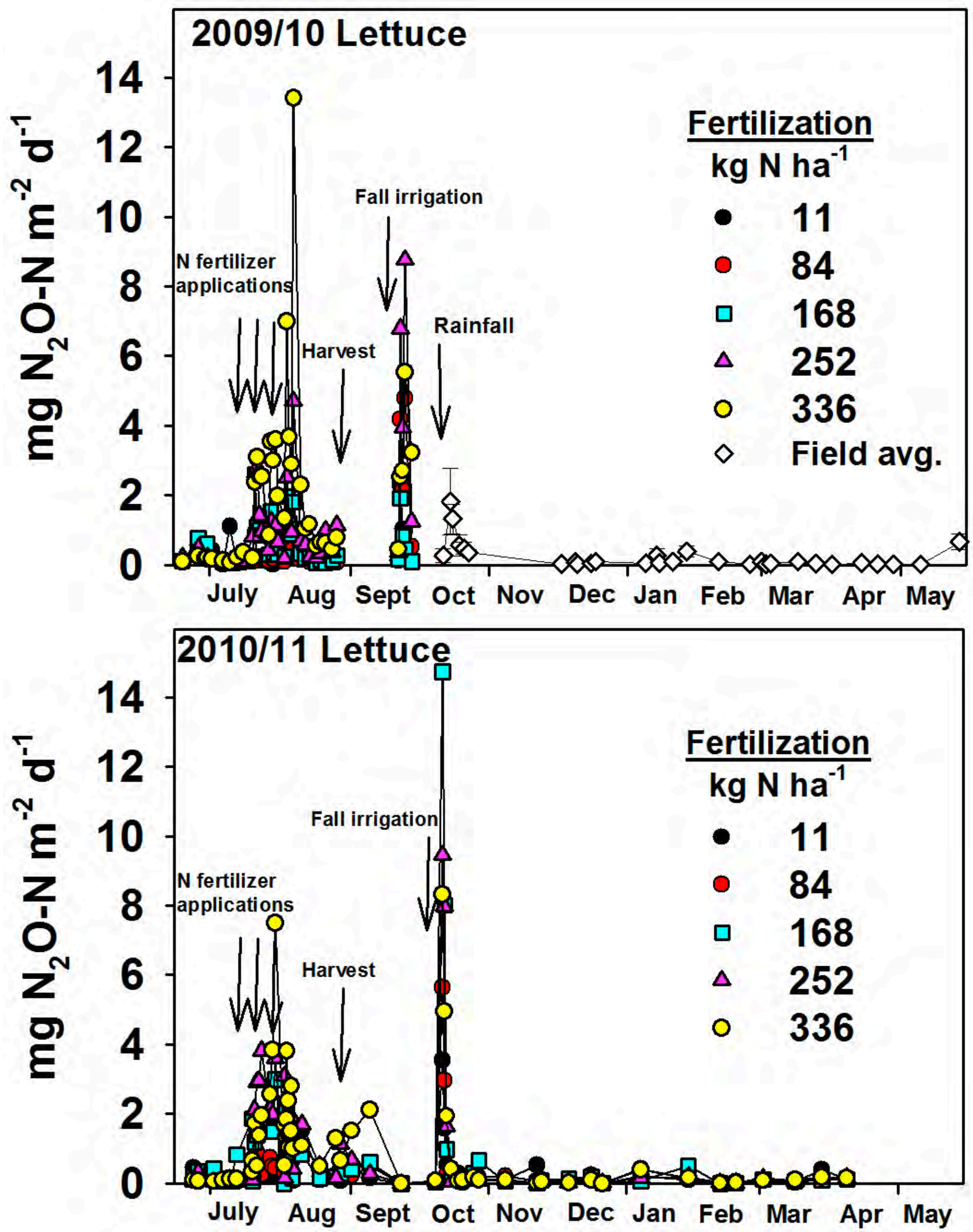


Figure 31. Mean daily N_2O fluxes in lettuce systems fertilized at five different N levels during one cropping cycle, a fall irrigation, and during the rainy season in each of two years (2009/10 & 2010/11). The N fertilizer was applied in the form of urea ammonium nitrate in three increments through the drip irrigation system (fertigation).

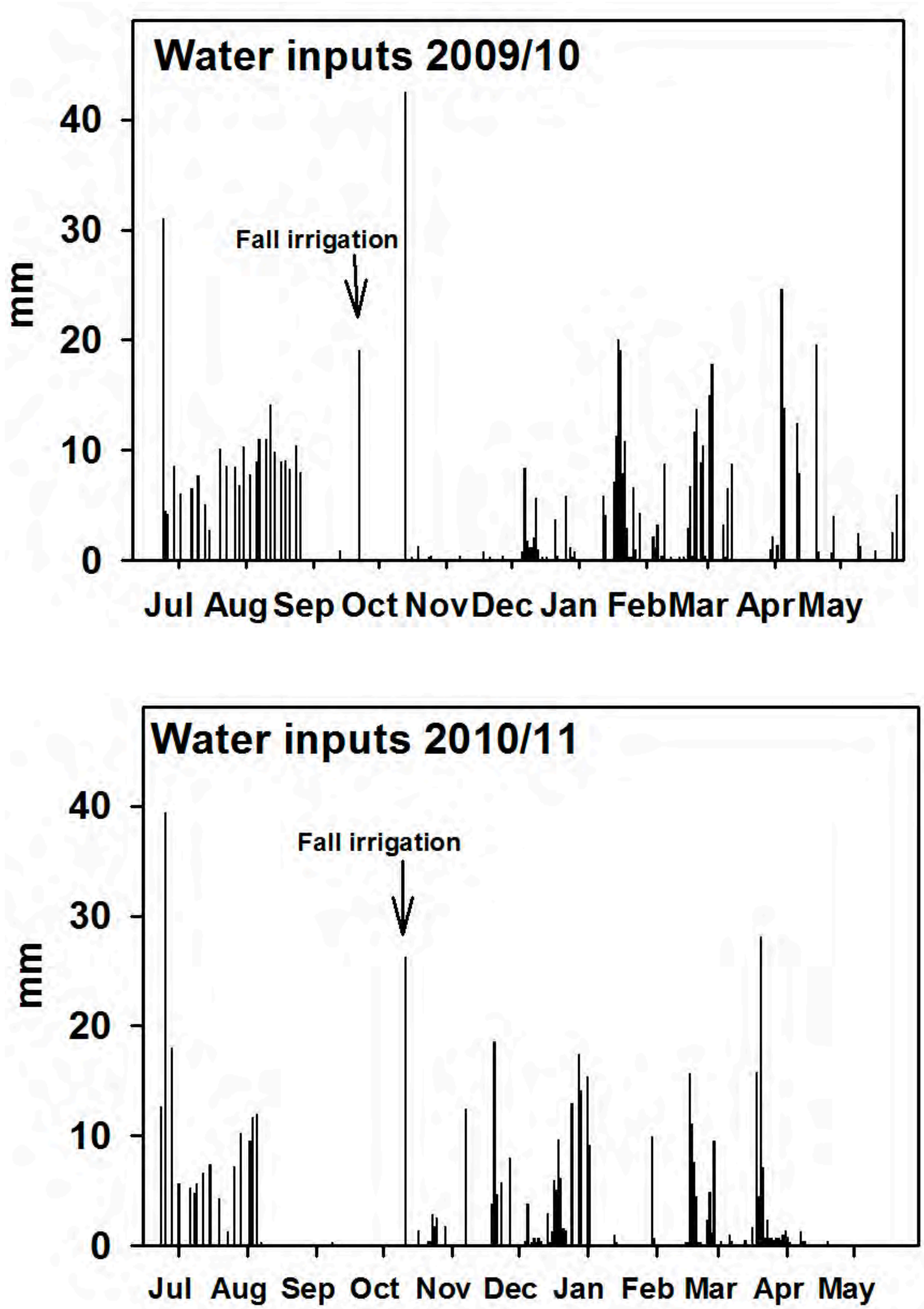
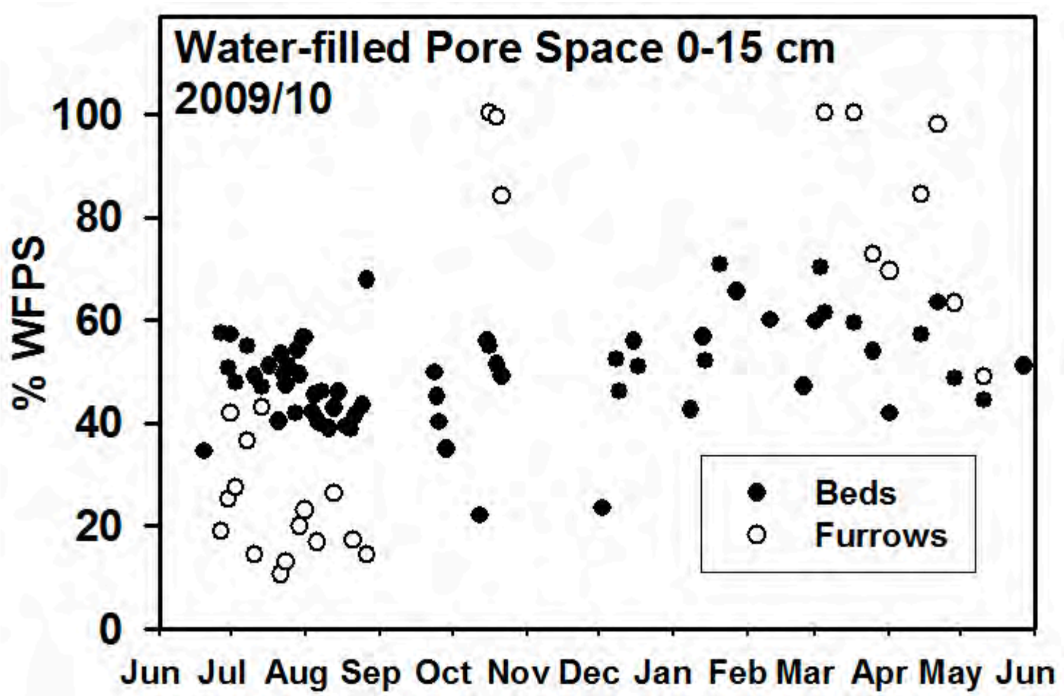


Figure 32. Daily water inputs as irrigation and precipitation in the experimental lettuce field in Salinas from summer 2009 to summer 2011.



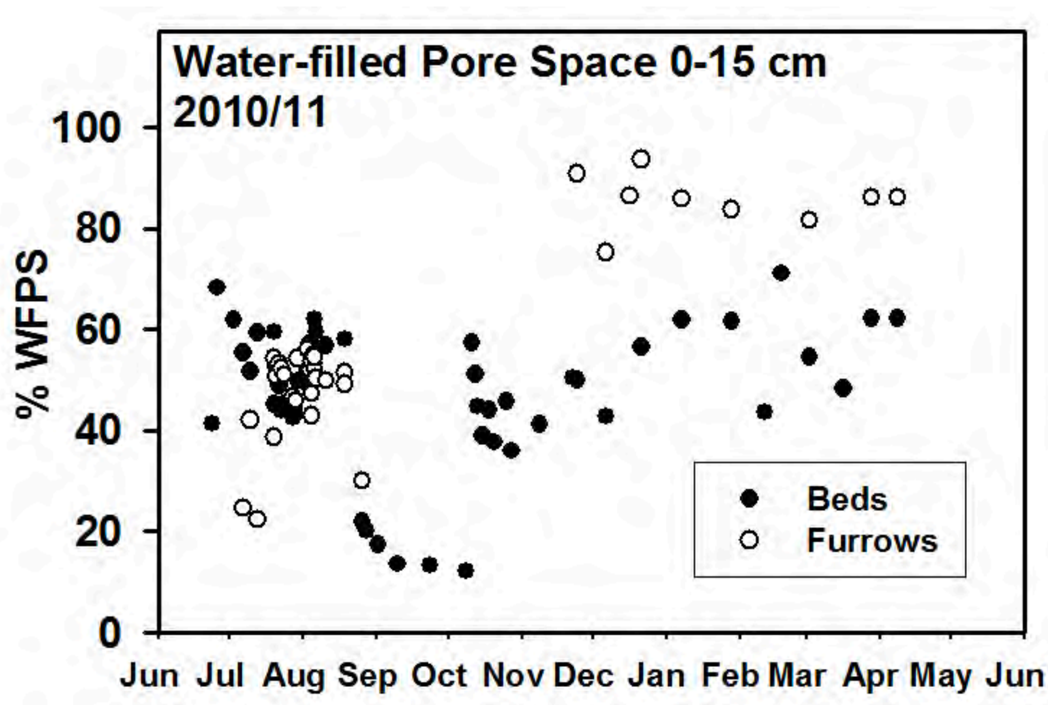
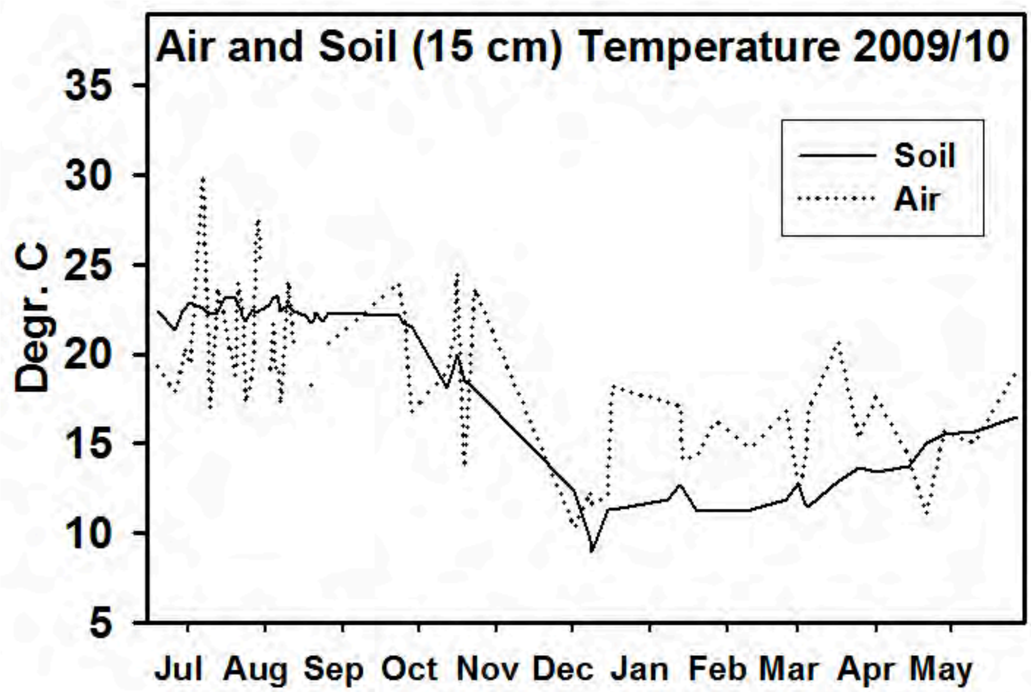


Figure 33. Water-filled pore space in the 0-15 cm layer of beds and furrows in 2010/11.



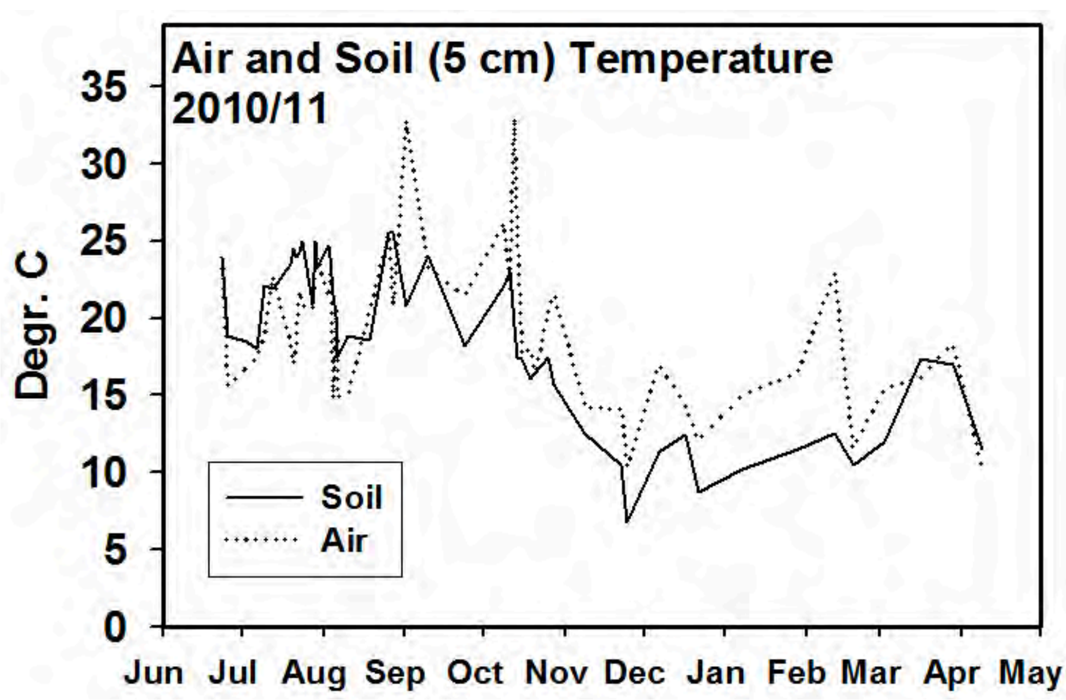


Figure 34. Ambient air and soil temperature in experimental lettuce plots. In 2009/10, soil temperature values (15 cm) recorded at the nearest CIMIS station are reported. In 2010/11, soil temperature was measured at 5 cm whenever chamber air samples were collected.

For year 1, the total N_2O emissions for each fertilizer level includes the N_2O emissions from June to October 2009 and, additionally, the average N_2O emissions in the entire experimental area from November 2009 through May 2010.

The field was ripped in late October, and this disturbance confounded the N fertilizer treatments. After this tillage event, N_2O fluxes were measured at random locations within the entire field. Therefore, we used the average N_2O emissions in the entire field as 2009/10 rainy season estimate for all the treatments (Figure 37). The reported annual N_2O emissions for 2009/10 can be considered a valid estimate because a) in 2010/11, the N_2O emissions in the rainy season did not significantly differ among fertilizer N treatments (statistical results not shown); b) the average N_2O emissions during the winter rainy season (Nov – May) were similar in the two years (0.36 and 0.21 kg N_2O-N ha^{-1} in 2009/10 and 2010/11, respectively); c) the annual N_2O emissions were similar in the two years in all but the 162 kg N ha^{-1} treatment.

In 2009, total N_2O emissions were significantly higher in the 252 and 336 kg N ha^{-1} than in the other treatments. In 2010/11, the total N emitted as N_2O was similar among the three highest N application treatments and significantly higher than in the plots fertilized with 84 and 11 kg N ha^{-1} (Figures 35 & 36). The increase in N_2O emissions with increasing fertilizer levels was linear (same statistical procedure as in tomato, above).

The EFs were similar in both years and within a narrow range (0.41 to 0.84%). In both years, the highest EF (0.84% in 2009/10 and 0.67% in 2010/11) was calculated for the 84 kg N ha^{-1} treatment (Table7).

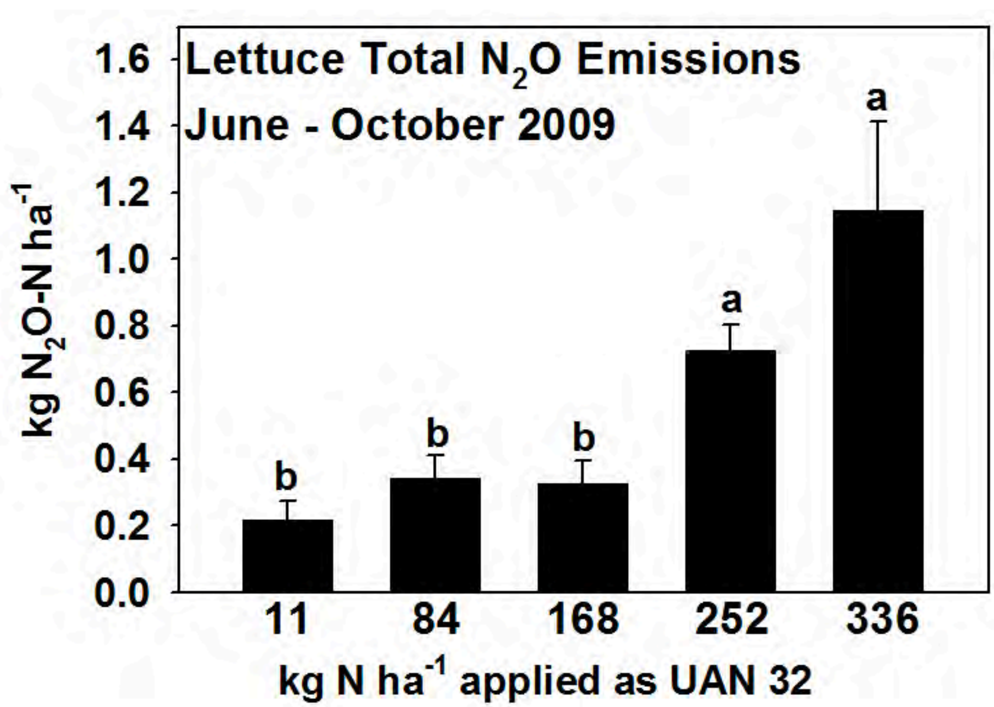


Figure 35. Mean annual N_2O emissions and standard errors during 2009/10 in the five N fertilizer treatments. Means designated with the same letter are not different from each other (p<.05). n=4.

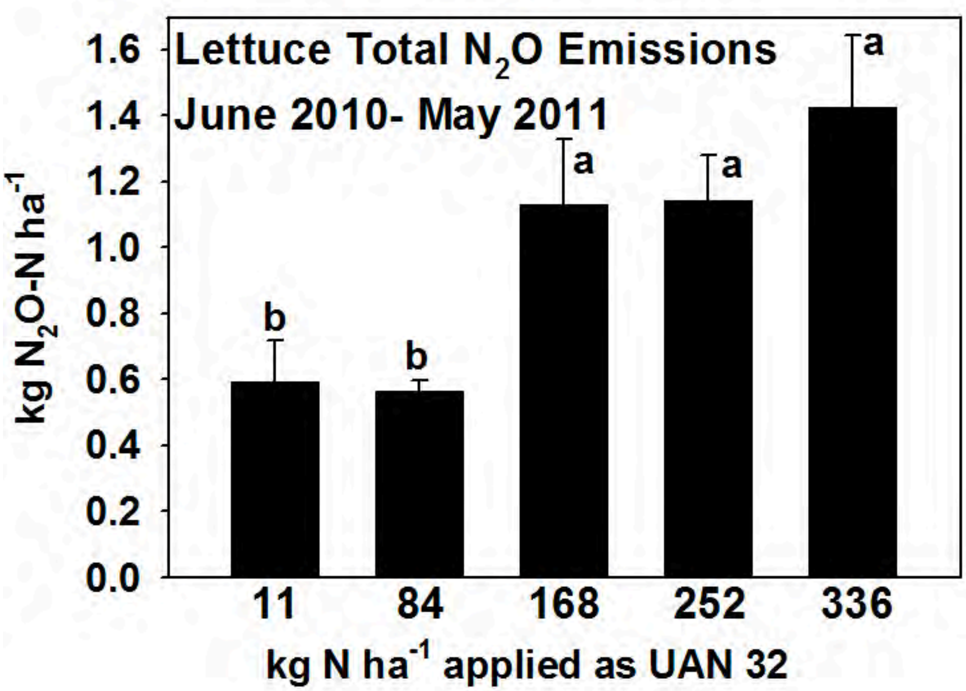


Figure 36. Mean annual N_2O emissions and standard errors during 2010/11 in the five N fertilizer treatments. Means designated with the same letter are not different from each other ($p<.05$). n=4.

In both years, the highest fluxes were measured after the fall irrigation, which took place 3.5 and 6.5 weeks after incorporating harvest residue by roto-tilling in 2009 and 2010, respectively. In 2009, 3 - 15% of the total annual N_2O was emitted during this event, whereas 24 – 62% was emitted during the rainy season (Figure 37). In 2010, the emissions after the fall irrigation accounted for 27 - 44% of the total annual N_2O emissions, and 14 to 46% of total N_2O emissions occurred during the rainy season (Figure 38). In both years, the contribution of the rainy season's N_2O fluxes to overall emissions decreased as total annual N_2O emissions increased.

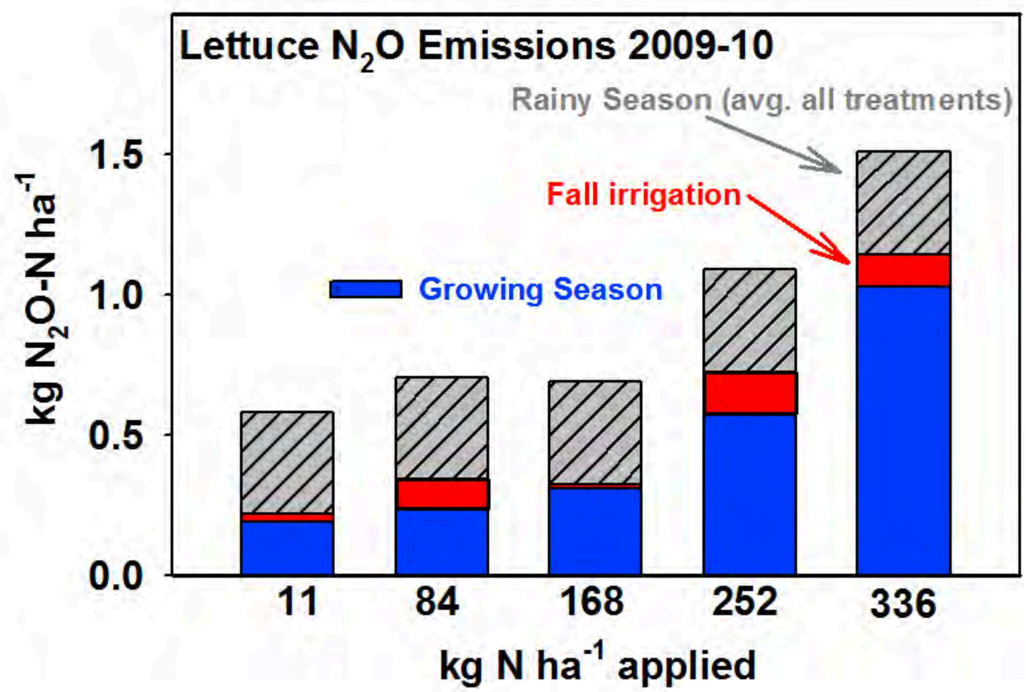


Figure 37. Seasonal distribution of N_2O emissions in 2009/10 at the different N fertilization levels. (n=4).

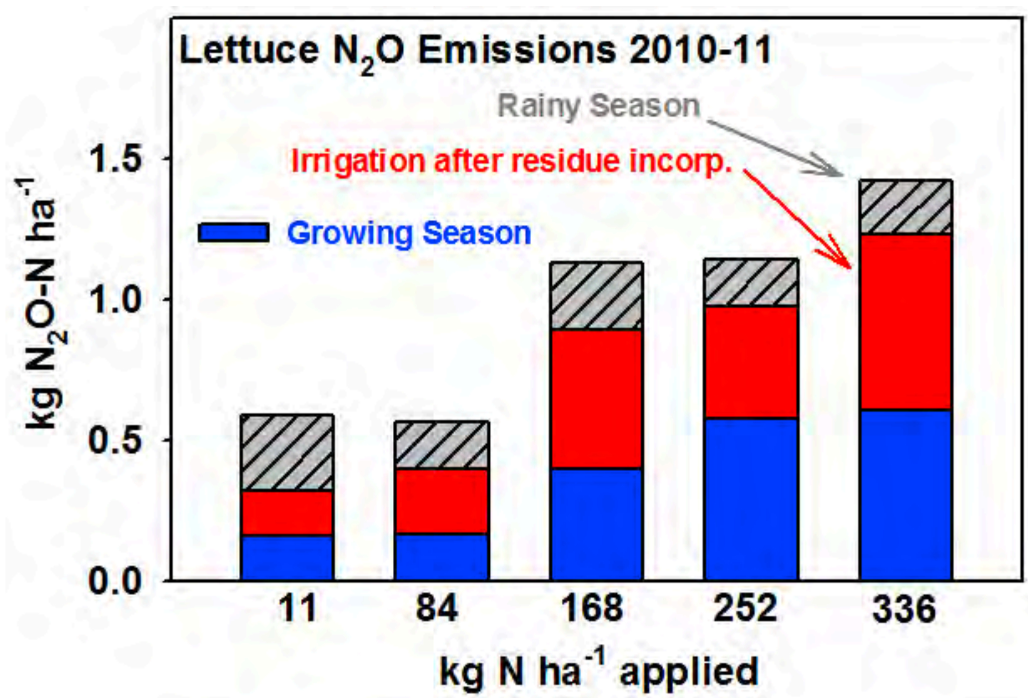
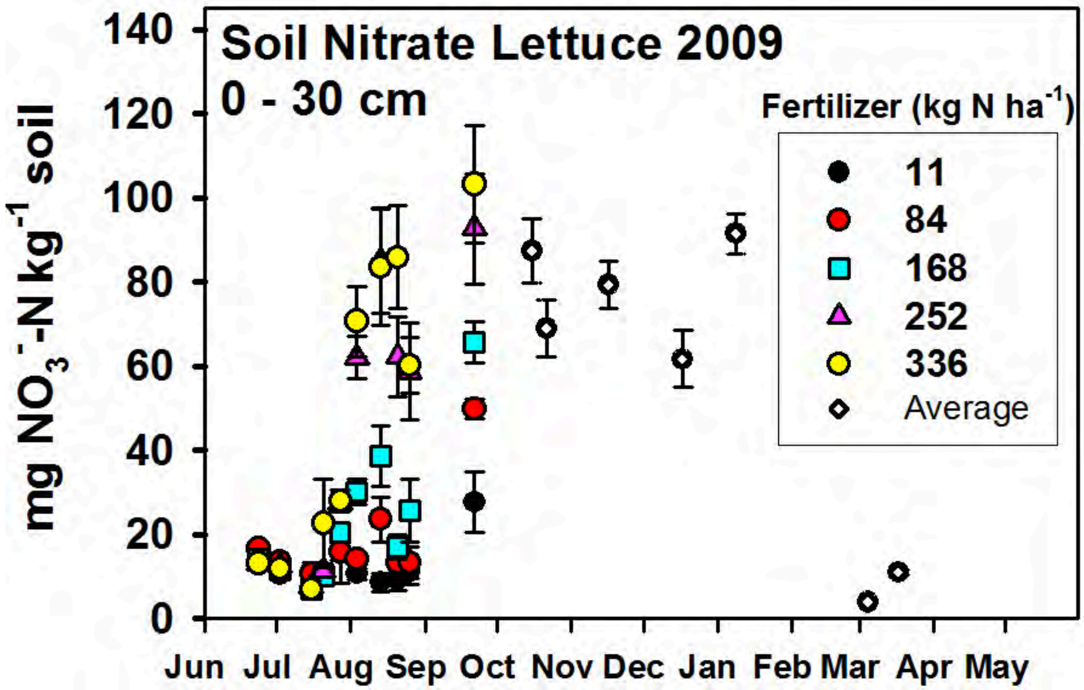
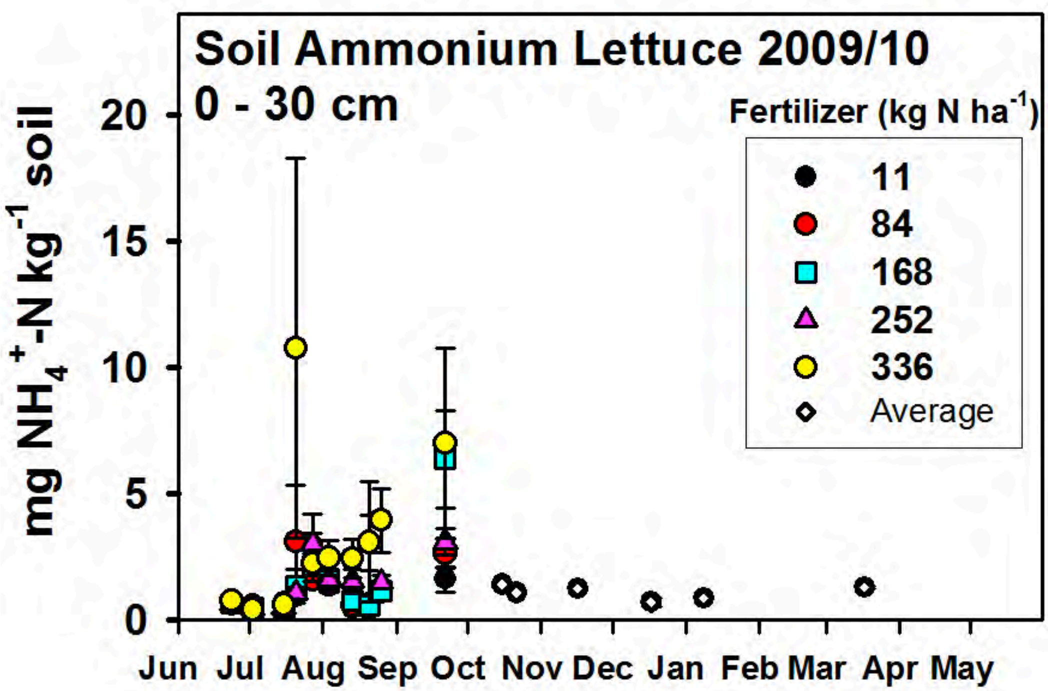


Figure 38. Seasonal distribution of N_2O emissions in 2010/11 at the five N fertilization levels (n=4).

In 2009, soil nitrate concentrations were at their highest level after harvest (Figure 39). Nitrate concentrations stayed high (60 – 100 mg N kg^{-1} soil) until January and declined only by March 2010. A similar pattern was observed in 2010/11, but the NO_3^- concentrations at the end of the growing season were between 40 and 60 mg N kg^{-1} soil. In both years, soil NH_4^+ concentrations were somewhat elevated during the growing season and declined to low levels in the fall.





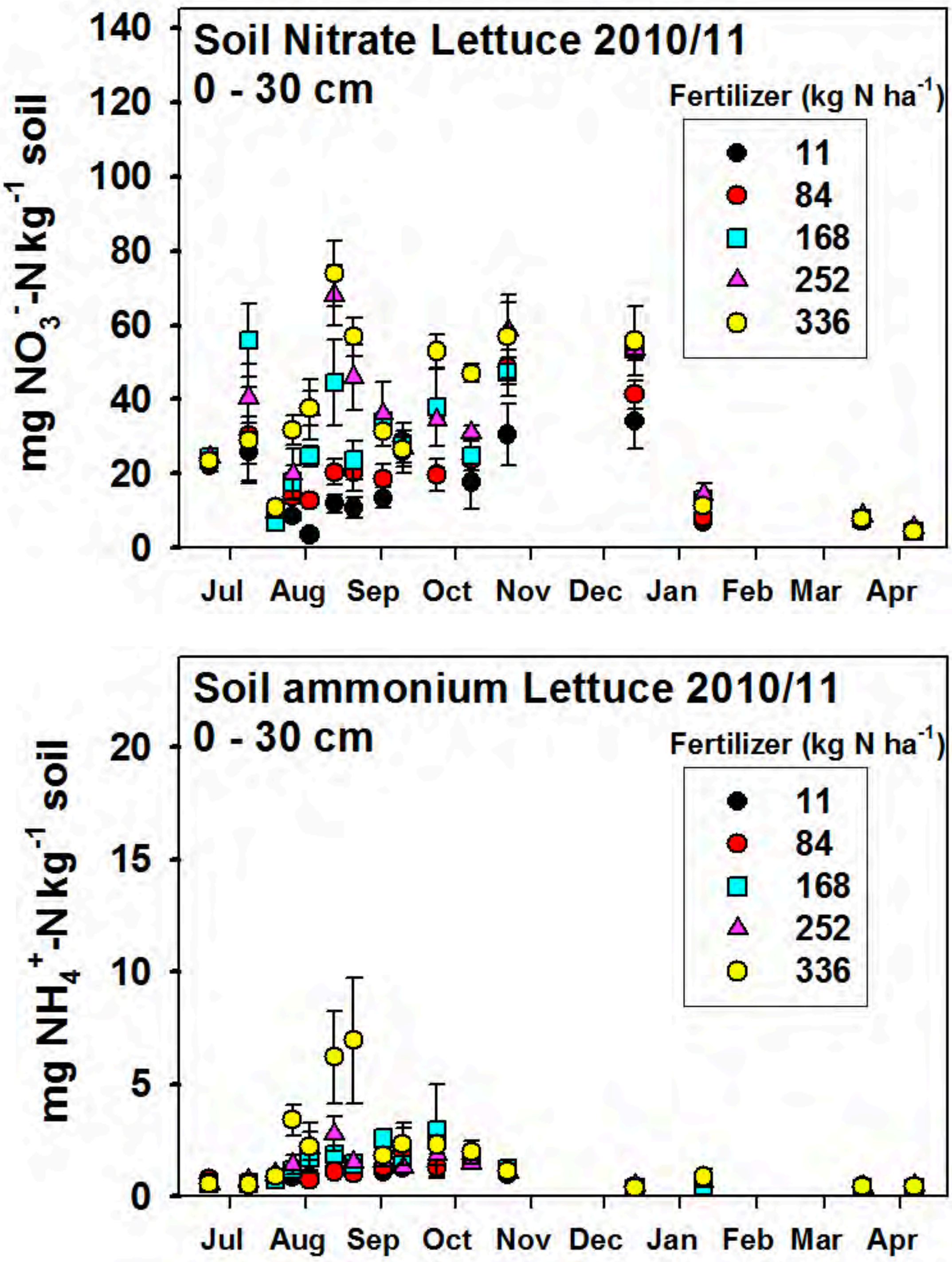


Figure 39. The above four panels show soil inorganic N (NO_3^- and NH_4^+) concentrations in the 0-30 cm layer in the experimental lettuce plots during 2009/10 and 2010/11.

In both years, marketable yield did not differ among any but the lowest N rate treatments, whereas the untrimmed biomass did not differ among the three highest N application levels (Figure 40). Overall, yields and biomass were lower in 2010 than in 2009. Nitrogen uptake and

N export (biomass N uptake minus trimmings) increased with increasing N applications (Table 14).

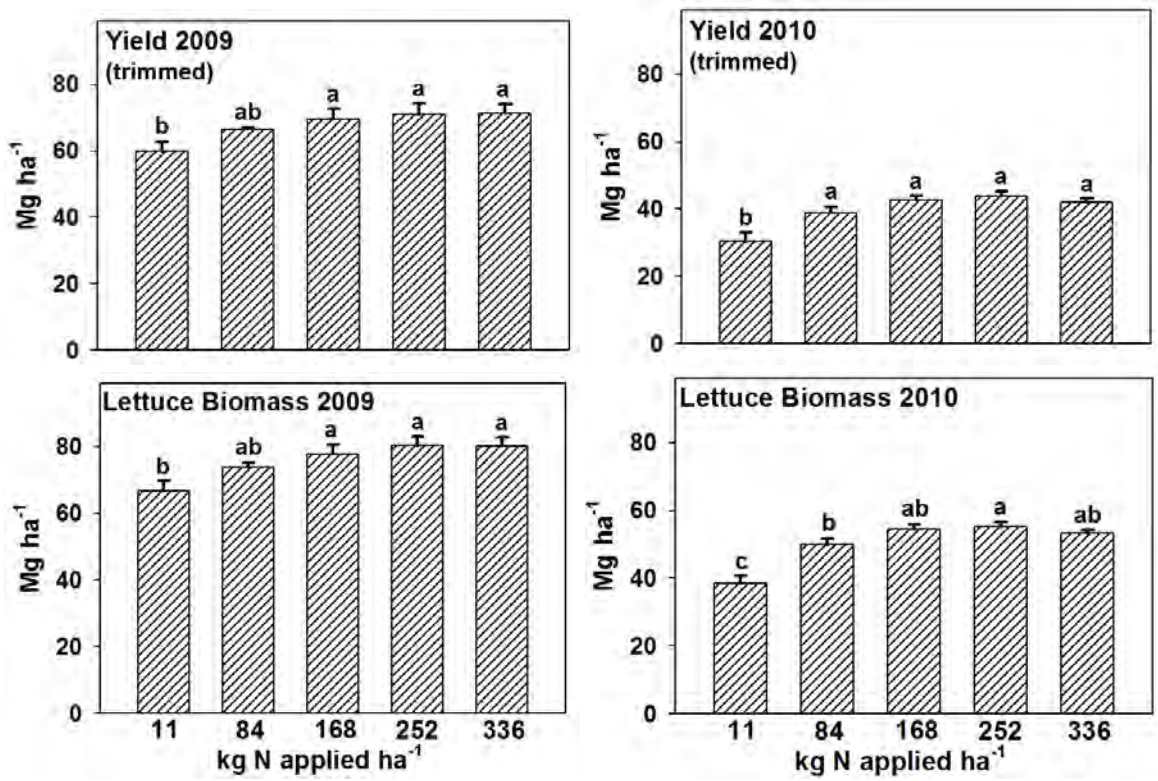


Figure 40. Mean marketable lettuce yields and total accumulated biomass in the different N fertilizer treatments in 2009 and 2010. Standard errors shown as line bars. Means designated with the same letter within each year and plant variable are not different from each other (p<.05). n=4.

Table 14. Mean N uptake by the lettuce biomass, mean N removal by the harvested crop, and the difference between the amount of N applied and crop N removal and standard errors in 2009 and 2010. Values within the same columns designated with the same letters are not different (p <.05). n = 4.

N applied	N uptake 2009	N uptake 2010	N removal 2009	N removal 2010	Application - export 2009	Application - export 2010
			kg N ha^{-1}			
11	98 ±7 ^{c}	64 ±8 ^{c}	88 ±7 ^{c}	51 ±7 ^{c}	-77	-40
84	115 ±4 ^{c}	97 ±7 ^{b}	103 ±4 ^{c}	75 ±6 ^{b}	-19	9
168	136 ±6 ^{b}	116 ±6 ^{ab}	122 ±7 ^{b}	91 ±5 ^{ab}	46	77
252	149 ±4 ^{ab}	125 ±6 ^{a}	132 ±5 ^{ab}	99 ±5 ^{a}	120	153
336	159 ±7 ^{a}	125 ±4 ^{a}	142 ±6 ^{a}	99 ±4 ^{a}	194	237

3.5. Rice

The N_2O emissions in rice systems were measured in spring before flooding of the rice fields, during fall drainage before harvest and during the post-harvest period, i.e. during draining after winter-flooding. In the first year (2009/10), the estimate of annual N_2O emissions is composed of measurements at the Biggs Rice Experiment Station (RES) and those in a grower field near Arbuckle (Table 15). The highest N_2O fluxes occurred during the two-month pre-flood period in the drill-seeded stale seedbed (no-till) treatment (DSNT). Under this management practice, 28 kg N ha^{-1} in the form of urea was applied at planting one month before flooding. The N_2O emissions due to this N fertilizer application approximately tripled the emissions compared to the treatment receiving no N fertilizer before flooding. The mean yields under the three management practices, each with three N fertilizer levels, as well as the corresponding data of plant N uptake and crop N export, are shown in Table 16.

Table 15. Mean N_2O emissions for three management types and three fertilizer N levels measured in spring before flooding (pre-flood), after fall drainage (drain to harvest), and during winter-drainage after winter-flooding (post-harvest) in 2009/10. Pre-flood and drain-to-harvest fluxes were measured at the Biggs Rice Experiment Station (n=4), and post-harvest fluxes (n=3) were measured at a grower field near Arbuckle.

	Wet-seeded convent.			**Drill-seeded no-till**			**Wet-seeded no-till**		
kg N ha^{-1} applied	**0**	**168**	**224**	**0**	**168**	**224**	**0**	**168**	**224**
	g N_2O-N ha^{-1}								
Pre-flood	4 ±4	4 ±4	4 ±4	285 ±57	885 ±368	885 ±368	154 ±53	154 ±53	154 ±53
Drain to harvest	62 ±24	61 ±23	66 ±12	111 ±64	50 ±13	120 ±34	121 ±64	124 ±26	209 ±101
Post-harvest	197 ±73								
Total	263	262	267	593	1132	1202	472	475	560

Table 16. Mean yields, crop grain N, crop straw N, and crop N uptake at Biggs Rice Experiment Station in 2009.

	Wet-seeded convent.			Drill-seeded no-till			Wet-seeded no-till		
kg N ha^{-1} applied	**0**	**168**	**224**	**0**	**168**	**224**	**0**	**168**	**224**
Yield (Mg ha^{-1})	5.6	10.8	10.6	5.6	10.1	9.2	3.5	9.5	9.5
kg grain N ha^{-1}	39	89	98	40	81	71	25	74	84
kg straw N ha^{-1}	22	55	69	19	50	47	15	61	59
kg N up-take ha^{-1}	61	145	167	58	130	118	39	135	142

Among the three management practices, the drill-seeded stale seedbed treatment had lower yields than the other two practices, which did not differ between each other (Table 17). The two N fertilizer levels produced similar yields, whereas the zero-N treatments resulted in the lowest yields.

In the second year (2010/11), the N_2O fluxes were measured in two grower fields in the Arbuckle area. Total annual N_2O emissions were similar in magnitude as in the first year although the distribution of N_2O emissions within the year was different from that in the previous year (Table 18). In the wet-seeded treatment, the drain-to-harvest and the post-harvest periods contributed about equally to the majority of the N losses as N_2O, whereas under the drill-seeded practice, it was mainly the drain-to-harvest period that contributed the bulk of the annual N_2O emissions. In 2010/11, under both practices, the pre-flood period produced relatively little N_2O. Yields tended to be lower under the drill-seeded than the water-seeded practice.

Table 17. Results of ANOVA for yields at Biggs Rice Experiment Station in 2009. Least squares means designated with the same letters are not different from each other (P<.05). n=4.

P > F	DF	Effect	LS Means
P<.01	2	Management practice	
		Wet-seeded convent.	8.9 ±.4 [a]
		Wet-seeded no-till	8.3 ±.4 [a]
		Drill-seeded no-till	7.5 ±.4 [b]
P<.0001	2	Fertilizer level	
		168 kg N ha^{-1}	10.1 ±.4 [a]
		224 kg N ha^{-1}	9.8 ±.4 [a]
		0	4.8 ±.4 [b]
NS	3	Management*Fertilizer	

Table 18. Mean N_2O emissions and standard errors (n=3) for the spring pre-flood, drain-to-harvest, and post-harvest periods, as well as total annual N_2O emissions, in grower fields near Arbuckle in 2010/11. Also shown are mean yields and grain N content of the rice crop and standard errors in the two fields. n=3.

	Wet-seeded	Drill-seeded
kg N ha^{-1} applied	140 (Aq. ammonia)	100 (urea)
	g N_2O-N ha^{-1}	
Pre-flood	36 ±5	-83 ±82
Drain to harvest	333 ±55	701 ±76
Post-harvest	484 ±83	118 ±43
Total annual	853 ±85	736 ±182
Yield (Mg ha^{-1})	12.0 ±0.2	8.0 ±0.2
Grain N (kg N ha^{-1})	64.8 ±2.9	90.0 ±2.0

The EFs in the first year ranged from 0.12 to 0.67%. In the second year, the EFs were 0.61 and 0.74% for the wet- and dry-seeded treatments, respectively (Table 7).

4. Discussion

This study focused on N fertility management and its effect on N_2O emissions in five types of CA cropping systems important in terms of acreage and economic value. In this section, we will first discuss the relationship between fertilizer level and N_2O emissions for each of the five systems and will also address other factors, such as soil moisture conditions, that may have influenced the magnitude of N_2O losses. We will address how N fertilizer inputs affected yields. The discussion will include explanations of each experiment's circumstances that may have influenced the outcomes. Furthermore, we will comment on the emission factors (EFs) derived in these experiments.

4.1. Tomato

In the tomato systems we observed a non-linear increase in N_2O emissions as fertilizer N inputs increased. Evidence of a non-linear increase of N_2O emissions in response to increasing rates of fertilizer N inputs has previously been reported in non-irrigated corn systems in the U.S. Midwest, Canada, and Australia (McSwiney and Robertson, 2005; Edis *et al.*, 2008; Ma *et al.*, 2010; Hoben *et al.*, 2011), while other researchers found linear N_2O responses in fertilizer N rate trials (Henault *et al.*, 1998; Halvorson *et al.*, 2008). We did not attempt to describe the exact form of this non-linear N_2O emissions response, given only the feasibility to conduct five fertilization rates, which is likely not sufficient to allow forming one accurate solution with confidence.

At the onset of this study, we had hypothesized that N_2O emissions would increase non-linearly when N was applied at levels greater than the maximum economic return to N rate (optimal economic N yield), as shown in studies of corn systems (McSwiney and Robertson, 2005; Hoben *et al.*, 2011). In 2011, we observed a significant increase in N_2O emissions when fertilizer was applied at 225 and 300 kg N ha^{-1}, whereas yields did not differ among the 162, 225 and 300 kg N ha^{-1} treatments. In 2010, the N application rates had no effect on yields, which was likely due to the low water inputs limiting yields in 2010, but N_2O emissions were also greater at the two highest N application rates. Our experiments suggest that N fertilization at a rate of 162 kg N ha^{-1} was sufficient to achieve the full yield potential at this site. University of California research has shown that under most circumstances maximum yields of furrow-irrigated processing tomatoes can be obtained with 112-168 kg N ha^{-1} (Hartz *et al.*, 1996). The 225 and 300 kg N ha^{-1} applications were excessive under the growing conditions in the two years of this study.

Some additional information on N rate, N_2O emissions, yields, and N use is worth noting:

1. The N_2O emissions occurring between the UAN32 side dress application and harvest increased non-linearly with increasing fertilizer N rates (Figures 8 & 9; statistical results not shown), suggesting that the N_2O response is directly related to N application rate.

2. The cropping season N_2O emissions were lower in 2010 than 2011, but adding the emissions after the first rainfall brought N_2O losses in the two years to similar values. The lower N_2O emissions during the 2010 cropping season may have been due to the lower irrigation water inputs (Figures 1,3, and 5). High N_2O emissions with the first rainfall after harvest have been observed earlier (Burger *et al.*, 2005; Kallenbach *et al.*, 2010). The reason for the much higher N_2O emissions after the first rainfall in 2010 than 2011 may have been due, in part, to the amount and intensity of this rainfall event, which was 15 mm within 5 h in 2010 vs. 10 mm more or less evenly distributed over a 4 d period in 2011. However, the

residual inorganic N levels at the end of the tomato season may have played a role too. Nitrate concentrations were significantly higher in 2010 (Figure 10) than in 2011 (Figure 11), and this may have been due to lower N uptake by the tomato plants (Tables 7 & 8) because of the inadequate soil moisture during the 2010 cropping season.

3. The N left in the soil or lost from the system (e.g. by leaching) increased linearly with increasing N rate and was higher in 2010 than 2011 (Tables 7 & 8), which supports the preceding argument regarding plant N uptake.

4. In 2010, fruit N content (crop N removal) was similar in all fertilized treatments (Table 7), indicating that a factor other than N (most likely water) was limiting yields, whereas in 2011, fruit N content was significantly lower in the 162 kg N ha^{-1} than in the higher N rate treatment (Table 8), which suggests N availability was lower in this than the higher N rate treatments.

Overall, the results provide strong evidence that under a variety of circumstances using a rate of 162 kg N ha^{-1} produced lower N_2O emissions than applying N at higher N rates without compromising yield.

To our knowledge, there are no published values of annual N_2O emissions for processing tomato systems. In earlier studies in tomato, measurements were not made frequently enough to reliably estimate annual N_2O emissions (Burger *et al.*, 2005; Kong *et al.*, 2009; Kallenbach *et al.*, 2010). In another horticultural crop system located in a Mediterranean climate (Spain), furrow-irrigated melon production fields that received 175 kg N ha^{-1} had emissions of 5.3 kg N_2O-N and an EF of 3.0% (Sanchez-Martin *et al.*, 2008). In broccoli production systems in SE Scotland, emissions ranged from 9.1 to 12.2 kg N_2O-N ha^{-1} with EFs from 5.2 to 7.0% (Dobbie *et al.*, 1999). For radish, bok choi, lettuce, and celery production systems in Nanjing, China, emissions were 1.3, 1.0, 2.9, 5.8 kg N_2O-N ha^{-1} with EFs of 0.4, 0.7, 2.2, and 0.7%, respectively (Pang *et al.*, 2009). In our study, the N_2O emissions in the tomato system may have been higher than in the lettuce system because the tomatoes were furrow-irrigated and received all the N fertilizer early on in the growing season, whereas lettuce was drip-irrigated and received N fertilizer in increments during growth. Other factors, such as soil type and temperatures may have played a role too in affecting the differences in N_2O emissions. Although the EFs in tomato were higher than in the other systems under study in this project, it is important to note that at the rate of 162 kg N ha^{-1}, the 2-year average EF (1.11%) was close to the IPCC default EF of 1%.

4.2. Wheat

The wheat system data were analyzed for each individual year since the experiments were in different locations and fertilizer applications were not the same. In both years, total annual N_2O emissions were greatest in the treatment that included AA as starter N fertilizer, which typically makes up about half of the applied N. Especially in 2009/10, the overall differences in N_2O emissions were mostly due to the emissions following the AA application. Although the AA application in 2010/11 did not affect annual N_2O emissions as much as in 2009/10, the emissions in this treatment were of the same magnitude as those resulting from the 50 kg N ha^{-1} greater application in the form of broadcast/disked in AS. We do not know of any comparisons between AA and AS, but in previous side-by-side field trials, in agreement with our results, N_2O emissions following AA applications have consistently been higher than those following broadcast urea (Breitenbeck and Bremner, 1986; Thornton *et al.*, 1996; Venterea *et al.*, 2005; Venterea *et al.*, 2010; Fujinuma *et al.*, 2011).

In 2010/11, total N_2O emissions increased with fertilizer N rates. It is not clear why emissions in 2009/10 in the 254 kg N ha^{-1} treatment were not higher than in the other AS starter treatments. Generally, in 2009/10, N_2O emissions were rather low, especially early on in the growing season, possibly because little precipitation occurred until the second half of January, and this may have obscured any differences that might have taken place with greater soil moisture. In both years, the emissions following the urea application were more comparable to each other. In both those years, spring precipitation was abundant.

The EFs ranged from 0.20% (254 kg N applied ha^{-1}) to 0.64% (203 kg N applied ha^{-1}) in 2009/10 and from 0.57% (154 kg N applied ha^{-1}) to 0.98% (210 kg N applied ha^{-1}) in 2010/11. According to a recent meta-analysis of 25 studies, in which N_2O emissions and yields were measured in wheat systems for at least one season, the average EF was calculated as 1.21% and the average amount of N fertilizer applied as 115 kg N ha^{-1} (Linquist *et al.*, 2012). So at all fertilizer levels, the EFs in our study were below this average EF derived from studies all over the world.

Wheat yields did not differ among the N application treatments, so determining the economic N rate in these fields was not possible. Since one of our objectives was to gather information to develop recommendations for best management practices, we also examined grain N content and N removal from the system by the harvested crop to estimate an adequate N rate for these wheat fields. Based on grain N content in 2010/11, N was not limiting yields since grain N was the same for all the fertilized treatments. Moreover, crop N removal by the harvested grain was either similar or greater than the amount of N applied as fertilizer. The field of the 2010/11 trial had previously been in alfalfa, which increases N fertility for a subsequent crop. An N credit to the crop following alfalfa is recommended because of the alfalfa residues, which are richer in N than those of non-legumes, left in the soil (Pettygrove and Putnam, 2009). In Mediterranean climates, the contribution of alfalfa to a subsequent crop has been estimated at 45 to 90 kg N ha^{-1} (Putnam *et al.*, 2001; Pettygrove and Putnam, 2009; Ballesta and Lloveras, 2010). Based on our yield data and crop grain N content, 154 kg N ha^{-1} would have been sufficient as N addition if a legume-credit was given due to the preceding alfalfa cultivation. However, N_2O emissions did not differ significantly between the 154 and 210 kg N ha^{-1} (using AS as starter fertilizer) rate, i.e. applying 154 instead of 210 kg N ha^{-1} would not lower N_2O emissions.

A similar conclusion might be drawn for the previous season (151 kg N ha^{-1} sufficient to achieve adequate yield), since neither yield nor grain N content differed among the three highest N application levels. The 2009/10 field had previously been in tomatoes and an N credit would have been due to the residual inorganic N content of <30kg N ha^{-1} in the top 30 cm of soil (results of pre-planting soil tests not shown). With a 151 kg N ha^{-1} fertilizer application, a surplus of 29 kg N ha^{-1} (crop N removal minus fertilizer N application) would have remained in the soil, which indicates that using this N rate would maintain the long term sustainability of this soil.

Nitrous oxide emissions in cropping systems can also be assessed in relation to crop yields, i.e. as N_2O loss per unit of grain yield (kg N_2O-N Mg^{-1} grain) or yield-scaled N_2O emissions, to take into account the fact that demand for producing more food on the limited land area available is increasing and best-possible solutions must be found for the N_2O problem (Van Groenigen *et al.*, 2010). This approach is particularly useful for comparisons of GHG emissions among major food staples, such as cereal grains. In our study, the highest yield-scaled emissions in 2009/10 were 0.15 kg N_2O-N Mg^{-1} grain (203 kg N applied ha^{-1}) and in 2010/11 0.21 kg N_2O-N Mg^{-1} (266 kg N applied ha^{-1}). Both these values are well below the average yield-scaled N_2O

emissions of 0.35 kg N_2O-N Mg^{-1} evaluated in the above meta-analysis of 25 wheat studies (Linquist *et al.*, 2012).

To summarize, the N_2O emissions in the fields of our study were below the average of those in other wheat production systems by several measures, such as the EFs and yield-scaled N_2O emissions. Perhaps most importantly, our study indicated that N_2O emissions could be reduced by not choosing AA as N fertilizer source.

4.3. Alfalfa

The major part of the N_2O emissions occurred during the alfalfa growing season in the summer. The N_2O released immediately following the flooding of the fields, which coincided with total or nearly total saturation of the soils, accounted for almost all of the annual emissions. The emissions during the winter (Nov – March), when alfalfa is dormant, were much lower than the ones during the growing season even though the WFPS was >70% for extended periods (2010/11). The much higher spikes of N_2O fluxes following check flood irrigation in the 5-year than in the 1-year old stand were largely responsible for the substantial difference in emissions between the two fields. We hypothesized that the field of the older stand was losing vigor and had therefore more decaying plant matter, providing more C to N mineralization to support the activities of denitrifying bacteria. However, we did not detect any differences in NH_4^+, NO_3^-, or DOC concentrations (paired t-tests) between the two fields. The WFPS during irrigation events was similar as well. Rochette (2004) also found larger emissions in a 4th year than in first- and one-year alfalfa stands and noted that soil mineral N, being lower in the 4th year than in the field of the one-year stand, was not a useful indicator of flux intensity. More measurements in other paired field trials might be needed to determine if stand age indeed increases N_2O emissions. Only a few studies have been conducted on N_2O emissions in alfalfa fields. The annual N_2O emission estimates of the 5- and 1-year alfalfa field are among the higher estimates of those reported in other studies by Duxbury *et al.* (1982) 2.3 – 4.2 kg N ha^{-1}, Robertson *et al.* (2000) 1.9 kg N ha^{-1}, Wagner-Riddle *et al.* (1996) 1.0 kg N ha^{-1}, and Rochette *et al.* (2004) 0.67 (1st and 1-year stand) – 1.45 kg N ha^{-1} (4th year stand).

The EFs of the alfalfa systems were by far the highest among those calculated in the present study although the annual N_2O emissions in alfalfa were comparable in magnitude to those of the tomato systems. However, the N inputs, calculated on a per year basis according to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, were much lower than in the annual systems receiving synthetic N fertilizer. The annual biomass N input was calculated as 34.6 and 46.9 kg N ha^{-1} for the fields of the 5-year and 1-year old alfalfa stands, respectively. The one-time fertilizer N application in the form of ammonium phosphate during the two years raised the annual total N input to 43.1 and 55.4 kg N ha^{-1}. The N inputs in the alfalfa systems calculated by the IPCC method may have been underestimated, especially in the 5-year old stand field. This recently adopted IPCC procedure to calculate biomass N inputs from pastures with perennial legumes has to date not been validated for alfalfa and may be revised to reflect more accurately the N inputs of N-fixing alfalfa plants.

4.4. Lettuce

In the lettuce system, the annual N_2O emissions were similar in magnitude from year to year, but both the emissions among treatments and the distribution of N_2O emissions during the year varied. In 2009/10, N_2O emissions were greater at the two highest N rates, but in 2010/11, the emissions in the three highest N treatments were similar. The irrigation several weeks after harvest provided important information. Data on N_2O emission after wetting of dry soil after harvest are indicative of responses to the first rainfall and are also relevant with regard to general lettuce production practices since fields are typically sprinkler-irrigated for seed germination. These irrigation events occur several times a year. In the Central Coast area, several crops per year are grown on most fields (annually 2.4 crops per unit land area).

The first rainfall after soils have been dry for some time often results in high N_2O emissions both in natural ecosystems (Davidson, 1992) and in cropping systems after harvest (Burger *et al.*, 2005; Kallenbach *et al.*, 2010; Garland *et al.*, 2011; tomato systems this study). Often, subsequent rainfall events produce much lower N_2O fluxes than the first rainfall (Garland *et al.*, 2011; tomato and lettuce, this study).

The emissions from the first rainfall after harvest seemed to have the greatest effect on the year-to-year differences. In 2010/11, the contribution of the emissions after the fall irrigation to overall N_2O emissions was greater than in 2009/10 ($p<.001$), but the total annual emissions were similar, at least for the two highest and the two lowest N rates. The emissions after the fall irrigation may have been greater because in 2010 7 mm more water was applied (26 vs. 19 mm in 2010 and 2009, respectively). Mineral N concentrations at the time of the fall irrigation, on the other hand, were greater in 2009 than 2010 even though crop N removal was lower in 2010.

The effect of incorporating lettuce trimmings on N_2O emissions was not clear. In 2010, a larger amount of lettuce trimmings was incorporated and the residue was left to decompose in the soil for a longer time than in 2009, but these factors may not have been the reason for the greater N_2O emissions after the simulated rainfall in 2010 than 2009 since the amount of N_2O emitted was not different from that in control plots, where no aboveground biomass had been incorporated (results not shown). Incorporation of residue with low C/N ratio (7.5) has been reported to increase N_2O emissions (Baggs *et al.*, 2000). However, in the present study, shoot and root C/N ratios in the highest N rate treatment were 11 and 15, respectively, which initially may have promoted immobilization of N by microorganisms rather than mineralization. It is common practice for growers to trim off 10 to 40% of harvested lettuce biomass and return it to the soil. Based on our results, this practice does not seem to promote N_2O emissions, but more studies are needed to confirm this.

The lower yields in 2010 may have been due to an increase in Sclerotinia disease since lettuce was grown at the site for the third year in a row, somewhat cooler conditions, and a shorter planting to harvest period (62 d in 2010 vs. 65 d in 2009). In both years, yields in all but the 11 kg N ha^{-1} treatment did not differ while N export by the harvested crop (marketable yield) was lower than the N fertilizer addition in the three highest N rate treatments. By considering N_2O emissions and yields under the conditions of both years, an N application somewhere between 168 and 84 kg N ha^{-1} is recommended to lower N_2O emissions without compromising yields. For lettuce production under drip irrigation, 20 to 30% lower fertilizer rates than in conventionally irrigated fields have been recommended by University of California researchers (Turini *et al.*, 2011).

The annual N_2O emissions in our study were much lower than those measured in vegetable production systems in St. Barbara county three decades earlier in the pioneering research by Ryden and Lund (1980). These researchers reported annual emissions from 20.2 to 41.8 kg N_2O-N ha^{-1} in lettuce and celery fields receiving annual inputs of 620 kg N ha^{-1}, which resulted in EFs ranging from 3.3 to 6.7%. The EFs in our study were lower by almost an order of magnitude (ranging from 0.41 to 0.84%). It is likely that more than one crop was grown at those sites, unlike in our study, which reports emissions in fields cropped only once per year. Another difference between the two study sites was the irrigation practice. The fields Ryden and Lund (1980) reported about were furrow-irrigated, whereas those of our study were surface-drip irrigated, which enabled us to apply N fertilizer in increments via the irrigation water (fertigation). This may have also been one of the reasons why N_2O emissions and EFs at our lettuce site were lower than in the tomato systems even though fertilizer N rates in the two systems were similar. The shorter cropping cycle, lighter soil texture, and lower temperatures at the lettuce site were likely other factors that contributed to this outcome. Comparisons of N_2O emissions and EFs with those of other horticultural crop systems were presented above (section 4.1, tomato).

4.5. Rice

The major greenhouse gas in rice production systems is methane (CH_4). Because of rice's importance as one of the world's food staples, many efforts are underway to lower its carbon footprint (Horwath, 2011). By shortening and interrupting the flooded period of the growing season, CH_4 emissions may be lowered, but these practices may increase N_2O emissions (Cai *et al.*, 1997; Zou *et al.*, 2007). Most N_2O emissions from rice systems occur during drainage events when NH_4^+ is converted to NO_3^-, which can then become subject to denitrification (Yao *et al.*, 2010). Reducing herbicide inputs and optimizing fertility management are other research priorities. We investigated how alternative management practices with the goals of lowering CH_4 emissions and improving weed and fertility management affect N_2O emissions.

Under the practice with the highest N_2O emission estimate in 2009/10 (DSNT, drill-seeded no-till), rice fields are flooded for 30 d less than under conventional practices. They also produce less CH_4 (Assa & Horwath, unpublished). However, the DSNT practice resulted in a yield reduction of approx. 10%. A similar yield trend was evident in 2010/11. However, in 2010/11, grain N content was higher under this practice even though the N application rate was lower than under the wet-seeded practice, thus suggesting greater N use efficiency in the dry-seeded treatment.

Applying N fertilizer before the fields were flooded was a one-time experiment that is not generally practiced by farmers because this practice did not result in higher yields than applying all the fertilizer immediately before flooding. This study showed that such a practice also increased N_2O emissions. We can therefore assume that without the pre-flood N fertilizer addition in 2009/10, wet- and dry-seeded no-till systems (DSNT and WSNT) produced a similar amount of N_2O emissions (0.5 to 0.6 N_2O-N ha^{-1}). If the emissions of the fertilized pre-flooding period were excluded, the EFs for the no-till systems in 2009/10 would have been within the narrow range of 0.25 to 0.31%. The emissions in the wet-seeded conventional (WSCT) treatments tended to be lower (about 0.3 kg N_2O-N ha^{-1}) with EFs of 0.12 and 0.15.

Using different fields for the research precluded a formal statistical analysis of N_2O emissions between fields and years. In 2010/11 the N_2O emissions in wet- and dry-seeded treatments were similar (0.85 and 0.74 kg N_2O-N ha^{-1}, respectively), with EFs of 0.61 and 0.74, respectively.

The EFs at our sites ranged from 0.12 to 0.74% (mean 0.42%), whereas in a meta-analysis comprising 17 studies (all in Asia) an average EF of 0.68% was calculated (Linquist *et al.*, 2012). In another review article, EFs of 0.22% were reported for continuously flooded rice systems and 0.37% for intermittently flooded rice systems (Akiyama *et al.*, 2005), whereas the 2006 IPCC guidelines recommend assuming an EF of 0.3% for rice paddies (IPCC, 2006). In order to develop best management practices, the N_2O results must be evaluated in conjunction with the CH_4 emission estimates (not addressed in this report) since, according to the above meta-analysis of 17 rice studies, N_2O, on average, contributed only 11% to the total global warming potential (CH_4 and N_2O) in rice production (Linquist *et al.*, 2012). Such an analysis should also include yield data that could be used in comparing the yield-scaled global warming potential of different management practices.

5. **Summary and Conclusions**

The objective of this study was to estimate annual N_2O emissions for five major California cropping systems — tomato, wheat, alfalfa, lettuce, rice — under typical management practices and at several fertilizer N addition rates. We measured N_2O fluxes and key environmental and agronomic factors that influence these emissions in representative systems for two years and assessed the relative importance of different management events and seasonal periods to the annual N_2O budgets. We also investigated the relationship among fertilizer N application, yields, and N_2O emissions. The study was conducted because there was a lack of California-specific field-level N_2O emission data to assess N_2O emission inventories based on regional, cropping system-specific emission factors (EF; the percentage of fertilizer N emitted as N_2O-N). Secondly, N_2O flux and ancillary data were needed to calibrate and validate existing biogeochemical models simulating N_2O emissions. Another objective was to identify management practices that minimize N_2O emissions without compromising yield potential.

A database of N_2O fluxes, soil moisture and temperature, and soil mineral N measurements, and other soil characteristics was compiled for the cropping systems studied. In tomato, wheat, and lettuce systems, 5 fertilizer N treatments were assessed. In alfalfa, stands of different ages were assessed. In rice, 3 management practices and 3 N levels were assessed. All treatments within cropping systems were replicated 3 or 4 times. Over the life of the project, >14,000 N_2O flux measurements were made.

Our results confirmed that synthetic fertilizer N is a primary driver of N_2O emissions. In tomato, wheat, lettuce and rice systems, N_2O emissions increased with increasing rates of N additions. Moreover, in tomato, both the annual N_2O emissions and the N_2O emissions during the period between the major N fertilizer (UAN32 side dress) application and harvest increased non-linearly (exponentially) with increasing N inputs.

In addition, in agreement with earlier investigations, we presented evidence that anhydrous ammonia (AA) produces greater N_2O emissions (on the order of at least 30%) than N fertilizers dispersed through the soil (not banded), as demonstrated in the wheat systems in the present study.

For tomato, wheat, and lettuce production systems, we identified the N fertilizer application rates that resulted in significantly higher N_2O emissions than those following lower N additions, and similarly, we attempted to determine the N rates, above which yields did not increase (i.e. the maximum economic return to N rate, or economic N yield) with the objective of relating yield and N_2O responses to N rates and developing best management practices. For tomato, N rates >162 kg N ha^{-1} significantly increased mean annual N_2O emissions from 1.8 to 4.1 (225 kg N applied ha^{-1}) and 4.3 kg N_2O-N ha^{-1} (300 kg N applied ha^{-1}), while not increasing yields. For wheat, either applying AA at a rate of about 200 kg N ha^{-1} or broadcast ammonium sulfate at a rate of 266 kg N ha^{-1} resulted in significantly higher N_2O emissions (2.1 and 2.2 kg N_2O-N ha^{-1} for AA and AS, respectively) than other N fertilizer applications, whereas yields and grain N content did not differ among N fertilizer levels of 154 kg N ha^{-1} and higher. For lettuce, N_2O emissions significantly increased in one year at an N rate >168 kg N ha^{-1} from 0.69 to 1.09 kg

N_2O-N ha^{-1}, and in the other year at a fertilizer level >84 kg N ha^{-1} from 0.56 to 1.13 kg N_2O-N ha^{-1}. However, yields did not differ among N fertilizer treatments of 84 kg N ha^{-1} and greater.

Nitrous oxide emissions in rice systems occur mainly before planting, between field drainage and harvest, and following drainage after winter flooding. Applying fertilizer N in dry-seeded rice systems before flooding more than doubled N_2O emissions while not increasing yields. Except for the pre-flood-fertilized treatments, the annual N_2O emissions in rice cropping systems were within 0.26 – 0.85 kg N_2O-N ha^{-1}, which is below the average loss of 0.88 kg N_2O-N ha^{-1} recorded in 17 other recent rice studies (Linquist *et al.*, 2012). The experimental results in rice systems should be evaluated in the context of total greenhouse gas emissions, including CH_4, which is not covered in this study.

Among the N fertilized systems, tomato had the highest EFs, ranging from 0.92 to 2.08%. In the other horticultural crop system, lettuce, which received similar amounts of N fertilizer, the EFs were between 0.41 and 0.84%. Part of the reasons for the difference in N_2O emissions between tomato and lettuce may have been the timing of fertilizer application and the irrigation technique. Lettuce was drip-irrigated and N fertilizer was applied in increments as fertigations during the exponential growth phase of the crop, whereas the tomato plants were furrow-irrigated and received all the fertilizer early on in the growing season. The EFs in wheat ranged from 0.20 to 0.98% and were below the average EF of 1.21% calculated in 25 other wheat studies (Linquist *et al.*, 2012). In the rice systems, the EFs ranged from 0.12 to 0.74% with a mean of 0.42%, whereas 0.68% was reported as the average EF among 17 other rice studies (Linquist *et al.*, 2012).

In alfalfa, which was not N fertilized, the main findings were that most of the total N_2O losses occurred as short, intense bursts immediately following flood irrigation, and that the mean annual N_2O emissions in a 5-year old stand were significantly higher and more than twice as large than those in a 1-year old stand (5.20 vs. 2.30 kg N_2O-N ha^{-1}). We hypothesized that more decaying plant material from root turn-over, and thus, more mineralizable substrate was available to microorganisms in the soil of the older stand, and that this lead to higher denitrification rates. However, we lack direct evidence supporting this notion. The EFs in the alfalfa systems were high (12.06 and 4.15% in the 5-year and 1-year old stand, respectively) mainly because the annual N inputs, which were calculated according to current IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) as the percentage of root and shoot biomass N incorporated into the soil at stand removal, were low.

This study indicated that, in general, N_2O emissions during the rainy season are relatively less important than those during summer, a finding that based on the literature was not known prior to this study. The winter N_2O fluxes contributed about 10% to overall emissions in alfalfa, 10-25% in tomato, and 14-50% in lettuce. In wheat systems, all the emissions occur in the winter growing season when the N fertilizer applications are made.

Overall, at several occasions, very high N_2O emissions occurred following the first rainfall after harvest after soils had been dry for some time. In tomato systems, large emissions occurred even from soil that had not received N fertilizer in that season. In the lettuce experiments the highest emissions took place following the fall sprinkler irrigations 3 to 6 weeks after harvest. These irrigations were comparable to precipitation events. The implication of these N_2O emission responses after periods of dry soil conditions is that in lettuce fields, where typically sprinklers are used to establish several crops per year (on average 2.4), the highest N_2O emissions may occur with the first (sprinkler) irrigations.

6. Recommendations

The strategies with the highest potential to lower N_2O emissions in California cropping systems are to increase N use efficiency and to control soil moisture through alternative irrigation techniques. This study demonstrated that N_2O is emitted in spikes, or bursts, of emissions in response to relatively high soil moisture content. Recently published evidence showed that sub-surface drip irrigation (SDI) reduces N_2O emissions compared to furrow-irrigation in tomato and other crops (Kallenbach *et al.*, 2010; Sanchez-Martin *et al.*, 2010). With SDI, fertilizer N can be delivered in increments according to crop N demand, thus allowing the crop to compete more effectively and efficiently with N_2O producing microorganisms.

It is recommended to focus N_2O emission research on SDI, where appropriate, e.g. in alfalfa, lettuce and tomato systems. In alfalfa, SDI could potentially lower N_2O emissions since this study showed that most of the emissions occurred following saturation of the soil after check flood irrigation. In lettuce, a formal comparison of N_2O emissions under drip and sprinkler irrigation would be valuable. There is potential to increase N use efficiency in lettuce by using drip irrigation. The conversion of processing tomato production to SDI is well underway with >50% of the crop irrigated in this fashion. In tomato, further gains to increase N use efficiency may be possible by using different formulations of N fertilizers delivered through the drip lines. More information is needed on tomato N use with drip irrigation. There are few data on the fate of N fertilizers delivered through the drip system. This is important knowledge because N not used by a crop may well become the substrate of N_2O production after the growing season, as demonstrated in this project. In addition, the use of cover crops to “catch” or immobilize N may be appropriate to reduce N_2O emissions.

In wheat systems, this study showed that fertilizer formulation affects N_2O emissions more than N rates. Therefore, future research must consider alternative N fertilizer types. The wheat system is also a good candidate for testing nitrification inhibitors because the winter rains potentially remove NO_3^- from the root zone, leaving less N for crop uptake. If, by using a nitrification inhibitor, more N could be preserved in the soil as NH_4^+, which is not prone to leaching, N use efficiency of wheat systems could potentially be improved.

More data are needed on the fate of N in alfalfa systems. There are virtually no data on N fixation, on responses by alfalfa root systems to cuttings, or harvests (specifically root system turnover that releases C and N), and on the amount of N released to a subsequent crop when an alfalfa stand is removed and plowed under and converted to the next crop. Better knowledge about these processes would enable farmers to manage N for crops following alfalfa better, using N fertilizer judiciously, and this could potentially increase N use efficiency. More data are also needed to validate the finding that more N_2O is released as an alfalfa stand ages.

Our study has both validated and produced refined emission factors for California crops. We have also shown the biophysical and management factors that lead to N_2O production. Finally, we have identified agronomic practices that have potential to reduce N_2O emissions. Though our work represent a substantial effort in collecting N_2O emission data across different cropping systems, care must still be exercised in broadly interpreting results. As more data becomes available and the data is used in biogeochemical models, a better understanding of the pathways of N losses and factors affecting N_2O emissions will help California farmers maintain crop productivity through utilizing fertilizers more efficiently.

7. References

Akiyama, H., Yagi, K., Yan, X.Y., 2005. Direct N_2O emissions from rice paddy fields: Summary of available data. Global Biogeochem. Cycl. 19.

Baggs, E.M., Rees, R.M., Smith, K.A., Vinten, A.J.A., 2000. Nitrous oxide emission from soils after incorporating crop residues. Soil Use and Management 16, 82-87.

Ballesta, A., Lloveras, J., 2010. Nitrogen replacement value of alfalfa to corn and wheat under irrigated Mediterranean conditions. Spanish Journal of Agricultural Research 8, 159-169.

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Emissions of N_2O and NO from fertilized fields: Summary of available measurement data. Global Biogeochem. Cycl. 16.

Breitenbeck, G.A., Bremner, J.M., 1986. Effects of rate and depth of fertilizer application on emission of nitrous-oxide from soil fertilized with anhydrous ammonia. Biol. Fert. Soils 2, 201-204.

Bremner, J.M., Blackmer, A.M., 1978. Nitrous oxide emission from soils during nitrification of fertilizer nitrogen. Science (Washington DC) 199, 295-296.

Burger, M., Jackson, L.E., Lundquist, E.J., Louie, D.T., Miller, R.L., Rolston, D.E., Scow, K.M., 2005. Microbial responses and nitrous oxide emissions during wetting and drying of organically and conventionally managed soil under tomatoes. Biol. Fert. Soils 42, 109-118.

Burger, M., Venterea, R.T., 2011. Effects of nitrogen fertilizer types on nitrous oxide emissions. In: Guo, L., Gunasekara, A.S., McConnell, L.L. (Eds.), Understanding Greenhouse Gas Emissions from Agricultural Management. American Chemical Society Symposium Book.

Cai, Z.C., Xing, G.X., Yan, X.Y., Xu, H., Tsuruta, H., Yagi, K., Minami, K., 1997. Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. Plant and Soil 196, 7-14.

California Air Resources Board, 2011. http://www.arb.ca.gov/cc/inventory/data/data.htm

CDFA, 2009. Fertilizing Materials. Tonnage Report. In: Agriculture, CDFA (Ed.), Sacramento CA.

Davidson, E.A., 1992. Sources of nitric oxide and nitrous oxide following wetting of dry soil. Soil Sci. Soc. Am. J. 56, 95-102.

Doane, T.A., Horwáth, W.R., 2003. Spectrophotometric determination of nitrate with a single reagent. Anal. Lett. 36, 2713-2722.

Dobbie, K.E., McTaggart, I.P., Smith, K.A., 1999. Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. J. Geophys. Res. Atmos. 104, 26891-26899.

Dumas, J.B., 1848. Sur les procédés de l'analyse organique. Ann. Chim. , 195-213.

Duxbury, J.M., Bouldin, D.R., Terry, R.E., Tate, R.L., 1982. Emissions of nitrous oxide from soils. Nature (London) 298, 462-464.

Edis, R.B., Chen, D., Wang, G., Turner, D.A., Park, K., Meyer, M., Kirkby, C., 2008. Soil nitrogen dynamics in irrigated maize systems as impacted on by nitrogen and stubble management. Australian Journal of Experimental Agriculture 48, 382-386.

Eichner, M.J., 1990. Nitrous oxide emissions from fertilized soils: Summary of available data. J. Environ. Qual. 19, 272-280.

Forster, J.C., 1995. Soil nitrogen. In: K., A., P, N. (Eds.), Methods in Applied Soil Microbiology and Biochemistry. Academic Press, San Diego, pp. 79-87.

Fujinuma, R., Venterea, R.T., Rosen, C., 2011. Broadcast Urea Reduces N_2O but Increases NO Emissions Compared with Conventional and Shallow-Applied Anhydrous Ammonia in a Coarse-Textured Soil. J. Environ. Qual. 40, 1806-1815.

Garland, G.M., Suddick, E., Burger, M., Horwath, W.R., Six, J., 2011. Direct N_2O emissions following transition from conventional till to no-till in a cover cropped Mediterranean vineyard (Vitis vinifera). Ag. Ecosyst. Env. 141, 234-239.

Halvorson, A.D., Del Grosso, S.J., Reule, C.A., 2008. Nitrogen, tillage, and crop rotation effects on nitrous oxide emissions from irrigated cropping systems. J. Environ. Qual. 37, 1337-1344.

Hartz, T., Miyao, E., Mickler, J., Lestange, M., Stoddard, S., Nuñez, J., Aegerter, B., 1996. Processing tomato production in California. Vegetable Production Series. University of California ANR.

Henault, C., Devis, X., Page, S., Justes, E., Reau, R., Germon, J.C., 1998. Nitrous oxide emissions under different soil and land management conditions. Biol. Fert. Soils 26, 199-207.

Hoben, J.P., Gehl, R.J., Millar, N., Grace, P.R., Robertson, G.P., 2011. Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Global Change Biol. 17, 1140-1152.

Horwath, W.R., 2011. Greenhouse gas emissions from rice cropping systems. In: Guo, L., Gunasekara, A.S., McConnell, L.L. (Eds.), Understanding Greenhouse Gas Emissions from Agricultural Management. American Chemical Society, Washington, D.C.

Hutchinson, G.L., Livingston, G.P., 1993. Use of chamber systems to measure trace gas fluxes. In: Rolston, D.E. (Ed.), Agricultural Ecosystem Effects on Trace Gases and Global Climate Change. ASA Special Publication no. 55, Madison, WI.

Hutchinson, G.L., Mosier, A.R., 1981. Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci. Soc. Am. J. 45, 311-316.

IFA, 2008. International Fertilizer Industry Association Database.

IPCC, 2006. Agriculture, forestry, and other land use. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies, Hayama, Japan, p. 66.

Kallenbach, C.M., Rolston, D.E., Horwath, W.R., 2010. Cover cropping affects soil N_2O and CO_2 emissions differently depending on type of irrigation. Ag. Ecosyst. Env. 137, 251-260.

Kong, A.Y.Y., Fonte, S.J., van Kessel, C., Six, J., 2009. Transitioning from standard to minimum tillage: Trade-offs between soil organic matter stabilization, nitrous oxide emissions, and N availability in irrigated cropping systems. Soil & Tillage Research 104, 256-262.

Lee, J., Hopmans, J.W., van Kessel, C., King, A.P., Evatt, K.J., Louie, D., Rolston, D.E., Six, J., 2009. Tillage and seasonal emissions of CO_2, N_2O and NO across a seed bed and at the field scale in a Mediterranean climate. Ag. Ecosyst. Env. 129, 378-390.

Li, C.S., W. Salas, and M. L. Huertos. 2004. Quantifying carbon dynamics and greenhouse gas emissions in agricultural soils of California: A scoping study. PIER Project Report, P500-04-038. California Energy Commission, Sacramento, California.

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48, 1267-1272.

Linquist, B., van Groenigen, K.J., Adviento-Borbe, M.A., Pittelkow, C., van Kessel, C., 2012. An agronomic assessment of greenhouse gas emissions from major cereal crops. Global Change Biol. 18, 194-209.

Ma, B.L., Wu, T.Y., Tremblay, N., Deen, W., Morrison, M.J., McLaughlin, N.B., Gregorich, E.G., Stewart, G., 2010. Nitrous oxide fluxes from corn fields: on-farm assessment of the amount and timing of nitrogen fertilizer. Global Change Biol. 16, 156-170.

McSwiney, C.P., Robertson, G.P., 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (Zea mays L.) cropping system. Global Change Biol. 11, 1712-1719.

Mikkelsen, R., 2009. The four fertilizer rights: Selecting the right source. Crops Soil May-June, 28-31.

Pang, X.B., Mu, Y.J., Lee, X.Q., Fang, S.X., Yuan, J., Huang, D.K., 2009. Nitric oxides and nitrous oxide fluxes from typical vegetables cropland in China: Effects of canopy, soil properties and field management. Atmos. Environ. 43, 2571-2578.

Pettygrove, G.S., Putnam, D.H., 2009. Legume N credit for crops following alfalfa. University of California Manure Technical Guide Series. University of California.

Putnam, D.H., Russelle, M.P., Orloff, S., Kuhn, J., Fitzhugh, L., Godfrey, L., Kiess, A., Long, R., 2001. Alfalfa, Wildlife and the Environment. California Alfalfa and Forage Association, Novato, CA.

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. Science (Washington DC) 289, 1922-1925.

Rochette, P., Angers, D.A., Belanger, G., Chantigny, M.H., Prevost, D., Levesque, G., 2004. Emissions of N_2O from alfalfa and soybean crops in eastern Canada. Soil Sci. Soc. Am. J. 68, 493-506.

Ryden, J.C., Lund, L.J., 1980. Nature and extent of directly measured denitrification losses from some irrigated vegetable crop production units. Soil Sci. Soc. Am. J. 44, 505-511.

Sanchez-Martin, L., Meijide, A., Garcia-Torres, L., Vallejo, A., 2010. Combination of drip irrigation and organic fertilizer for mitigating emissions of nitrogen oxides in semiarid climate. Ag. Ecosyst. Env. 137, 99-107.

Sanchez-Martin, L., Vallejo, A., Dick, J., Skiba, U.M., 2008. The influence of soluble carbon and fertilizer nitrogen on nitric oxide and nitrous oxide emissions from two contrasting agricultural soils. Soil Biol. Biochem. 40, 142-151.

Simojoki, A., Jaakkola, A., 2000. Effect of nitrogen fertilization, cropping and irrigation on soil air composition and nitrous oxide emission in a loamy clay. Eur. J. Soil Sci. 51, 413-424.

Steenwerth, K., Belina, K.M., 2008. Cover crops and cultivation: Impacts on soil N dynamics and microbiological function in a Mediterranean vineyard agroecosystem. Applied Soil Ecology 40, 370-380.

Stehfest, E., Bouwman, L., 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosyst. 74, 207-228.

Thornton, F.C., Bock, B.R., Tyler, D.D., 1996. Soil emissions of nitric oxide and nitrous oxide from injected anhydrous ammonium and urea. J. Environ. Qual. 25, 1378-1384.

Turini, T., Cahn, M., Cantwell, M., Jackson, L.E., Koike, S., Natwick, E., Smith, R., Subbarao, K., Takele, E., 2011. Iceberg lettuce production in California. Vegetable Production Series. University of California Agriculture and Natural Resources, Richmond, CA.

Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J., Van Kessel, C., 2010. Towards an agronomic assessment of N(2)O emissions: a case study for arable crops. Eur. J. Soil Sci. 61, 903-913.

Venterea, R.T., Burger, M., Spokas, K.A., 2005. Nitrogen oxide and methane emissions under varying tillage and fertilizer management. J. Environ. Qual. 34, 1467-1477.

Venterea, R.T., Dolan, M.S., Ochsner, T.E., 2010. Urea Decreases Nitrous Oxide Emissions Compared with Anhydrous Ammonia in a Minnesota Corn Cropping System. Soil Sci. Soc. Am. J. 74, 407-418.

Wagner-Riddle, C., Thurtell, G.W., King, K.M., Kidd, G.E., Beauchamp, E.G., 1996. Nitrous oxide and carbon dioxide fluxes from a bare soil using a micrometeorological approach. J. Environ. Qual. 25, 898-907.

Wrage, N., Velthof, G.L., van Beusichem, M.L., Oenema, O., 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33, 1723-1732.

Yao, Z.S., Zhou, Z.X., Zheng, X.H., Xie, B.H., Mei, B.L., Wang, R., Butterbach-Bahl, K., Zhu, J.G., 2010. Effects of organic matter incorporation on nitrous oxide emissions from rice-wheat rotation ecosystems in China. Plant and Soil 327, 315-330.

Zou, J.W., Huang, Y., Zheng, X.H., Wang, Y.S., 2007. Quantifying direct N(2)O emissions in paddy fields during rice growing season in mainland China: Dependence on water regime. Atmos. Environ. 41, 8030-8042.

Glossary of Terms, Abbreviations and Symbols

AA	Anhydrous ammonia
AB32	Assembly Bill 32
ANOVA	Analysis of Variance
AS	Ammonium sulfate
CARB	California Air Resources Board
C	Carbon
°C	Degree(s) Celsius
CEC	Cation exchange capacity, the sum of negatively charged sites in soils with the ability of retaining cations (such as ammonium)
CH_4	Methane
CO_2	Carbon dioxide
d	Day(s)
DSCT	Dry-seeded conventional tillage
DSNT	Dry-seeded no-till
DOC	Dissolved organic carbon
ECD	Electron capture detector (in gas chromatographs)
EF	Emission factor, the percentage of fertilizer N emitted as N_2O
GHG	Greenhouse gas
h	Hour(s)
IPCC	Inter-governmental panel of climate change
min	Minute(s)
N	Nitrogen
NH_4^+	Ammonium
$(NH_4)_2 SO_4$	Ammonium sulfate
NO_3^-	Nitrate
N_2O	Nitrous oxide
NPK	Nitrogen phosphorus potassium (fertilizer)
O_2	Oxygen
P	Phosphorus
P5	Mixture of 95% argon and 5% methane used as carrier gas in gas chromatographs

PVC	Poly-vinyl chloride
Tg	Teragram, $1x10^{12}$ gram, million metric ton
U	Urea
UAN32	Urea ammonium-nitrate
RES	Rice Experiment Station
WFPS	Water-filled pore space
WSCT	Wet-seeded conventional tillage
WSNT	Wet-seeded no-till

APPENDIX

More details regarding alternative rice system management that was implemented at the Rice Experiment Station in Biggs, Colusa County, to test different weed control strategies are shown in Figure 1A.

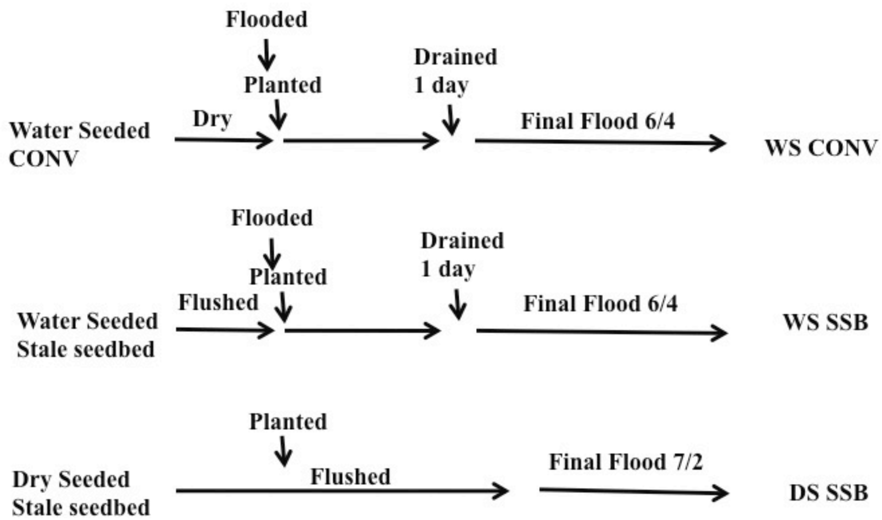


Figure 1A. Timeline of alternative rice cropping system management (WSNT and DSNT) compared to conventional management (WSCT) in 2009.

Table 1A. Annual N_2O emissions and standard errors, calculated by the combined Hutchinson & Mosier (Hutchinson and Mosier, 1981; Hutchinson and Livingston, 1993) and linear method (H&M model) used in the present study and solely by linear regression (Linear Regr.)in tomato, lettuce, wheat, and alfalfa systems.

Crop	**N inputs**	**H&M model**	**Linear Regr.**	**H&M model**	**Linear Regr.**
		Year 1		**Year 2**	
	kg N ha^{-1}	kg N_2O-N ha^{-1} yr^{-1}			
Tomato	0	1.37 ±.27	0.77 ±0.10	0.67 ±.04	0.57 ±0.05
	75	1.33 ±.22	1.20 ±0.08	1.12 ±.08	0.71 ±0.07
	162	2.11 ±.21	1.31 ±0.15	1.50 ±.29	1.22 ±0.23
	225	3.43 ±.80	2.17 ±0.44	4.69 ±.55	3.67 ±0.47
	300	4.38 ±1.61	2.72 ±0.92	4.29 ±.70	3.16 ±0.48
Lettuce	11	0.58 ±.05	0.51 ±0.02	0.59 ±.13	0.42 ±0.06
	84	0.71 ±.07	0.60 ±0.06	0.56 ±.03	0.44 ±0.02
	168	0.69 ±.07	0.57 ±0.05	1.13 ±.20	0.89 ±0.15
	252	1.09 ±.08	0.76 ±0.04	1.14 ±.14	0.78 ±0.05
	336	1.51 ±.27	0.97 ±0.12	1.42 ±.22	1.04 ±0.16
Wheat	0	0.24 ±.07	0.18 ±0.06		
	91	0.31 ±.08	0.26 ±0.08		
	151	0.57 ±.12	0.51 ±0.14		
	203 (AA)	1.31 ±.35	1.09 ±0.25		
	254	0.50 ±.13	0.47 ±0.14		
	0			0.72 ±.22	0.61 ±0.19
	154			0.88 ±.18	0.72 ±0.13
	210			1.42 ±.10	1.21 ±0.10
	210 (AA)			2.05 ±.17	1.88 ±0.23
	266			2.15 ±.23	1.60 ±0.05
Alfalfa	43.1 (5-yr)			5.20 ±.79	4.45 ±0.67
	55.4 (1-yr)			2.30 ±.26	1.96 ±0.24

Figure 2A. Gas chromatograph with autosampler in Dr. Horwath's laboratory.

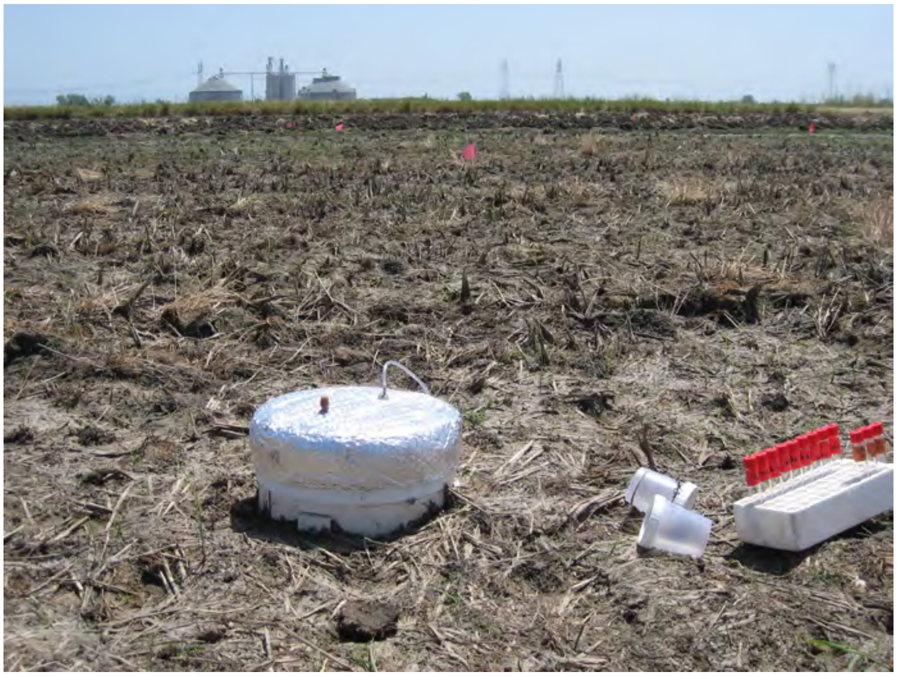


Figure 3A. Flux chamber with sampling port (red septa) and vent tube at the Rice Experiment Station, Biggs, in spring 2009.

Figure 4A. During N_2O flux measurements at the Rice Experiment Station, Biggs, after fall drainage a few weeks before harvest (2009).

Figure 5A. Irrigation manifold with five ports used to inject fertilizer during irrigation (fertigation) at the Hartnell College Ranch, Salinas, where the lettuce crop was fertilized at five different N rates.

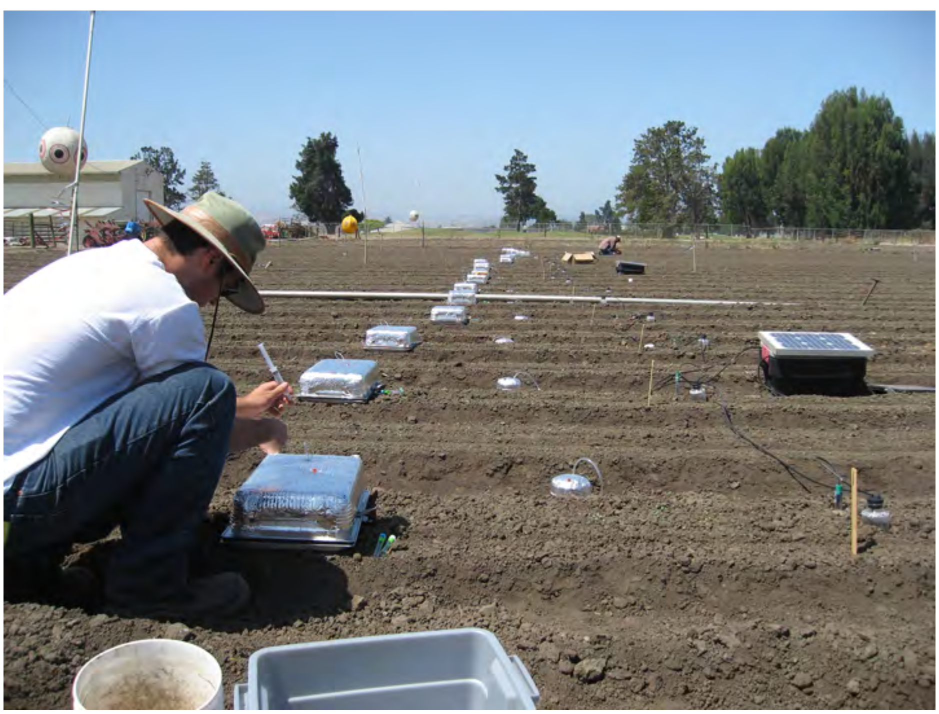


Figure 6A. Experimental set-up of rectangular flux chambers on the beds and smaller, round chambers in the furrows at the Hartnell College Ranch, Salinas.

Figure 7A. Experimental wheat site during aerial application of urea in February 2011. The plot in the foreground, designated as the zero-N plot, was covered with tarps.

Figure 8A. Round PVC chamber base used in the wheat and rice systems.

Figure 9A. Rectangular chamber base and chamber used in tomato and lettuce systems on the beds, and round chamber base and chamber used in the furrows of those systems.

Figure 10A. Bed and furrow chambers closed during flux measurements.

Figure 11A. Chamber base and chamber used in alfalfa systems.