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ORIGINAL ARTICLE

Nitrogen loss and greenhouse gas flux across an intensification gradient in diversified vegetable rotations

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Abstract Vegetable production area is growing rapidly world-wide, yet information on nitrogen (N) losses, greenhouse gas emissions, and input efficiency is lacking. Sustainable intensification of these systems requires improved understanding of how to optimize nutrient and water inputs for improved yields while minimizing N losses. In this study, a 3-year vegetable crop rotation spanning an intensification gradient is investigated in Kentucky, USA: (1) a low input organic (LI), (2) high tunnel organic (HT), and (3) conventional (CONV) system. The objectives were to (1) characterize soil mineral N pools and NO_3^-–N leaching, (2) quantify CO_2 and N_2O fluxes, and (3) relate crop yield to global warming potential (GWP) caused by CO_2 and N_2O losses in these three vegetable production systems. HT maintained consistently higher soil NO_3^-–N; the average NO_3^-–N content during the entire rotations in HT was twice as high as in the CONV and three times as high as in the LI system. Key N loss pathways varied between the systems; marked N_2O and CO_2 losses were observed in the LI and NO_3^- leaching was greatest in the CONV system. The 3-year cumulative CO_2 emission in LI was 50% higher than in the CONV and HT systems. Cumulative N_2O emission over the 3-year vegetable rotations from the LI was twice as high as in the CONV system, whereas 60% more N_2O was produced from the HT than from the CONV system. Yield-scaled GWP was greater in the LI for all crops compared to HT and CONV systems.

Keywords Sustainable intensification · Organic agriculture · CO_2 · N_2O · Global warming potential · Nitrate leaching

Introduction

Meeting society's growing need for food while minimizing harm to the natural resource base upon which food production depends has been characterized as the collective "grand challenge" for agriculture (Foley et al. 2011). There is broad understanding that this challenge must be met largely on existing agricultural lands, through managing natural resources more efficiently than they are currently (FAO 2011; Tilman et al. 2011). Sustainable intensification

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invokes environmental goals such as optimizing the use of external inputs (Matson et al. 1997; Pretty 1997, 2008), increasing rates of internal nutrient recycling, decreasing nutrient loss pathways (Gliessman 2007), and closing yield gaps (Mueller et al. 2012; Pradhan et al. 2015; Wezel et al. 2015). To date, research on intensification efforts has focused largely on staple grain systems. However, efforts to sustainably intensify fruit and vegetable production systems are particularly timely due to the rapid growth of vegetable production area, which has increased 2.6-fold globally in the past 50 years (from 20.5 million ha in 1964 to 55.2 m ha in 2014), with the bulk of this growth occurring since 1980 (FAOSTAT 2018). In addition to growth in production area, agricultural intensification in vegetable crops has increased yields on existing lands through improved irrigation, fertilizer, and pest management practices, and decreasing duration of fallow periods (Stefanelli et al. 2010). The growth of protected agriculture systems has been significant, with nearly 586,300 ha of specialty crop production estimated to be in greenhouses and high tunnels (passive solar greenhouses) (Lamont 2009). As in other parts of the world, in the United States horticultural crops grown in protected culture has grown rapidly, increasing 44% from 2009 to 2014 in the United States (National Agriculture Statistics Service 2014).

Meta-analyses indicate that intensification efforts have resulted in increased yields and decreased labor, as well as improved nutrient and water use overall (Stefanelli et al. 2010). However, there is indication that the effects of intensification efforts may depend on the initial conditions of the systems they seek to improve. For example, in reduced input systems, the effects of intensification are most marked when they correct critically limiting production factors. In this case, relatively minor increases in inputs and subtle modifications of management practices can offer the potential of substantial yield increases (Foley et al. 2011). Such improvements may be particularly pronounced in low-input and organic systems, which frequently experience significant gaps in actualized yield relative to potential yield (yield gaps) (Garbach et al. 2016; Ponisio et al. 2015; Shrestha et al. 2013). However, in highly intensive systems, nutrient losses and declining water and nutrient use efficiency have been observed (Thompson et al. 2007).

In the US, fertilizer inputs in vegetable crops are routinely high; for example, 98 percent of tomatoes grown in the US in 2010 received inputs of $\geq$ 160 kg N ha^{-1} (National Agriculture Statistics Service 2011). At the extreme, these rates may be as high as 1000 kg N ha^{-1} in covered vegetable areas of China (Ju et al. 2007; Zhu et al. 2005). Although increased N fertilization rates have been shown to directly correlate to increase in crop yields in certain crop families (e.g. cole crops), fertilizer N inputs above 150–180 kg N ha^{-1} year^{-1} typically increase leaching rates (Goulding 2000), and extensive NO_3^- leaching has been observed in areas of widespread vegetable production. In addition to NO_3^- leaching, work in grain crop systems has linked exponential increase in N_2O emissions to increased soil available N contents (e.g. Grassini and Cassman 2012; Cui et al. 2013).

There are a number of factors influencing input use efficiency, yield increases, and soil and water quality impacts in vegetable production system, as these systems are highly variable in their management and process-based study is lacking in many systems. Further, processes such as N_2O emissions have been shown to vary not only between climates, but within the same climate between agricultural ecosystems with different management practices as a function of soil N, soil temperature, and soil moisture dynamics (Xu et al. 2016). It is necessary to understand the contribution of vegetable production systems to global greenhouse gas inventories, and to develop strategies to mitigate emissions from vegetable systems (Norris and Congreves 2018). The objectives of this study were to (1) characterize soil mineral N pools and NO_3^-–N leaching, (2) quantify CO_2 and N_2O fluxes, and (3) project crop yield relative to global warming potential (GWP) in three vegetable production systems.

Materials and methods

Research site

This 3-year rotational study was initiated in early spring 2014 at two sites in central Kentucky (1) The University of Kentucky Horticulture Research Farm (UK HRF) in Lexington, KY (37°58′29″N, 84°32′05″W), and (2) a local organic farm in Scott

County, Kentucky (38°13′20″N, 84°30′38″W). Both farms are in the central Bluegrass region of Kentucky, with similar rainfall, temperature, and soil type (Maury silt loam, a fine, mixed, active, mesic Typic Paleudalf). The annual precipitation was 1209, 1475 and 1011 mm, and the average air temperature 12 °C, 13.3 °C and 14.2 °C in 2014, 2015 and 2016, respectively. Each system contained six replicate plots measuring 9 m × 1.5 m. Initial soil conditions for each system are listed in Table 1. Soil pH was measured with a glass electrode in 1:1 soil:water. Soluble salts were analyzed by the electrical conductivity method (Rhoades 1996). Soil P and K were extracted with Mehlich III and analyzed by inductively coupled plasma spectroscopy (Varian, Vista Pro CCD, Palo Alto). Total C and N were analyzed by combustion (LECO Corporation, St. Joseph) (Nelson and Sommers 1982).

Cropping systems

The three vegetable production systems were selected to represent a gradient of intensification, as characterized by duration of fallow periods, tillage intensity, and irrigation and nutrient inputs (Table 2). The Low Input Organic system (LI) consisted of an 8-year rotation beginning with 5-year mixed grass/legume pasture that was rotationally grazed or cut for hay for grass-finished beef and calf production. After the 5-year fallow period, the pasture was broken with deep inversion plowing, disking and surface rototilling to transition fields into a 3-year rotation of annual crops. No supplemental fertilizer was added, and drip irrigation was used exclusively for pepper production, as the crop was grown on plastic mulch. Table beets, collards and beans were produced on bare ground and received only natural rainfall, with no supplemental irrigation. For the past 15 years, the farm has grown diversified organic vegetables in the annual crop portion of the rotation, after transitioning from two generations of conventional tobacco production in a similar rotation. This experiment follows the 3-year vegetable crop rotation.

The two more intensive systems (Conventional and High Tunnel Organic) are representative of common commercial vegetable production systems, and were located at the UK HRF. The Conventional system (CONV) consisted of a winter wheat (*Triticum aestivum*) cover crop terminated with tillage in early spring (Table 2) followed by seasonal annual vegetable production (Table 3). Inputs included mineral fertilizers applied pre-plant and in-season, split-application via fertigation when required for the crop as per commercial vegetable production recommendations for the study region (UK Cooperative Extension Service 2014). Crops were scouted weekly for insects and pathogens, and treated with prophylactic fungicides (pepper and table beets) and insecticides (collards only) according to recommendations. All crops were drip irrigated in every 2–3 days interval in summer and 3–4 days interval in the winter season depending on rainfall.

The High Tunnel Organic system (HT) consisted of three, unheated, replicated 9.1 m × 22 m steel structures with polyethylene film coating, with two plots per tunnel. As is typical for management of these

Table 1 Initial soil conditions at study depths of three study vegetable systems in Kentucky, USA

Agricultural system	Soil depth (m)	Soil pH	Soluble salts (mmhos m^{-1})	Total N (%)	P (kg ha^{-1})	K (kg ha^{-1})	Total C (%)	Soil NO_3^-–N (kg ha^{-1})
Low input organic (LI)	0–0.15	5.42 ± 0.05	18 ± 0.8	0.16 ± 0.010	580 ± 19	196 ± 14	1.75 ± 0.05	61 ± 7
	0.15–0.30	5.36 ± 0.02	13 ± 1.5	0.11 ± 0.005	668 ± 33	170 ± 12	1.31 ± 0.05	39 ± 5
	0.30–0.50	5.29 ± 0.04	10 ± 0.8	0.07 ± 0.004	828 ± 48	163 ± 12	0.87 ± 0.05	27 ± 2
Conventional (CONV)	0–0.15	5.26 ± 0.07	16 ± 3.1	0.07 ± 0.020	153 ± 9	473 ± 28	1.10 ± 0.03	45 ± 8
	0.15–0.30	5.24 ± 0.08	13 ± 1.1	0.08 ± 0.003	134 ± 6	399 ± 10	1.09 ± 0.03	28 ± 2
	0.15–0.30	5.16 ± 0.01	9 ± 0.4	0.07 ± 0.005	92 ± 9	357 ± 14	0.83 ± 0.06	19 ± 1
High tunnel organic (HT)	0–0.15	6.13 ± 0.25	30 ± 8.0	0.13 ± 0.030	254 ± 58	388 ± 61	1.56 ± 0.28	76 ± 14
	0.15–0.30	5.90 ± 0.17	27 ± 6.2	0.13 ± 0.010	237 ± 40	358 ± 13	1.48 ± 0.09	19 ± 3
	0.30–0.50	5.65 ± 0.12	16 ± 1.1	0.09 ± 0.010	173 ± 17	313 ± 54	1.12 ± 0.06	19 ± 1

Table 2 Management characterization of three study agroecosystems in Kentucky, USA

Agricultural system	Cash crop production (typical months/year)	Tillage frequency (approx. depth in m)	Nutrient input regime	Irrigation method and frequency
Low input organic (LI)	8–9	Semi-annual soil preparation with primary inversion tillage (0.30 m), Secondary soil preparation with disc (0.20 m). In-season weed control via sweep cultivation (0.15 m)	Five year fallow prior to cropping cycle, annual cool-season cover crop between cash crops	Drip irrigation in plasticulture beds, applied at time of planting for pepper. Bare ground crops precipitation only
Conventional (CONV)	8–9	Semi-annual soil preparation with a soil spader (historically inversion tillage, 30 cm), secondary soil preparation with disc (0.20 m). In season weed control via sweep cultivation (0.10 m)	Annual cool-season cover crop between cash crops, Synthetic Fertilizer applied pre-plant and split-application via fertigation in long-season crops	Drip-irrigated
High tunnel organic (HT)	12	Quarterly secondary tillage with rototiller (0.20 m). In season weed control via surface cultivation (0.05 m) with hand tools	Semi-annual compost application, pre-plant granular organic fertilizer (pelletized poultry litter-based)	Drip-irrigated

Systems span a gradient of intensification, as characterized by cropping system duration, and tillage, nutrient and irrigation input intensities

structures, crops are grown in soil without supplemental heat or light, and are only passively ventilated through manual opening of doors and side curtains. High tunnel systems are “season extending” technologies used in specialty crop production, allowing for lengthening the growing season of warm-season crops by approximately 1 month each in the spring and fall, and allowing for production of cool-season vegetables throughout the winter in the study region. Also typical to these systems, cover crops are not used, as these intensive production systems often are used for year-round production of high value crops. The use of managed fallows is not considered economically efficient unless they address a production issue, such as pathogen or pest management. Crop residues were removed from the system to minimize pathogen presence. Pre-plant fertilizer consisted of composted horse manure (C:N ratio 25:1) and granular organic fertilizer (Harmony 5-4-3, BioSystems, LLC, Blacksburg, VA) incorporated into the soil before planting at a rate of 67 kg N ha^{-1}, and 45 kg N ha^{-1} respectively. Supplemental fertigation with liquid organic fertilizer (Brown’s Fish Fertilizer 2-3-1, C.R. Brown Enterprises, Andrews, NC) was applied in-season to the sweet pepper crop, at flower initiation and heavy fruit set (twice total) at the recommended rate of 28 kg N ha^{-1} at each fertigation event.

Water was applied in the HT system via drip irrigation, as the plastic cover over the structure excluded all rainfall. All crops were drip irrigated every 2–3 days in summer and weekly in the winter season. The crop rotation and timing of management activities are detailed in Table 3.

Soil sampling

Soils were sampled monthly at 0–0.15, 0.15–0.30, and 0.30–0.50 m depths for mineral N (NH_4^+–N and NO_3^-–N). Three cores were taken in per depth in each plot, homogenized, and bulked for a single analysis per plot. Fresh soil samples were kept refrigerated (~ 4.4 °C) until processing, passed through a 2 mm sieve and processed within 24 h of sampling. Soil mineral N was extracted from a 5 g subsample of fresh soil in 1 M KCl (Rice and Smith 1984) and analyzed via microplate spectrophotometer (Epoch Model,

Table 3 Crop timing and fertilizer rates in three study agroecosystems in Kentucky, USA

System	Crop rotation				
	2014		2015–2016		2016
	Pepper (*Capsicum annum* L., 'Aristotle')	Lettuce (*Lactuca sativa* L., 'Dov')	Beet (*Beta Vulgaris* L., 'Red Ace')	Collard (*Brassica oleracea var. medullosa,* L. 'Champion')	Bean (*Phaseolus vulgaris* L., 'Provider')
Low input organic (LI)					
PD to TD	14 May–9 Sept	–	8 June–3 Sept	11 Oct–23 March	28 May–4 Aug
Conventional (CONV)					
PD to TD	20 May–1 Aug	–	24 April–7 Aug	19 Aug–26 Feb	7 May–26 July
Fertilizer	78 kg N ha^{-1} at planting on 20 May; split application of 9 kg N ha^{-1} on 29 May, 8 June, 16 June, 20 June, 27 June, 9 July	–	56 kg N ha^{-1} at planting on 24 April	56 kg N ha^{-1} at planting on 19 Aug; split application of 9 kg N ha^{-1} on 8 Sept, 15 Sept, 22 Sept, 28 Sept, 2 Oct	56 kg N ha^{-1} at planting on 16 May
High Tunnel Organic (HT)					
PD to TD	22 April–29 July	15 Sept–21 Nov	12 March–12 June	25 Sept–26 Feb	28 April–8 July
Fertilizer	Horse manure compost equiv. to 24 ton ha^{-1}, 45 kg N ha^{-1} of pelleted organic fertilizer (5-4-3) at planting	Same as for pepper	Same as for pepper	Same as for pepper	Same as for pepper

Timing of the crop rotation is detailed by planting date (PD) to final termination date (TD) by mechanical tillage or crop removal, as appropriate to the system and crop. Fertilizer inputs consist of external fertilizer inputs only, and do not include cover crop inputs or background soil mineralization rates. No external fertilizer is used in the low input organic system

BioTek Instruments, Inc., Winooski, VT, USA), after NO_3^--N was reduced using a cadmium reduction device (ParaTechs Co., Lexington, KY, USA) (Crutchfield and Grove 2011).

Ion exchange resin (IER) methods were used to assess net N mineralization via IER resin bags placed at the mid-depth point of the 0–0.15 m and 0.15–0.20 m depths (0.075 m and 0.225 m depths, respectively). Nitrate leaching was assessed using IER lysimeters placed below the plant rooting zone (0.50 m depths). A mixed bed resin was used in both resin bags and lysimeters (Purolite MB400, Bala Cynwyd, PA, USA). IER bags were made from 1000 mm^2 knit swimwear fabric, filled with 1 teaspoon of resin and sealed with a ~ 0.10 m-long cable tie. Resin bags were replaced monthly, at the time of soil sampling. After recovery, resin bags were rinsed of loose soil using DI water, resin mineral N extracted in 2 M KCl, and analyzed by colorimetric analysis, as described above. IER lysimeters were constructed from PVC tubing with 5 cm diameter after the method of Susfalk and Johnson (2002), using 2 teaspoonfuls of resin per lysimeter. Lysimeters were inserted carefully under soil that had not been disturbed through previous excavation by digging a horizontal installation trench approximately 0.20 m perpendicular to the main vertical excavation trench. IER lysimeters were replaced every 3 months, and once recovered, disassembled, with resin mineral N extracted using the 2 M KCl method described above.

Trace gas fluxes (N_2O and CO_2) were measured weekly in 2014 and bi-weekly in 2015 and 2016 (excluding periods when the ground was frozen) using a FTIR-based field gas analyzer (Gasmet DX4040, Gasmet Technologies Oy, Helsinki, Finland). The static chamber method (Parkin and Venterea 2010) was used, with rectangular stainless-steel chambers (0.16 m × 0.53 m × 0.15 m) installed in each plot.

Chambers were installed after planting of initial crops in the rotations, and kept in the soil for the duration of the 3-year study, except during tillage operations. When pans were removed periodically, chambers were replaced at least 24 h prior to sampling events. At the time of gas sampling, the gas analyzer was connected to the field chamber by affixing a matching rectangular gas pan connected to the analyzer, clamped tightly in place, and measured continuously for ten minutes. The gas fluxes were calculated by using the following equation (Iqbal et al. 2013):

$$(F) = \frac{\Delta C}{\Delta t}\frac{V}{A}\rho$$

where F is the gas flux rate (mg m^{-2} h^{-1}), ΔC/Δt indicates the increase/decrease of gas concentration (C) in the chamber over time (t), V is the chamber volume (m^3), A is the chamber cross-sectional surface area (m^2), ρ denotes density of gas (kg m^{-3}) at 25 °C. Cumulative gas fluxes were estimated by interpolating trapezoidal integration of flux versus time between sampling dates and calculating the area under the curve (Venterea et al. 2011).

Soil water potential was measured using granular matrix sensors (Watermark, Irrometer Co., Riverside, CA, USA) installed at three depths in the soil profile (0.10, 0.30, and 0.50 m depths), with one sensor per depth and per plot. Watermark sensor data was transmitted continuously to a wireless data logger (Watermark Monitor 900 M, Irrometer, Co., Riverside, CA, USA), with readings taken each time when water potential changed. Additional hand-made tensiometers were constructed of 21.5 mm diameter plastic pipe with 22.2 mm diameter ceramic porous cups at the lower end and installed at 0.10 m, 0.30 m, 0.50 m and 0.70 m depth in each plot. Tensiometer readings were taken weekly using a digital Tensimeter (Soil Measurement System, Tucson, AZ, USA). The soil water potential data from watermark sensors and tensiometers were converted to volumetric soil water content (m^3 m^{-3}) using the van Genuchten (1980) equation:

$$\theta = \theta r + \frac{\theta s - \theta r}{\left[1 + |\alpha h|^{n}\right]^{m}}$$

where $\theta r = 0.067$, $\theta s = 0.45$, $\alpha = 0.02$, $n = 1.41$, $m = 1 - 1/n$ for silty loam for all depths (van Genuchten et al. 1991).

Plant sampling

Fresh vegetable yields were measured from the entire plot area from each plot. Pepper fruit, collard leaves and beans were harvested at multiple times as the harvestable portion reached marketable stage. Table beets were harvested once, as roots reached marketable size. Plant biomass samples were collected from two 0.25 m^2 samples per plot at the end of the growing season, dried at 60 °C until a constant mass was achieved. Dried samples were homogenized on a Wiley Mill and a subsample ground on a jar mill (U.S. Stoneware, East Palestine, OH). Crop plant samples were analyzed for C and N content via flame combustion (Flash EA 1112 elemental analyzer, CE Elantech Inc., Lakewood, CA).

Data analysis

Shapiro–Wilk's W-test was used to test for normality of data. CO_2 and N_2O flux data were log-transformed to meet normality conditions. Non-parametric Spearman rank correlations were conducted using JMP Pro 13.2 (SAS Institute, Cary, NC, USA) for CO_2, and N_2O fluxes with soil temperature and soil mineral N content.

Direct N_2O emissions were converted to global warming potential (GWP) units of carbon dioxide equivalents (CO_2 eq) within a 100-year horizon by multiplying by a radiative forcing potential equivalent to CO_2 of 298 for N_2O (IPCC 2001). Cumulative greenhouse gas emission CO_2 eq was calculated by adding cumulative CO_2 eq N_2O and CO_2 emissions. Cumulative GHG emissions (ton CO_2 eq ha^{-1}) divided by crop yield (ton ha^{-1}) equaled yield-scaled GWP (ton CO_2 eq ton^{-1} crop yield) for a crop growing season (Schellenberg et al. 2012).

Results

Time series data by system

Low input organic system

Soil NO_3^-–N and NO_3^-–N leaching rates, were consistently greatest in the LI system at the start of the rotation. After this initial period of high soil mineral N (NH_4^+–N and NO_3^-–N) content, values were low compared to the other systems and peaked seasonally at each sampling depth in late spring of each year. Peak soil mineral N content in the surface layer (0–0.15 m) was 64 kg ha^{-1} in May 2014, 55 kg ha^{-1} in April 2015, and 50 kg ha^{-1} in June 2016 (see Fig. 1d for additional depths). It is of note that NH_4^+–N peaked at the end of the rotation (2016), corresponding to the bean crop portion of the rotation. During this peak, NH_4^+–N content was greater than ~ 40% of total mineral N, but was typically less than 12% during the remainder of the rotation, excepting for seasonal peaks in the early spring. It is interesting that soil NH_4^+–N was lower than 2 kg N ha^{-1} in the LI system for the entire crop growing season from 2014 to 2016 except for the sampling campaign after collard harvest when it was > 20 kg N ha^{-1} (Fig. 1e). Soil and IER mineral N values were lower in the LI system compared to other systems at the majority of the sampling dates (Fig. 1f). The IER data indicate the mineralization rates were low at these

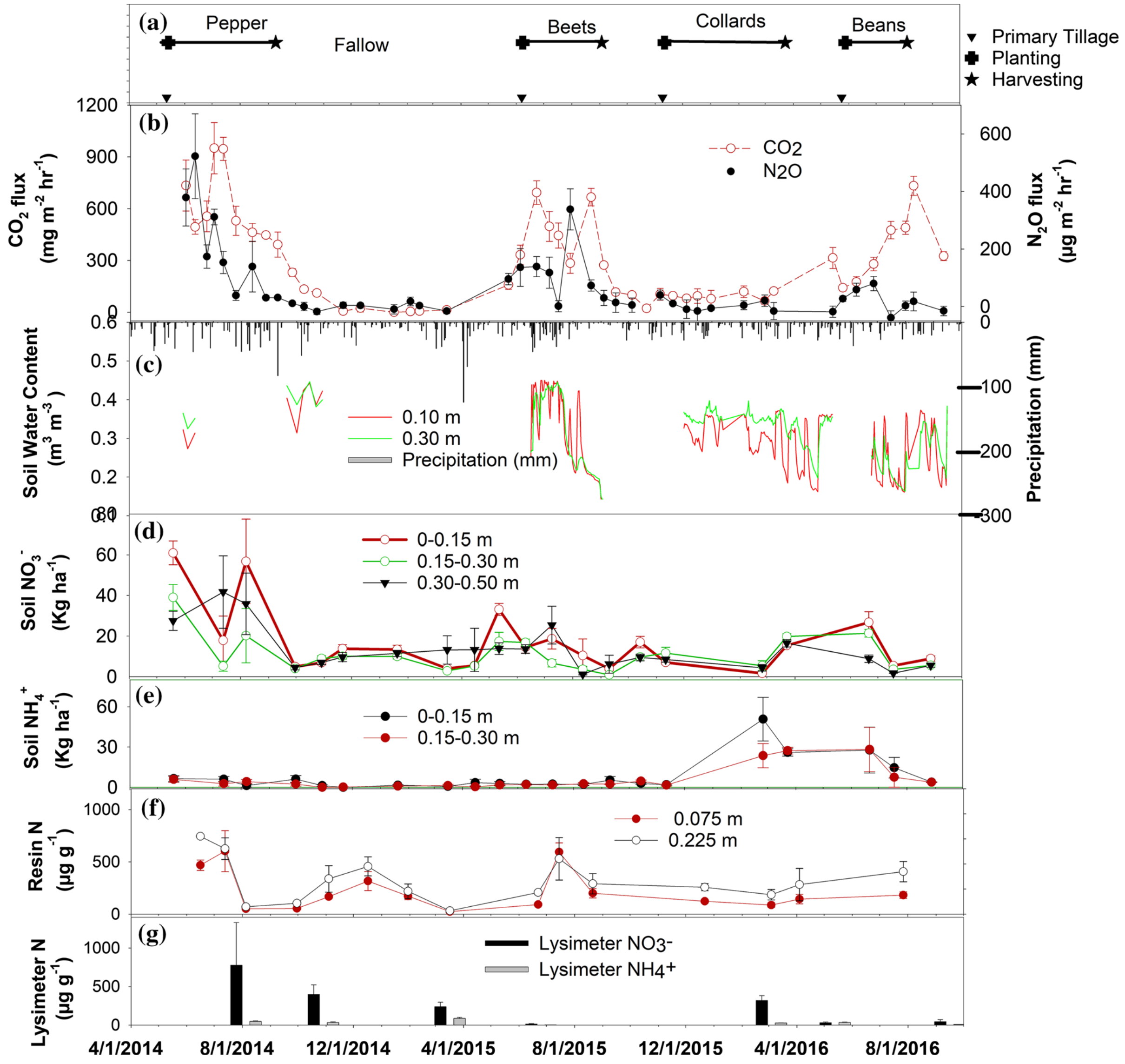


Fig. 1 Time series data from the Low Input Organic (LI) system from 2014 to 2016, including CO_2 and N_2O flux, soil water content and precipitation, and soil NH_4^+–N and NO_3^-–N, total mineral N extracted from ion exchange resin bags, and leaching losses measured via ion exchange resin lysimeters

times as well, with monthly IER resin bag values < 200 μg g^{-1} resin. Cumulative mineral N adsorbed to IER resin bags and IER lysimeters are presented in Table 4. The highest lysimeter NO_3^-–N was observed during the first growing season (pepper crop, 2014) and decreased in subsequent years. Soil volumetric water content was consistently driest in the LI system compared to the other two study systems, due to the sparse irrigation regime in the LI system.

CO_2 fluxes were seasonally-dependent and significantly correlated to soil temperature (Table 5). The largest CO_2 flux rates were observed in mid-summer each year, on 2 July 2014 (950 mg m^{-2} h^{-1}), 22 June 2015 (732 mg m^{-2} h^{-1}) and 10 August 2016 (732 mg m^{-2} h^{-1}) (Fig. 1b). CO_2 fluxes were negligible from November to early April each year. Similarly, N_2O fluxes were seasonally-influenced, with peak rates typically occurring after rainfall or irrigation events, early in summer as soils warmed and after tillage events. Peak daily N_2O fluxes occurred on 11 June 2014 (522 μg N m^{-2} h^{-1}), 29 June 2015 (393 μg N m^{-2} h^{-1}), and 8 June 2016 (58 μg N m^{-2} h^{-1}). N_2O emissions were not strongly correlated with soil mineral N, soil temperature, or soil water content values. As with soil mineral N content, fluxes and peak fluxes declined over the 3-year rotation.

Conventional system

Soil mineral N content in the CONV system was seasonally-dependent, with peak values at the beginning of each cropping season (Fig. 2d, e). Soil NO_3^-–N levels remained consistently at peak levels throughout the pepper growing season due to regular application of soluble inorganic fertilizer applied through the drip irrigation lines (Fig. 2d, Table 3). Peak values declined over the duration of the rotation, concomitant with decreasing quantities of fertilizer applied for the crops in the rotation. The highest observed soil mineral N contents were 170 kg ha^{-1} in May 2014, 51 kg ha^{-1} in June 2015, and 26 kg ha^{-1} in June 2016. As in the LI system, the relative percentage of NH_4^+–N in total mineral N (NH_4^+–N + NO_3^-–N) was greater than 30% of the overall mineral N composition in soil and IER bag samples at the majority of the sampling dates throughout the rotation (Fig. 2f). In the CONV system, the highest lysimeter NO_3^-–N values were also observed during the pepper crop.

Soil moisture content in the CONV system exhibited some drying at the 0.10-m-depth, but was generally consistently between field capacity and saturation for the silt loam soil type (field capacity = 0.29 m^3 m^{-3}; saturation = 0.43 m^3 m^{-3}). This relatively high soil water content is reflective of precipitation and regular irrigation inputs consistent with commercial vegetable production recommendations (UK Cooperative Extension Service 2014).

CO_2 fluxes in the CONV system were seasonally-dependent, and were correlated to soil temperature (Table 5) although the correlation was weaker than in the other two systems. The greatest CO_2 fluxes were observed in mid-summer each year, with the greatest fluxes on 23 June 2014 (428 mg m^{-2} h^{-1}), 6 July

Table 4 Average soil NO_3^-–N and NH_4^+–N content, cumulative mineral N trapped in resin bags and Lysimeter, and cumulative CO_2 and N_2O fluxes for the entirety of the rotation from 2014 to 2016 in three vegetable production systems

Parameters	Unit	Low input organic system	Conventional system	High tunnel organic system
Average soil NO_3^-–N (0–0.15 m)	kg ha^{-1}	18 ± 2.3	32 ± 1.3	64 ± 6.6
Average soil NO_3^-–N (0.15–0.30 m)	kg ha^{-1}	11 ± 0.5	20 ± 1.2	21 ± 2.4
Average soil NO_3^-–N (0.30–0.50 m)	kg ha^{-1}	14 ± 1.4	22 ± 0.8	42 ± 4.1
Average soil NH_4^+–N (0–0.15 m)	kg ha^{-1}	6 ± 01.7	4 ± 0.3	5 ± 0.8
Cumulative resin mineral N at 0.075 m	g kg^{-1} resin	3.2 ± 0.1	6.5 ± 0.7	2.7 ± 0.3
Cumulative resin mineral N at 0.225 m	g kg^{-1} resin	3.5 ± 0.5	8.9 ± 0.6	3.8 ± 0.3
Cumulative lysimeter NO_3^-–N	g kg^{-1} resin	1.8 ± 0.4	2.3 ± 0.4	2.1 ± 0.5
Cumulative CO_2 flux	ton CO_2–C ha^{-1}	12.8 ± 0.1	8.4 ± 0.3	8.3 ± 0.3
Cumulative N_2O flux	kg N_2O–N ha^{-1}	5.7 ± 0.7	2.8 ± 0.3	4.5 ± 0.9

Table 5 Spearman rank correlation values for nitrous oxide flux and soil total mineral nitrogen (NO_3^-–N and NH_4^+–N) and soil temperature, and carbon dioxide flux and soil temperature in three study vegetable production systems in Kentucky, USA

Environmental variables	Low input organic	Conventional	High tunnel organic
	N_2O		
Soil mineral N (0–0.15 m)	r = 0.30	r = 0.08	r = 0.20
Soil mineral N (0.15–0.30 m)	r = 0.14	r = 0.02	r = 0.32
Soil mineral N (0.30–0.50 m)	r = 0.28	r = 0.12	r = 0.13
CO_2	r = 0.46	r = 0.26	r = 0.16
Soil temperature (°C, 0.10 m)	r = 0.35	r = 0.07	r = 0.15
	CO_2		
	r = 0.80	r = 0.55	r = 0.55

2015 (511 mg m^{-2} h^{-1}) and 10 August 2016 (478 mg m^{-2} h^{-1}). CO_2 fluxes were negligible from November to early April in 2014, and low but with occasional fluxes during the same period in 2015, likely due to warmer soil temperatures and more moderate temperatures in winter of 2015. N_2O fluxes were seasonally-influenced, with peak rates typically occurring early in the cropping season, coinciding with pre-plant tillage and fertilizer incorporation. N_2O fluxes in the CONV system were the lowest of the three systems, with daily peak values occurring on 31 May 2014 (65 µg m^{-2} h^{-1}), 22 April 2015 (145 µg m^{-2} h^{-1}), and 8 June 2016 (144 µg m^{-2} h^{-1}). It is notable that after peak N_2O events, low and even negative fluxes were observed.

High tunnel organic system

Soil mineral N content remained consistently higher in the HT system than in the other studied systems throughout the experiment. At the surface (0–0.15 m) layer, the average NO_3^-–N content in the HT systems was twice that of the CONV system, and three times higher than in the LI system (Table 4), when averaged across the rotation. Average soil NO_3^-–N at the lowest sampled depth (0.30–0.50 m) was markedly greater in the HT system. Similar to the other systems, mineral N decreased over the duration of the rotation. The largest soil mineral N contents were observed after fertilization events in May 2014 (147 kg ha^{-1}), September 2014 (198 kg ha^{-1}), June 2015 (91 kg ha^{-1}), and June 2016 (57 kg ha^{-1}) (Fig. 3d). The highest lysimeter NO_3^-–N was found during the beet growing season. In the HT system, the NH_4^+–N contents were higher than 30 kg N ha^{-1} after each compost and organic fertilizer application (Fig. 3e).

Soil water content in the HT system fluctuated between saturation and 75% of field capacity during active crop production in the structures. Soil water content was solely representative of irrigation inputs, as rainfall was excluded in this system. When fallow, soils were not irrigated and exhibited soil water content as low as ~ 0.2% VWC for the 2 week–3-month fallow periods (Fig. 3c).

Peak CO_2 fluxes in the HT system were comparatively lower than in the other systems, and occurred ~ 1 month earlier than in the open field systems. The CO_2 flux was well correlated with soil temperature (Table 5). Peak CO_2 fluxes occurred on 23 June 2014 (274 mg m^{-2} h^{-1}, 8 May 2015 (313 mg m^{-2} h^{-1}), and 8 June 2016 (303 mg m^{-2} h^{-1}). However, CO_2 fluxes were consistently higher in the HT system than in the open field systems during the winter months, and correlated with higher soil temperatures in the HT structures. Similarly, N_2O emissions were greater than in the other systems during the winter months, although fluxes were still low, even given the relatively high mineral N content throughout the soil profile. Peak annual N_2O flux coincided with tillage and incorporation of pre-plant fertilizer. Peak N_2O fluxes occurred on 26 June 2014 (95 µg m^{-2} h^{-1}), 29 July 2015 (257 µg m^{-2} $h^{-1)}$ and 8 June 2016 (153 µg m^{-2} h^{-1}).

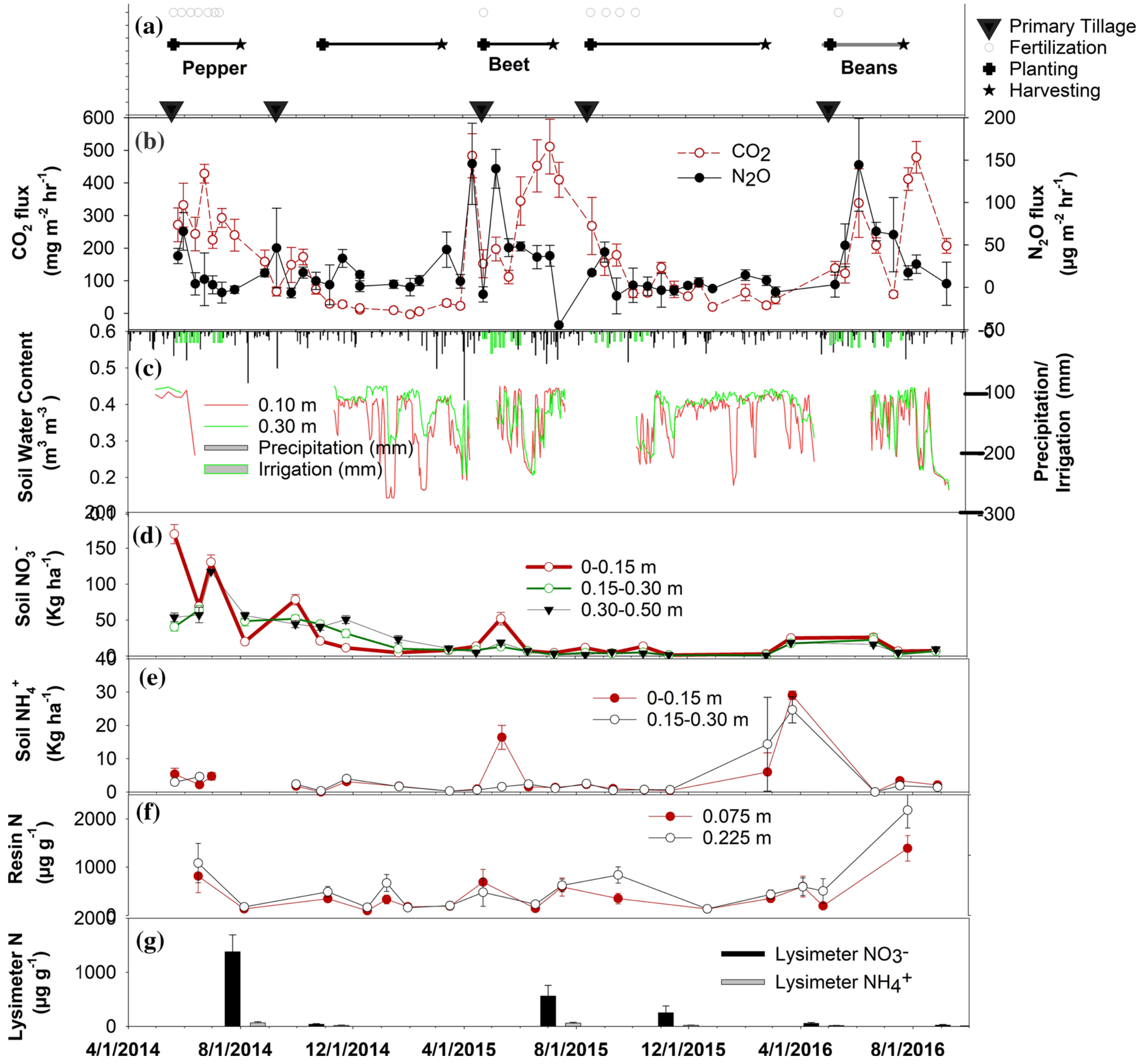


Fig. 2 Time series data from the Conventional (CONV) system from 2014 to 2016, including CO_2 and N_2O flux, soil water content and precipitation, and soil NH_4^+–N and NO_3^-–N, total mineral N extracted from ion exchange resin bags, and leaching losses measured via ion exchange resin lysimeters

Systems-level measures

Cumulative CO_2 and N_2O fluxes

Cumulative CO_2 flux, calculated for length of the entire rotation, was greatest in the LI system, and did not differ appreciably between the CONV system, and the HT system (Table 4). Cumulative N_2O emissions in the LI systems were double that of the CONV system. Mean cumulative N_2O emissions were 60% greater in the LI system as compared to the HT system, though these organic systems did not differ substantially when accounting for the standard error in the cumulative flux measures (Table 4).

Cumulative fluxes are presented in CO_2 equivalents (ton CO_2 equivalents ha^{-1}) in Fig. 4a. These results demonstrate that the greatest differences between systems result from the relatively large fluxes resulting

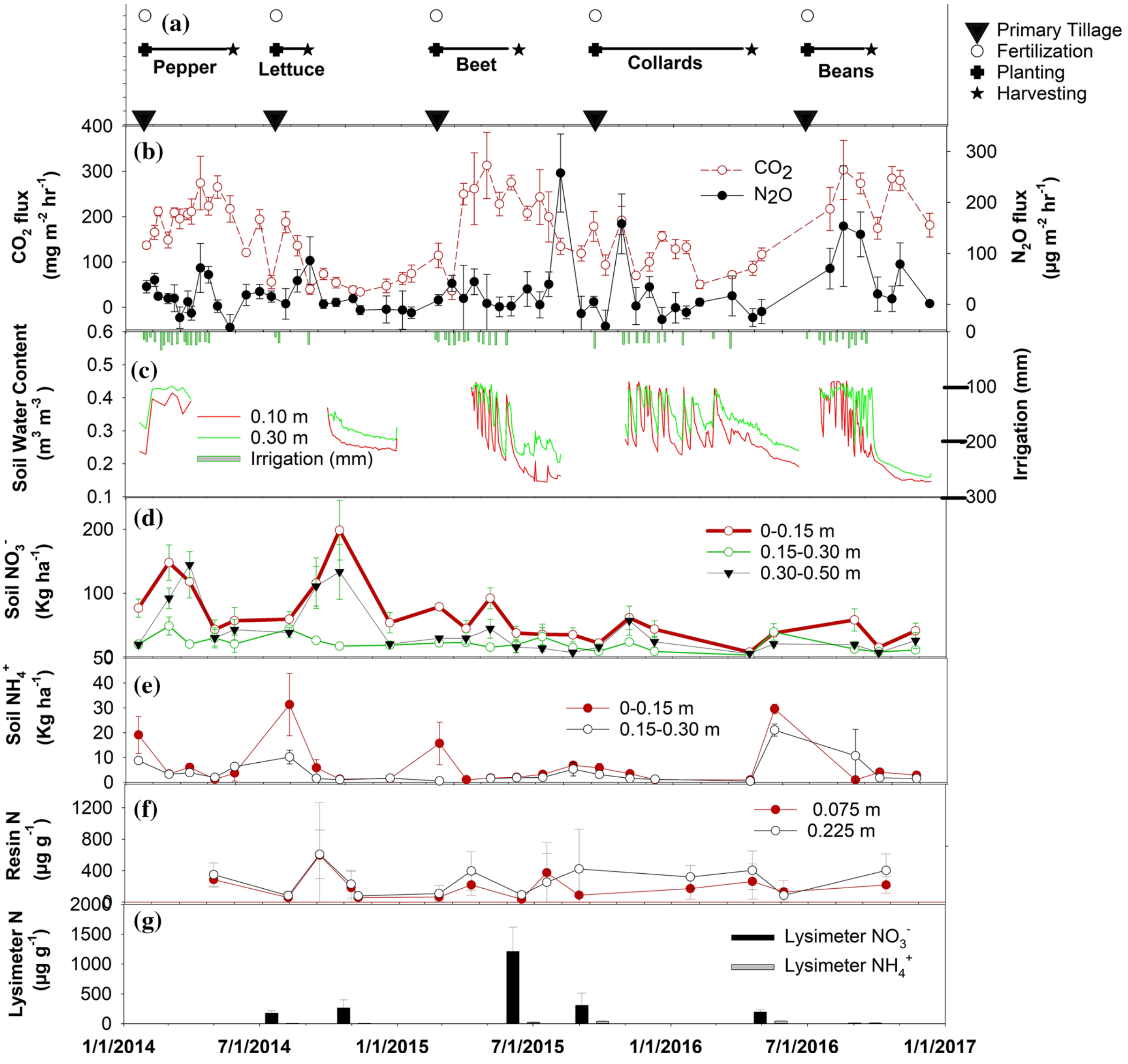
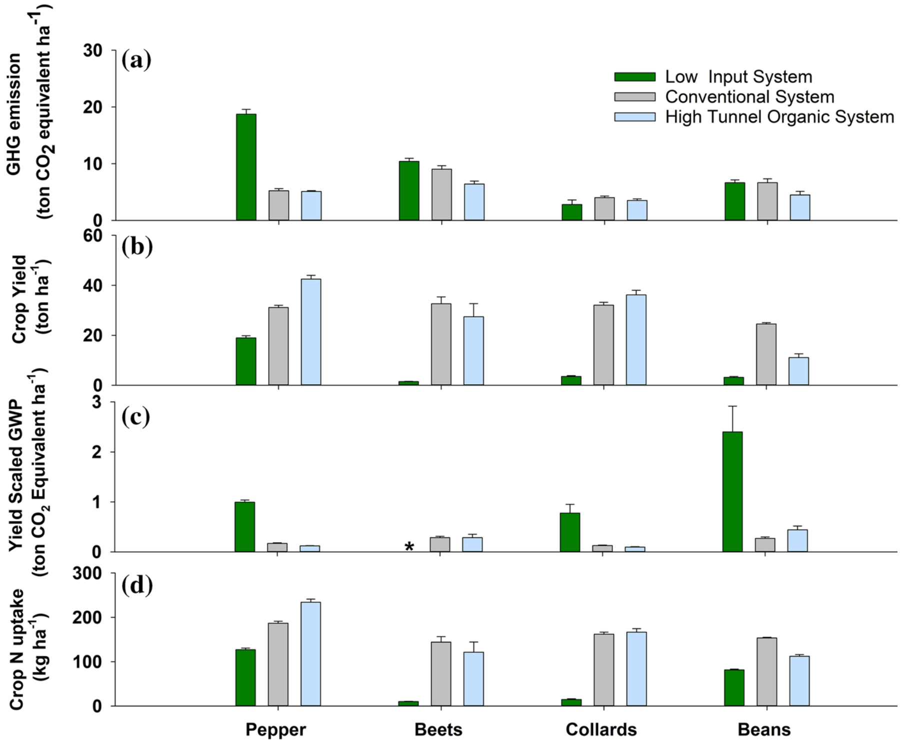


Fig. 3 Time series data from the High Tunnel Organic (HT) system from 2014 to 2016, including CO_2 and N_2O flux, soil water content and precipitation, and soil NH_4^+–N and NO_3^-–N, total mineral N extracted from ion exchange resin bags, and leaching losses measured via ion exchange resin lysimeters

from the initial pasture conversion in the LI system. Differences between systems are reduced substantially after the first 1.5 years of the rotation (Fig. 4a).

Yield and yield-scaled global warming potential

Crop yields were consistently lowest in the LI system (Fig. 4b). Yields were similar between the CONV and HT systems. Yield-scaled GWP, a measure relating yield to cumulative GWP, demonstrated consistently greater GWP per unit of yield in the LI system (Fig. 4c). These results in the LI system are driven both by greater fluxes in this system at the beginning of the rotation, following the conversion from pasture to vegetable crops, as well as by relatively low yields on a per-area basis.



* Yield Scaled GWP of LI beets (7.2 ±0.6) not included in graph for better view.

Fig. 4 Systems-level comparison of crop yields, cumulative trace gas flux, and crop N uptake by crop in the 2014–2016 crop rotation including **a** cumulative greenhouse gas (GHG) emission (ton CO_2 equivalent ha^{-1}), **b** crop yield (ton ha^{-1}), **c** yield-scaled global warming potential (GWP) (ton CO_2 Equivalent ton^{-1}), and **d** crop N uptake (kg ha^{-1})

Discussion

Environmental and management effects on soil processes across systems

Soil mineral N

Soil N dynamics have been shown to be highly variable between production systems, based on differing fertilizer inputs and soil and water management (Eichner 1990). Time-series data (Figs. 1, 2, 3) demonstrate that management practices such as tillage, fertilizer application, and irrigation management affected soil mineral N content in each system. However, the extent to which these activities drive peak mineral N content vary by system. All systems in this study demonstrated increased soil NO_3^-–N content after tillage events (Figs. 1d, 2d, 3d), but the effect was most pronounced in the LI system. Large peaks of soil NO_3^-–N content were observed in the LI system after the primary tillage/pasture conversion in 2014, and is also associated with high initial total soil N and C concentrations (Table 1). Although soil NO_3^-–N content decreased in subsequent years, soil NO_3^-–N peaks were observed after each tillage event throughout the study. Similarly, Eriksen and Jensen (2001) reported increase in soil inorganic N content following cultivation of pasture land in the early crop growing season when crop N uptake is insignificant.

Fertilizer application is largely responsible for soil mineral N peaks in the CONV and HT systems, though the effects are somewhat different due to the nature of the fertilizer sources and soil water dynamics. In the CONV system, fertilizer was regularly applied through split applications of soluble fertilizer applied through irrigation lines. Soil NO_3^-–N levels remained relatively stable for the majority of crops in the rotation, with the exception of the pepper crop. As discussed in further detail below, these results indicate that for many crops fertilizer application rates and methods may be reasonably well-timed with crop uptake. However, in this open-field system leaching losses may flush excess nutrient salts below the crop rooting zone. The HT soils maintained consistently greater levels of mineral N in all soil layers (Table 4), likely due to the lack of rainfall that would have leached the NO_3^-–N in this system to deep soil layers (Zikeli et al. 2017). Further, the highest levels of soil

NH_4^+–N were observed in the HT system after each compost and organic fertilizer application, likely due NH_4^+–N release from compost and manure decomposition that was later nitrified into NO_3^-–N form (He et al. 2000).

IER resin bag values were not well correlated with soil mineral N content ($R^2 < 0.12$ for top 30 soil layer for all system) nor with driving abiotic parameters such as soil water. In several studies, particularly for surface soil layers, strong correlations were found between resin N and soil water content (Binkley and Matson 1983), mineral N (Kramer et al. 2006) and soil temperature (Johnson et al. 2005), respectively. Our results are consistent with studies in which no correlation between resin N and soil mineral N pools (Hanselman et al. 2004; Johnson et al. 2005) or soil water content (Allaire-Leung et al. 2001) were detected. It should be noted that resin N content revealed less variability within and between systems than soil samples. This result may indicate that in this application, resin N was a less sensitive methodology in detecting changes in soil mineral N pools than soil sampling because mineral soil N can either be taken up by plants, leached or adsorbed in resin bags. Moreover, it is possible that the resin bag-based results were affected by insufficient nutrient desorption from the resin material that would perhaps have benefited from a series of KCl extractions (Kolberg et al. 1999) compared to a single extraction.

The NO_3^-–N leaching measured via IER resin lysimeters, largely during pepper growing season may be explained by high soil NO_3^-–N status generated by N mineralization of incorporated residue in the LI and fertigation in the CONV system (Fig. 2g). These losses mainly occurred after planting of crops as the small seedlings were unable to capture the fertilizer applied and could not consume more water during the early growth stage (Errebhi et al. 1998). One reason for high leaching despite low soil mineral N in the LI system might be poor plant establishment due to lack of irrigation and fertilizer at the beginning. Poor crop establishment not only resulted in poor crop yield, but also increased losses of N, that otherwise could have been utilized by plants if their stand had been better.

Trace gases

CO_2 fluxes were well correlated with soil temperature in all systems ($R^2 > 0.55$) (Table 5), which has been reported by many researchers (e.g. Case et al. 2012; Chen et al. 2015). This is notable, however, as the three systems in this study differed substantially in tillage regime, fallow management, and inputs. Except for the initially large fluxes in the LI system at the beginning of the rotation after inversion tillage and breaking of the pasture fallow, annual CO_2 peaks were not substantively different from those measured in the CONV system after 2 years. This may indicate that within annual vegetable production systems, CO_2 flux may be more affected by climate and soil type than by management within a given region (Raich and Schlesinger 1992).

N_2O fluxes were not consistently well-correlated to any single abiotic factor, but did peak seasonally in the mid-late summer with mid-season peaks after fertilizer and tillage events in all systems. Although N_2O flux was not well-correlated with soil mineral N content, as found in other studies (e.g. Deng et al. 2015), soil mineral N is a necessary pre-cursor to N_2O emissions. It is notable that the HT system lacked large N_2O peaks, despite maintaining consistently higher soil mineral N content throughout the soil profile, as compared to the two other systems. In the protected environment of the HT, where water was only provided via drip irrigation, soils were never exposed to saturation from rainfall and soil water content fluctuated less and on a shorter time scale during crop production cycles than in other systems (Fig. 3c). This may have created conditions less favorable to denitrification (Sanchez-Martin et al. 2010), and thereby avoided the high N_2O fluxes that are sometimes measured in open field conditions after natural rainfall (Jamali et al. 2016). Additionally, HT system during fallow period might be attributed to an overriding effect of dry soil moisture conditions on N_2O emissions in N-fertilized vegetable soil even though enough soil N substrate was present (Xu et al. 2016). These results demonstrate the interactive (and sometimes restrictive) effect of temperature and soil moisture content on N_2O emissions across agroecosystems with variable management regimes (Xu et al. 2016).

N_2O flux was strongly correlated with CO_2 flux in the LI system (Table 5). Additions of crop residues have been shown to not only stimulate microbial respiration, but also to enhance oxygen depletion by stimulating microbial respiration and promoted anaerobic conditions for triggering denitrification and N_2O

production (Chen et al. 2013). Further, N_2O production has been shown to increase substantially after organic matter additions (Deng et al. 2013; Thomas et al. 2008) and after incorporation of crop residues which may increase soil water content, accelerating N_2O emissions in residue-amended soil (Kravchenko et al. 2017). Regardless of the mechanism, increased trace gas flux, and C and N mineralization following extensive tillage and pasture-conversion are well-documented (e.g. Pinto et al. 2004). The effects may be short-lived (Pinto et al. 2004), but may have profound effects on the cumulative emissions of the subsequent crop. In the LI system in this study, 25% of the cumulative emissions for the 2-year study accumulated in the first month of measurement following pasture conversion.

Sustainable intensification of horticultural systems

This systems-level comparison is limited to a 3-year sampling period and does not include all parameters that might be associated with the sustainability intensification of horticultural farming systems, such as energy returned on energy invested (Schramski et al. 2013), irrigation water use efficiency (Mueller et al. 2012), or other interdisciplinary, holistic measures of agroecosystem sustainability. However, nutrient uptake and losses data demonstrate that paths to sustainably intensifying horticultural systems may vary by system due to the highly variable nature of inputs and environmental factors.

Crop yields

As this work was conducted in a systems context, yield data (as all other parameters) are compared between systems as a function of a combination of factors in a system, not any one particular input or management scenario. As such, mechanisms discussed here as they relate to yield differences are presumed to be contributing factors, but not sole drivers of differences in crop yields.

The difference in yield of some crops between the HT and CONV systems may be explained in part by differences in crop sensitivities to the inputs or environmental factors. The HT system exhibited greater pepper yield than the CONV system, as this crop has been shown to benefit from the protective cover of the structure in decreasing fungi-foliar disease incidence (Powell et al. 2014). The CONV system had greater bean yields, which may be due to flower drop due to higher daily max temperatures (Monterroso and Wien 1990) during summer in the HT system. It is notable that the HT system, an intensively-managed, organic production system, did not experience a “yield gap” when compared to the CONV system, which is a commonly observed phenomenon in organic production systems (e.g. Seufert et al. 2012; de Ponti et al. 2012).

Yields in the LI system were highly variable across the rotation, with two of the crops experiencing near crop-failures (table beets and collards) due to poor crop establishment and weed pressure. Lower yields in the LI system may be due in part to stunted crop growth during establishment, as crops were not typically irrigated or provided fertilizer at transplant. Although, mineralized N was available later in the growth cycle it is possible that this could not be utilized by stunted plants. Yield data in the LI system are consistent with others that show that low-input organic farming systems may be good candidates for sustainable intensification (Garbach et al. 2016; Ponisio et al. 2015).

Yield-scaled impacts

The LI system exhibited a much greater GWP per unit yield for each crop in the rotation (Fig. 4c) due to low yields and higher greenhouse gas fluxes. However, after the initial pasture conversion in the LI system, trace gas fluxes do not differ greatly from the other systems. Rather, higher yield-scaled GWPs in the LI system are a function of substantially lower crop yields. Crop growth may have been limited by N availability and dry soil water conditions, particularly during crop establishment. Weed pressure, though not measured here, was observed to be greater in this system and was pronounced in the beet and bean crop.

These findings suggest that relatively minor changes to the LI system - such as increases in irrigation at critical times (e.g. during establishment) more efficacious weed management, or small applications of fertilizer at critical crop phenological stages - may have strong influence on yields and reduce yield-scaled GWP values. Further, such efforts to optimize yields through targeted inputs may allow for decreased area needed for vegetable production, allowing for increased length of the fallow portion of the rotation or other land-sparing efforts, thus decreasing the substantial leaching and trace gas losses after pasture conversion. Additional research is needed to evaluate how rotations incorporating long-term pastures and/or grazed fallows, and other low external input measures can be optimally managed to sustain soil while minimizing nutrient and carbon losses. Relatively high leaching rates from CONV system indicate that additional system-specific research on fertilizer rate, timing, and irrigation practices in the study region may be warranted to sustainably manage CONV vegetable production.

The HT and CONV systems, which were more intensive than the LI system by comparison, did not differ greatly in yield-scaled GWP, nor did the organic HT system experience a yield-gap with the conventional, as described above. However, it is of note that HT infrastructure can be costly and management-intensive. Although the exclusion of rainfall allows for more precise control of the soil water environment and reduce disease incidence, irrigation and daily monitoring of temperature and ventilation are required. Management inputs and economic aspects are not measured in this project, but can limit the scalability of HT systems.

Conclusions

This study quantified the soil mineral N dynamics, CO_2 and N_2O fluxes, and yields from a suite of diversified vegetable systems representing a gradient of input and management intensification. Key loss pathways in the Low Input Organic (LI) system were via greenhouse gas fluxes, whereas in the Conventional (CONV) system they were via leaching. Although the High Tunnel Organic (HT) system was expected to produce higher gas fluxes than in the other two systems, this was not observed, although the peak timing and basal flux patterns differed from the open field systems. Yield-scaled GWP was greater in the LI system compared to CONV and HT system, driven both by greater fluxes as well as lower yields. From the perspective of sustainable intensification in these three systems, our study suggests CONV systems may benefit from reduced fertilizer inputs in combination with irrigation management to minimize downward directed hydraulic gradients particularly just after planting of crops; LI systems may benefit from targeted additional fertilizer and irrigation inputs; and this work supports literature indicating the need to examine long-term soil impacts in HT systems over longer timelines.

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