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ORIGINAL ARTICLE

Soil greenhouse gas fluxes from tropical vegetable farms, using forest as a reference

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Abstract Field-based quantification of soil greenhouse gas emissions from the Philippines' agriculture sector is missing for vegetable production systems, despite its substantial contribution to agricultural production. We quantified soil N_2O emission, CH_4 uptake, and CO_2 efflux in vegetable farms and compared these to the secondary forest. Measurements were conducted for 13 months in 10 smallholder farms and nine forest plots on Andosol soil in Leyte, Philippines using static chambers. Soil N_2O and CO_2 emissions were higher, whereas CH_4 uptake was lower in the vegetable farms than in the forest. Vegetable farms had annual fluxes of 12.7 ± 2.6 kg N_2O-N ha^{-1} yr^{-1}, -1.1 ± 0.2 kg CH_4-C ha^{-1} yr^{-1}, and 11.7 ± 0.7 Mg CO_2-C ha^{-1} yr^{-1}, whereas the forest had 0.10 ± 0.02 kg N_2O-N ha ha^{-1} yr^{-1}, -2.0 ± 0.2 kg CH_4-C ha^{-1} yr^{-1}, and 8.2 ± 0.7 Mg CO_2-C ha^{-1} yr^{-1}. Long-term high N fertilization rates in vegetable farms resulted in large soil mineral N levels, dominated by NO_3^- in the topsoil and down to 1-m depth, leading to high soil N_2O emissions. Increased soil bulk density in the vegetable farms probably increased anaerobic microsites during the wet season and reduced CH_4 diffusion from the atmosphere into the soil, resulting in decreased soil CH_4 uptake. High soil CO_2 emissions from the vegetable farms suggested decomposition of labile organic matter, possibly facilitated by plowing and large N fertilization rates. The global warming potential of these vegetable farms was 31 ± 2.7 Mg CO_2-eq ha^{-1} yr^{-1} (100-year time frame).

Keywords Andosol soil · Greenhouse gas budget · Philippine agriculture · Soil N_2O · CH_4 · CO_2 fluxes · Soil nutrient stocks · Soil organic carbon

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Introduction

Globally, agriculture contributes 78% of anthropogenic nitrous oxide (N_2O) emissions, 39% of methane (CH_4) emissions (Jia et al. 2019), and land-use change accounts 14% of anthropogenic carbon dioxide (CO_2) emissions (Friedlingstein et al. 2019). Aside from being potent greenhouse gases (GHG), N_2O and CH_4 are also the most important ozone-depleting substances in the stratosphere (Ravishankara et al. 2009; Wuebbles and Hayhoe 2002). Although productive agricultural soils can be a sink of atmospheric CO_2, continued harvest export combined with reduced biomass input and decomposition of available organic matter often turn croplands into a net CO_2 source with low soil organic carbon (SOC) stocks (Sauerbeck 2001).

Since the 1950's, forest conversion has been substantial in the Philippines, driven by upland migration, agricultural expansion, and pressure from a growing population (Carandang et al. 2013; Kummer 1992). One of the economically important commodity crops of these converted lands is vegetable production, comprising 6% of the total agricultural area (supplied 6638 ± 85 Mg yr^{-1} vegetables in 2015–2019) and > 30% of the total agricultural production in the Philippines (Briones 2009; FAOSTAT 2021). Vegetable production is commonly associated with intensive management practices, including multiple cropping periods in a year, large mineral fertilizer inputs, pesticide applications, frequent cultivation and irrigation, and has been estimated to contribute 9% of the global N_2O emissions from nitrogen (N) fertilizers (Librero and Rola 2000; Rezaei Rashti et al. 2015). Long-term high N fertilization mainly drives large N_2O emissions from tropical agricultural systems, as N fertilization increases soil ammonium (NH_4^+) and nitrate (NO_3^-) which are substrates of nitrification and denitrification-the main microbial-mediated processes producing N_2O in the soil (Mosier et al. 2004; Veldkamp et al. 1998). As depicted by the hole-in-the-pipe (HIP) model, the two main soil regulating factors of soil N_2O emissions are N availability and water content (Davidson et al. 2000), which are both favorable in intensively managed vegetable production systems. Moreover, land-use conversion can affect soil CH_4 fluxes due to changes in soil bulk density, water content, and management practices including N fertilization and liming (Bodelier and Laanbroek 2004; Veldkamp et al. 2008, 2013). Losses of SOC from forest conversion to agriculture can reach as much as 58–66% (Beheshti et al. 2012; Powers et al. 2011; Veldkamp et al. 2020) and are attributed to direct C losses from land use conversion (biomass and organic matter burning; Achard et al. 2014), decrease in C input combined with harvest export (e.g., Meijide et al. 2020), and continued decomposition even decades after conversion due to regular cultivation, microclimate modification, and other changes in soil properties that influence SOC stabilization (e.g., van Straaten et al. 2015; Veldkamp et al. 2020).

The Philippines' agriculture sector is claimed to be a significant source of GHG, contributing ~ 30% of the country's total emissions (UNFCCC 1999, 2014). However, this GHG accounting was largely based on indirect data. Field-based quantification of soil GHG fluxes remains sparse at present and is particularly missing for vegetable production systems, despite its ~ 20% consumption of the country's mineral fertilizer use (Briones 2014, 2016). Thus, there is a knowledge gap in the Philippine GHG inventories with large uncertainties, attributed to the lack of reliable GHG estimates and deficient field-based measurements (Kawanishi et al. 2019, 2020; Ogle et al. 2013). Information on soil N_2O, CH_4, and CO_2 fluxes from agriculture, including the economically important vegetable production systems in the country, will provide a basis for mitigation measures and policy implementation in addressing climate change.

Our present study aimed to (1) assess the differences in soil GHG fluxes between secondary forest and vegetable farms on an Andosol soil in Leyte, Philippines, and (2) generate a GHG budget for a typical smallholder vegetable production system. Field-based measurements of soil N_2O, CH_4, and CO_2 fluxes were carried out for a year in these two adjacent land uses. We hypothesized that compared to the secondary forest, smallholder vegetable farms will have increased soil CO_2 efflux as a consequence of regular soil tillage under multiple cropping periods (Sauerbeck 2001), substantial N_2O emissions due to large N fertilizer inputs (Rezaei Rashti et al. 2015), and reduced CH_4 uptake in the soil as a result of soil compaction (Mosier et al. 2004). Our field-based quantification of soil GHG budget contributed in filling the knowledge gap for improving the GHG accounting of the Philippines' agriculture sector.

Materials and methods

Site description

This study was conducted in Cabintan, Ormoc, Leyte, Philippines (Electronic supplementary material Fig. 1). This area is characterized by a volcanic landscape (600–900 m above sea level and 11° 03–06' N, 124° 42–45' E) and comprises about 4100 ha, wherein 3600 ha are categorized as forest and 500 ha are utilized for agriculture and other uses (Department of Environment and Natural Resources 1981). The vegetable production in Cabintan had started in the early 2000s and is the most recent agricultural activity which was preceded by cycles of shifting cultivation (i.e., between periods of cultivation, the area is fallowed allowing re-growing of secondary forest). The general land-use history of Cabintan for the last 60 years is given in the Electronic supplementary material Table 1. The mild upland climate and favorable physical properties of the volcanic soil, coupled with the use of fertilizers and pesticides, permit continuous vegetable production in this area possible.

The area had a mean annual precipitation of 3000 ± 390 mm (± SE of the mean) and temperature of 28 ± 0.2 °C (2014–2017 data from Tacloban City weather station of the Department of Science and Technology Philippine Atmospheric, Geophysical and Astronomical Services Administration (DOST PAG-ASA), Leyte, Philippines). During our study year (2018–2019), annual rainfall was 2730 mm with monthly rainfall ranging from 360 to 485 mm from December to February, between 110 and 130 mm during March to May, and an average of 180 mm in the remaining months, which allowed vegetable cropping two to three times a year. Average monthly air temperature ranged between 23 and 32 °C (2018–2019 data from weather stations at the City Agriculture Office, Ormoc City and DOST PAG-ASA, Tacloban City).

Soils in the study area have developed from intermediate volcanoclastics (55% SiO_2) of trachytic basalt-andesite rocks from late quaternary period (Asio et al. 1998; Jahn and Asio 1998). The soils are characterized by inherently low soil bulk density, low pH, high P adsorption capacity, and dominance of amorphous allophane and imogolite minerals (Asio et al. 1998; Jahn and Asio 1998; Navarrete et al. 2008), and classified as Typic Hapludand (USDA soil classification) or Umbric Andosol (FAO soil classification).

Experimental design and farm management practices

We selected nine plots in the secondary forest, representing the original land use before conversion, and 10 farms presently engaged in vegetable production each owned by a smallholder farmer, all located on the same soil type and climate. Forest plots were at least 600 m away from the vegetable farms. These plots were 64 m^2 each, with a distance between plots of at least 25 m, and were spatially independent in terms of soil GHG flux measurements, based on statistical test using the von Neumann's ratio test (Bartels 1982). The forest plots, therefore, can be considered as replicates in the succeeding statistical analysis of the repeatedly measured variables during the study period. Within each forest plot, we counted the number of individuals of all tree species present and reported the top eight species with the highest number of individuals in each plot (Table 1). These tree species are endemic in the Philippines and classified under the families of an old dipterocarp forest in Leyte Island (Margraf and Milan 1996; Schneider et al. 2014). There were no slash and burn activities near our forest plots during our study. The forest is described as secondary growth since it recovered from serious disturbance caused by the super typhoon Haiyan in 2013 (wind speed exceeding 220 km per hour; Cinco et al. 2016), which was considered the strongest cyclone in the Western North Pacific (Mori et al. 2014; Takagi et al. 2017).

We randomly selected 10 smallholder vegetable farms based on field survey (e.g., distance to the forest site, comparability of land-use independent soil property (see below)) and farmers' interviews. Farm-scale information included farm size, vegetables grown, yield, fertilization rates and sources, crop protection practices, and previous land use. Our studied farms had an area allocated for vegetables of ≤ 0.30 ha with 1.5–2 km distance between farms. Farmers practiced two to three cropping periods in a year with typically varied fertilization rates (Table 1), depending on the vegetables grown, and applied in combinations of synthetic fertilizers and chicken manure (containing 288 ± 10 g C kg^{-1}, 51 ± 3 g N kg^{-1}, 30 ± 2 g P kg^{-1}, 15 ± 1 g Mg kg^{-1}, 54 ± 5 g Ca kg^{-1}, 56 ± 2 g K kg^{-1}). Occasionally, some farmers also applied lime

Table 1 Site characteristics of the secondary forest (reference, n = 9 plots) and small-scale vegetable farms (n = 10 farms), all located on an Andosol soil in Leyte, Philippines

Land use	Tree families present	Dominant tree species (local names and in parentheses are scientific names) [a]
Secondary forest	Myristicaceae, Apocynaceae	Duguan (*M. philippensis* Lamk.), Lanete (*W. pubescens* R. Br.),
	Fagaceae, Myrtaceae	Ulayan (*L. ovalis* (Blanco) Rehd.), Maka-asim (*S. nitidum* Benth.),
	Euphorbiaceae, Dilleniaceae	Balante (*M. caudatifolia* Elm.), Katmon (*D. philippinensis* Rolfe),
	Guttiferae, Melastomaceae	Bitanghol (*C. blancoi* Pl. and Tr.), *Melastoma* sp.

Small-scale vegetable farm [b]	Crops (common names and in parentheses are scientific names)	Cropping period	Harvested yield (kg dry biomass ha^{-1})	N fertilization rate (kg N ha^{-1} cropping $period^{-1}$)	
				Synthetic fertilizer	Chicken manure
Farm 1	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	First	1010	210	125
	Cabbage (*B. oleracea* var. *capitata*)	Second	544	475	125
	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	Third	40	230	125
Farm 2	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	First	1078	155	175
	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	Second	414	50	170
	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	Third	1311	110	150
Farm 3	Tomato (*S. lycopersicum* L.)	First	479	370	200
	Cabbage (*B. oleracea* var. *capitata*)	Second	857	100	300
Farm 4	Tomato (*S. lycopersicum* L.)	first	82	140	140
	Cabbage (*B. oleracea* var. *capitata*)	Second	44	200	155
	Cabbage (*B. oleracea* var. *capitata*)	Third	452	85	130
Farm 5	Tomato (*S. lycopersicum* L.)	First	420	85	240
	Squash (*Cucurbita* sp.)	Second	312	125	110
	Sweet pepper (*C. annuum* L.)	Third	792	150	180
Farm 6	Sweet pepper (*C. annuum* L.)	First	486	125	85
	Tomato (*S. lycopersicum* L.)	Second	153	170	140
Farm 7	Spring onion (*A. fistulosum* L.)	First	593	280	0
	Spring onion (*A. fistulosum* L.)	Second	382	360	245
	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	Third	816	170	350
Farm 8	Tomato (*S. lycopersicum* L.)	First	138	150	245
	Eggplant (*S. melongena* L.)	Second	846	285	540
Farm 9	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	First	168	75	70
	Eggplant (*S. melongena* L.)	Second	1254	145	115

Table 1 (continued)

Small-scale vegetable farm [b]	Crops (common names and in parentheses are scientific names)	Cropping period	Harvested yield (kg dry biomass ha^{-1})	N fertilization rate (kg N ha^{-1} cropping $period^{-1}$)	
				Synthetic fertilizer	Chicken manure
Farm 10	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	First	786	140	225
	Chinese cabbage (*B. rapa* subsp. *pekinensis*)	Second	45	260	170
	Chili pepper (*Capsicum* sp.)	Third	1292	70	80

[a] These tree species are the most dominant in terms of the number of individuals present in the forest plots

[b] These farms have commonly varied fertilization regimes in accordance to the cultivated vegetables and reflected the common management practices of farmers in the study area

($CaCO_3$) but they could not give the exact rate. The synthetic fertilizers commonly utilized were urea (46% N), muriate of potash (50% K), ammonium phosphate (16% N–9% P), ammonium sulfate (21% N), diammonium phosphate (18% N–20% P), and complete formula (14% N–6% P–12% K).

Cultivation practices employed in all farms during the cropping periods depict distinct zones, namely plant rows (~70% areal coverage), where crops were planted and received fertilizers, and inter-rows (~30% areal coverage), where access for rendering manual management activities occurred. These cultivation zones were established using an animal-drawn (*Bubalus bubalis carabenesis*) plow to a depth of ~0.6 m. Vegetables can vary from leafy, mainly of the Brassicaceae family, to fruit-bearing types, which included tomato, eggplant, pepper, and squash (Table 1). Fertilization frequency of synthetic fertilizers ranged from weekly to biweekly, depending on the grown vegetable, whereas air-dried chicken manure was applied in once to twice instances during the early stage of the cropping period. Fertilizers were applied directly to the soil close to the crop base (~0.1 m away) during the wet season or first dissolved in water for application during low rainfall months. Crop protection practices included insecticide, molluscicide, and fungicide that range from highly hazardous (Methomyl in Scorpio 40 SP™, Lannate 40 SP™, Check Mate 40 SP®), moderately hazardous (Cypermethrin in Magnum®, Metaldehyde in Stop®, Cartap hydrochloride in Contract 50 SP®, Difenoconazole in Montana®) to least hazardous (Chlorothalonil in Rover®) (based on LD_{50} test; World Health Organization 2010). A mixture of at least two to three of these pesticides was frequently practiced, and their use was largely based on the extent of pest incidence during cropping and a day before each harvest. As to weed control, the farmers applied broad-spectrum herbicides (Glyphosate IPA in Tekweed 480 SL®, Sharp Shooter 480 SL®, Mower 48 SL®) as a cheaper alternative to manual weeding. The vegetable farms were intensively managed (Table 1), and the produced vegetables were marketed not only in Leyte Island but also to adjacent islands, including the country's capital in the north and cities in the southern provinces.

Soil physical and biochemical properties

In an area adjacent to each farm or forest plot, we dug a pit (1 × 1 × 1 m) to measure soil bulk density (core method; Blake and Hartge 1986) at depth intervals of 0–0.1, 0.1–0.3, 0.3–0.5, and 0.5–1.0 m. Within each forest plot or farm, we collected soil samples once in 2018 at the same depth intervals. The collection was done on five sampling points within each plot for each depth using an auger; the five soil samples were composited, totaling to 76 samples (4 depths × (9 forest plots + 10 farms)). Soils were air-dried and sieved using 2 mm mesh. These soil samples were determined for texture at the Visayas State University, Leyte, Philippines, using hydrometer method (ISRIC 2002) after pre-treatments for removal of organic matter and with 1 mol L^{-1} NaOH to increase particle dispersion. Soil samples were transported by air to the laboratory of Soil Science of Tropical and Subtropical Ecosystems (SSTSE), University of Goettingen, Germany for biochemical analysis. Soil pH (1:2.5 ratio of soil to distilled H_2O) was measured potentiometrically. Effective cation exchange capacity

(ECEC) was determined by percolating with unbuffered 1 mol L^{-1} NH_4Cl solution and the percolates were analyzed for exchangeable cations (Mg, Ca, K, Na, Al, Fe, Mn) using an inductively coupled plasma-atomic emission spectrometer (ICP-AES; iCAP 6300 Duo VIEW ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). Base and Al saturations were calculated as the mole charge ratio of exchangeable bases (Mg, Ca, K, Na) and Al on ECEC, respectively. SOC and total N were analyzed from finely ground soil samples using a CN analyzer (Elementar Vario EL; Elementar Analysis Systems GmbH, Hanau, Germany).

To check for comparability of the initial soil conditions between the reference forest and the vegetable farms, we used a land-use independent soil variable - the soil texture in the 0.5–1.0 m depth - as used by other studies (e.g., Allen et al. 2016; de Blécourt et al. 2013; Tchiofo Lontsi et al. 2019; van Straaten et al. 2015; Veldkamp 1994). The forest plots and vegetable farms had similar ($P = 0.76$) soil texture (forest, mean ± SE: 10 ± 1% clay, 72 ± 3% sand, 18 ± 2% silt; farms: 10 ± 3% clay, 64 ± 3% sand, 26 ± 3% silt). To evaluate the changes in element stocks attributable to land-use change, we used equal soil mass between the land uses by taking the soil bulk density of the secondary forest (as the original land use) and used this in the conversion of element concentrations from soil mass basis to area basis for each depth interval down to one meter (de Blécourt et al. 2013; van Straaten et al. 2015; Veldkamp 1994). We present the element stocks in the top 0.5 m as the sum of the three depth intervals (0–0.1, 0.1–0.3, and 0.3–0.5 m) and for the other soil properties as depth-weighted averages.

Soil greenhouse gas flux measurement and soil controlling variables

Soil N_2O, CH_4, and CO_2 fluxes were measured from May 2018 to May 2019 at monthly interval (13 measurement periods) on each replicate plot, employing the vented static chamber method (e.g., Tchiofo Lontsi et al. 2020). A chamber base (made of a polyvinyl chloride pipe with 0.04 m^2 area, 0.05 m height, and inserted into the soil to ~ 0.02 m) was installed permanently in each of the nine forest plots. For the 10 vegetable farms, each had a total of eight chamber bases with four chambers installed on the plant rows and another four on the inter-rows. On each measurement day, a polyethylene chamber hood (equipped with a Luer lock sampling port) was placed on the chamber base, totaling the chamber head space volume to 8.4 L. Gas samples were taken using a syringe at 2, 12, 22, and 32 min following chamber closure. A 22 mL gas sample was collected at each time interval and stored in pre-evacuated 12 mL glass vials (Exetainer; Labco Limited, Lampeter, United Kingdom); these exetainers have been tested to be leak-proof, as shown by their maintained overpressure and no change in concentrations of stored standard gases (e.g., Hassler et al. 2015; Iddris et al. 2020). The gas samples were transported by air to the laboratory of SSTSE, Goettingen University, Germany and were analyzed for N_2O, CH_4, and CO_2 concentrations using a gas chromatograph (GC; SRI 8610C, SRI Instruments Europe GmbH, Bad Honnef, Germany) equipped with an electron capture detector (^{63}Ni, ECD; for N_2O with a make-up gas of 5% CO_2–95% N_2), a flame ionization detector (for CH_4 and CO_2 with a methanizer), and an autosampler. These three gases were analyzed from the same gas sample. We regarded the linear increase of CO_2 concentration with chamber closure time as our reference for quality check of the other GHG concentrations. All chamber measurements showed significant linear increases in CO_2 concentrations during the 32 min chamber closure ($R^2 = 0.99$), justifying that all N_2O and CH_4 fluxes must all be included in our data analysis. Soil gas fluxes from each chamber were calculated based on the linear increase (CO_2, N_2O) or decrease (CH_4) with chamber closure, corrected with the in-situ measured air pressure and temperature (as elaborated in our earlier studies; e.g., Hassler et al. 2015; Iddris et al. 2020; Koehler et al. 2009a, 2009b; Tchiofo Lontsi et al. 2020). Annual N_2O, CH_4, and CO_2 fluxes from each chamber were calculated using the trapezoidal rule by interpolating the measured fluxes between monthly intervals during the study period from May 2018 to May 2019. The annual soil GHG fluxes from the vegetable farms were further weighted by the areal coverages of the two distinctive cultivation zones (70% for plant rows and 30% for inter-rows).

To assess if we missed possible high N_2O fluxes immediately after fertilization, we established a supplementary study covering two seasons whereby we conducted intensive measurements following N fertilizer applications. For this follow-on investigation, we selected four from the 10 farms, and in each farm we delineated 20 m^2 plot; each plot received a specific N fertilization rate (equivalent to 75, 180, 230, and 480 kg N ha^{-1} yr^{-1}, based on our farmers' practices; Table 1), which was split into eight weekly applications, similar to farmers' practice. In each plot, we installed three chambers on the plant rows and three chambers on the inter-rows. We measured soil GHG fluxes a day after each weekly fertilizer application for two months. In summary, we found that the soil GHG fluxes from weekly measurements (data not reported) did not differ from the monthly measurements ($P = 0.15$; based on statistical analysis described below) indicating that we have captured the pulse of N_2O fluxes after fertilization.

Concurrent with each measurement of soil GHG fluxes, we also determined soil variables (i.e., mineral N, water-filled pore space (WFPS), and soil temperature) in the top 0.05 m. In each forest plot on each measurement day, we took five soil samples at ~ 1 m away from the chamber and mixed into one composite sample. In each vegetable farm on each measurement day, we collected four soil samples at ~ 0.5 m from each of the four chambers deployed on the plant rows or inter-rows, and subsequently mixed into one composite sample for each cultivation zone. A portion of the composite soil was oven-dried (105 °C) for one day for gravimetric moisture determination; this was expressed as WFPS using the measured soil bulk density and a particle density of 2.65 g cm^{-3}. In-situ extraction of soil NH_4^+ and NO_3^- was conducted by bringing to the field prepared polyethylene bottles filled with 150 mL and 0.5 mol L^{-1} K_2SO_4 into which subsample of the composite soil from each forest plot or cultivation zone of each vegetable farm was added. Extraction continued upon arrival at the field station by shaking the extraction bottles for an hour and filtering the extracts, which were then immediately frozen. Soil extracts were kept frozen and transported by air to the laboratory of SSTE, Goettingen University, Germany, where analysis was conducted. Mineral N concentrations were determined using continuous flow injection colorimetry (SEAL Analytical AA3, SEAL Analytical GmbH, Norderstedt, Germany).

We also quantified the soil extractable NH_4^+, NO_3^-, and organic N within 1-m depth at four depth intervals (0–0.1, 0.1–0.3, 0.3–0.5, and 0.5–1.0 m) once in the dry season and another in the wet season of 2019 for additional evidence to explain the large N_2O emissions from these farms. In each vegetable farm, we took five soil samples for each depth interval and composited for each depth. These soil samples underwent the same in-situ extraction procedure as described above. We report the stocks of extractable N separately for the top 0.5 and 0.5–1.0 m soil depths.

We calculated the overall soil GHG footprint of smallholder vegetable production system at our study area in the following:

$$\mathrm{GWP} = (\mathrm{NEP}) + \left(\mathrm{soil\,N_2O} \times 298\right) + \left(\mathrm{soil\,CH_4} \times 25\right) \tag{1}$$

For each replicate farm, GWP is the global warming potential (Mg CO_2-eq ha^{-1} yr^{-1}). NEP is net ecosystem productivity (Mg C ha^{-1} yr^{-1}) = net ecosystem C exchange (NEE) + C exported by harvested yield. NEE (Mg C ha^{-1} yr^{-1}) = heterotrophic respiration-[net primary production (NPP; above- and belowground NPP) + chicken manure] (Malhi et al. 1999). Aboveground NPP was derived from our actual measurements of harvested yields (Table 1) and the vegetables' harvest indices (HI = harvestable yield ÷ (herbage + harvestable yield)); Electronic supplementary material Table 2). We expressed the yields in dry mass using the moisture contents of the harvested vegetables (Table 1), determined by drying samples at 70 °C for at least three days or until stable weights were attained. Belowground NPP was calculated based on the root:shoot ratios of these vegetables (Electronic supplementary material Table 2). The NPP and the C exported by the harvested yield were converted to biomass-C based on C content analysis of each vegetable (measured as described above). Heterotrophic respiration was represented by the inter-rows as this cultivation zone did not have plant roots. For the final GWP calculation, NEP was multiplied by 3.67 to convert C to CO_2. Soil N_2O (Mg N_2O ha^{-1} yr^{-1}) and CH_4 fluxes (Mg CH_4 ha^{-1} yr^{-1}) were converted to 100 yr CO_2 equivalent by multiplying with 298 for N_2O and 25 for CH_4 (IPCC Guidelines for National Greenhouse Gas Inventories 2006). Our GWP calculation was limited only during the crop production of our study period and to the farm level, while excluding other GHG sources (e.g., fertilizer manufacturing and transport to the field).

Statistical analysis

We first tested the soil GHG fluxes and soil variables measured in the top 0.05 m for normality of distribution (Shapiro Wilk's test) and equality of variance (Levene's test); variables with non-normal distributions were log-transformed (i.e., soil N_2O, CH_4, CO_2 fluxes, mineral NH_4^+ and NO_3^-, WFPS, soil temperature). To answer our main objective (i.e., quantify the changes in soil GHG fluxes between forest and smallholder vegetable farms), the nine forest plots were considered replicates of the reference land use whereas the 10 vegetable farms were regarded as replicate plots to represent the inherent varied management practices (Table1) common to smallholders at our study area. Thus, the eight chambers (four on plant rows and four on inter-rows) at each farm on each measurement day were taken as subsamples and were nested within each farm (replicate) in the succeeding statistical analysis. We used the linear mixed-effects (LME) models (Crawley 2013) for repeatedly measured parameters (i.e., soil GHG fluxes and soil variables) in assessing differences between forest and vegetable production, or between plant rows and inter-rows if the purpose was to test between these cultivation zones within farms. In LME, land use or cultivation zone (for vegetable production only) was considered as fixed effect whereas measurement day and replicate plot were random effects. In the LME models, we included (1) a first-order temporal autoregressive process that assumes a decreasing correlation between measurements with increasing time distance (Zuur et al. 2009), and (2) a variance function that allows different variances of the fixed effect (Crawley 2013). The best LME model was chosen based on the Akaike information criterion.

We also determined the temporal correlations of soil GHG fluxes with the soil variables. We used the average of the nine forest plots or 10 vegetable farms on each measurement period and conducted Pearson's correlation test over the 13 monthly measurements (n = 13 for the forest and n = 26 for the vegetable farms, with 2 cultivation zones × 13), as the correlation patterns remained statistically the same when using the individual replicate plots of each land use.

For the additional measurements of extractable mineral N down to 1-m depth, the difference between seasons was assessed using either paired *T* test (normal distribution) or the non-parametric Wilcoxon signed-rank test. Similarly, for soil physical and biochemical properties measured once, comparison between the two land uses for each depth was conducted using either independent *T* test or non-parametric Wilcoxon rank-sum test. We considered the significant level at $P \leq 0.05$ for all the tests. Statistical analyses were carried out using R version 4.0.2 (R Core Team 2020). R packages central to our analyses involved car (Fox and Weisberg 2019), lme4 (Bates et al. 2015), MASS (Venables and Ripley 2002), multcomp (Hothorn et al. 2008), pgirmess (Giraudoux 2022), and ggplot2 (Wickham 2016) packages.

Results

Soil properties and nutrient stocks

In the top 0.5 m, soil bulk density was larger in the vegetable farms than the forest ($P < 0.01$; Table 2). Soil pH and stocks of Ca, K, and Mn did not differ between land uses ($P = 0.10$–0.65). Al saturation was larger in the forest than the vegetable farms ($P < 0.01$; Table 2). In contrast, base saturation was ~ 50% higher in the farms than the forest ($P < 0.01$). ECEC, SOC, and total N stocks decreased by 36, 61, and 40%, respectively, in the vegetable farms compared to the forest (all $P \leq 0.01$; Table 2).

In the 0.5–1.0 m, soil bulk density in the vegetable farms remained higher than in the forest ($P < 0.05$; Table 2). Soil pH, ECEC, and stocks of Al and Fe were similar between land uses ($P = 0.14$–0.80). Vegetable farms showed larger Mg, Ca and K stocks, and base saturation than the forest ($P \leq 0.01$–0.03; Table 2). Conversely, Al saturation, total N, and SOC stocks were lower in the vegetable farms compared to the forest (all $P < 0.01$; Table 2).

Differences in soil greenhouse gas fluxes between land uses

The forest soil consistently emitted N_2O, consumed CH_4, and emitted CO_2 over the entire 13 month measurements (Table 3; Fig. 1a–c). Of the soil variables quantified during the measurement period, only NH_4^+ and WFPS were larger in the wet than in the dry season ($P \leq 0.01$–0.02); soil temperature did not show seasonal variation (Table 4). Extractable mineral N

Table 2 Soil physical and biochemical properties in the top 0.5 m and 0.5–1.0 m of the secondary forest (mean ± SE, n = 9 plots) and small-scale vegetable farms (mean ± SE, n = 10 farms) on an Andosol soil in Leyte, Philippines

Soil depth and characteristic	Secondary Forest	Small-scale vegetable farm
0–0.5 m		
Bulk density (g cm^{-3})	0.3 ± 0.0b	0.5 ± 0.0a
pH (1:2.5 soil-H_2O ratio)	5.1 ± 0.0a	5.1 ± 0.1a
ECEC ($cmol_c$ kg^{-1})	4.1 ± 0.3a	2.6 ± 0.3b
Base saturation (%)	31.1 ± 3.1b	64.7 ± 6.6a
Al saturation (%)	67.0 ± 3.1a	32.4 ± 6.3b
SOC concentration (g C kg^{-1})	93.9 ± 3.3a	36.7 ± 3.6b
Total soil N concentration (g N kg^{-1})	6.2 ± 0.2a	3.4 ± 0.3b
C:N Ratio	15.0 ± 0.4a	10.8 ± 0.2b
SOC stock (kg C m^{-2})	15.3 ± 0.5a	6.0 ± 0.6b
Total soil N stock (kg N m^{-2})	1.0 ± 0.0a	0.6 ± 0.1b
Exch. Mg (g Mg m^{-2})	6.1 ± 0.3a	4.2 ± 0.3b
Exch. Ca (g Ca m^{-2})	22.2 ± 2.2a	31.7 ± 3.8a
Exch. K (g K m^{-2})	10.5 ± 0.6a	15.5 ± 3.1a
Exch. Na (g Na m^{-2})	2.7 ± 0.4a	1.0 ± 0.2b
Exch. Al (g Al m^{-2})	40.1 ± 4.0a	15.8 ± 4.5b
Exch. Fe (g Fe m^{-2})	0.9 ± 0.3a	0.1 ± 0.0b
Exch. Mn (g Mn m^{-2})	1.7 ± 0.2a	3.1 ± 0.7a
0.5–1.0 m		
Bulk density (g cm^{-3})	0.4 ± 0.0b	0.5 ± 0.0a
pH (1:2.5 soil-H_2O ratio)	5.1 ± 0.0a	5.0 ± 0.1a
ECEC ($cmol_c$ kg^{-1})	0.8 ± 0.1a	1.2 ± 0.2a
Base saturation (%)	40.6 ± 3.2b	79.8 ± 6.5a
Al saturation (%)	57.3 ± 3.4a	17.3 ± 6.3b
SOC concentration (g C kg^{-1})	37.1 ± 2.4a	17.0 ± 2.2b
Total soil N concentration (g N kg^{-1})	2.7 ± 0.2a	1.6 ± 0.2b
C:N Ratio	14.0 ± 0.3a	11.0 ± 0.2b
SOC stock (kg C m^{-2})	7.8 ± 0.5a	3.6 ± 0.5b
Total soil N stock (kg N m^{-2})	0.6 ± 0.0a	0.3 ± 0.0b
Exch. Mg (g Mg m^{-2})	1.7 ± 0.2b	3.3 ± 0.6a
Exch. Ca (g Ca m^{-2})	6.6 ± 0.4b	26.4 ± 4.1a
Exch. K (g K m^{-2})	5.2 ± 0.4b	10.8 ± 2.2a
Exch. Na (g Na m^{-2})	1.7 ± 0.2a	1.2 ± 0.3b
Exch. Al (g Al m^{-2})	9.3 ± 1.3a	5.5 ± 2.7a
Exch. Fe (g Fe m^{-2})	0.0 ± 0.0a	0.0 ± 0.0a
Exch. Mn (g Mn m^{-2})	0.9 ± 0.1b	1.7 ± 0.3a

Means followed by different lowercase letters within a row indicate significant difference between land uses (Independent *T* test or Wilcoxon rank-sum test at $P \leq 0.05$). For the top 0.5 m, values for each replicate plot are weighted average of the sampled depth intervals at 0–0.1, 0.1–0.3, and 0.3–0.5 m, except for element stocks which are the sum of the entire depth

showed NH_4^+ dominance over NO_3^- throughout the measurement period (Table 4).

In the vegetable farms, soil N_2O and CO_2 emissions were consistently higher whereas soil CH_4 uptake was lower compared to the forest (all $P < 0.01$; Table 3; Fig. 1d–f). Between cultivation zones of the vegetable farms, soil N_2O emissions from the plant row were higher than from the inter-row ($P < 0.01$; Table 3; Fig. 1d) whereas soil CH_4 and CO_2 fluxes did not differ ($P = 0.06–0.17$; Table 3; Fig. 1e, f). Moreover, only the soil N_2O emissions from the plant row exhibited seasonal changes with 44% higher emissions during the dry than the wet season ($P < 0.01$; Fig. 1d). In the inter-row, only soil CH_4 uptake showed seasonal pattern, which was lower in the wet than the dry season ($P = 0.02$; Fig. 1e).

Table 3 Soil trace gas fluxes and annual fluxes (top 0.05 m) in the secondary forest (mean ± SE, n = 9 plots) and small-scale vegetable farms (mean ± SE, n = 10 farms), with monthly measurements from May 2018 to May 2019, on an Andosol soil in Leyte, Philippines

Land use	Cultivation zone	N_2O flux (μg N m^{-2} h^{-1})	CH_4 flux (μg C m^{-2} h^{-1})	CO_2 flux (mg C m^{-2} h^{-1})	N_2O flux (kg N ha^{-1} yr^{-1})	CH_4 flux (kg C ha^{-1} yr^{-1})	CO_2 flux (Mg C ha^{-1} yr^{-1})
Secondary forest		1.2 ± 0.2B	−24.1 ± 1.5B	92.1 ± 3.1B	0.10 ± 0.02	−2.0 ± 0.2	8.2 ± 0.7
Small-scale vegetable farm		133.4 ± 19.7A	−12.9 ± 0.9A	130.3 ± 5.0A	12.7 ± 2.6	−1.1 ± 0.2	11.7 ± 0.7
	Plant row	187.1 ± 30.3a	−13.8 ± 0.9a	136.1 ± 5.2a			
	Inter-row	54.3 ± 7.1b	−9.6 ± 1.3a	109.2 ± 4.0a			

Means followed by different capital letters indicate significant difference between land uses, and different lowercase letters denote significant difference between cultivation zones of vegetable farms (linear mixed-effects models with Fisher's LSD test at $P \leq 0.05$). Annual soil trace gas fluxes are calculated using the trapezoidal rule between monthly measurement intervals; for vegetable farms, annual emissions are area-weighted with 70% plant rows and 30% inter-rows

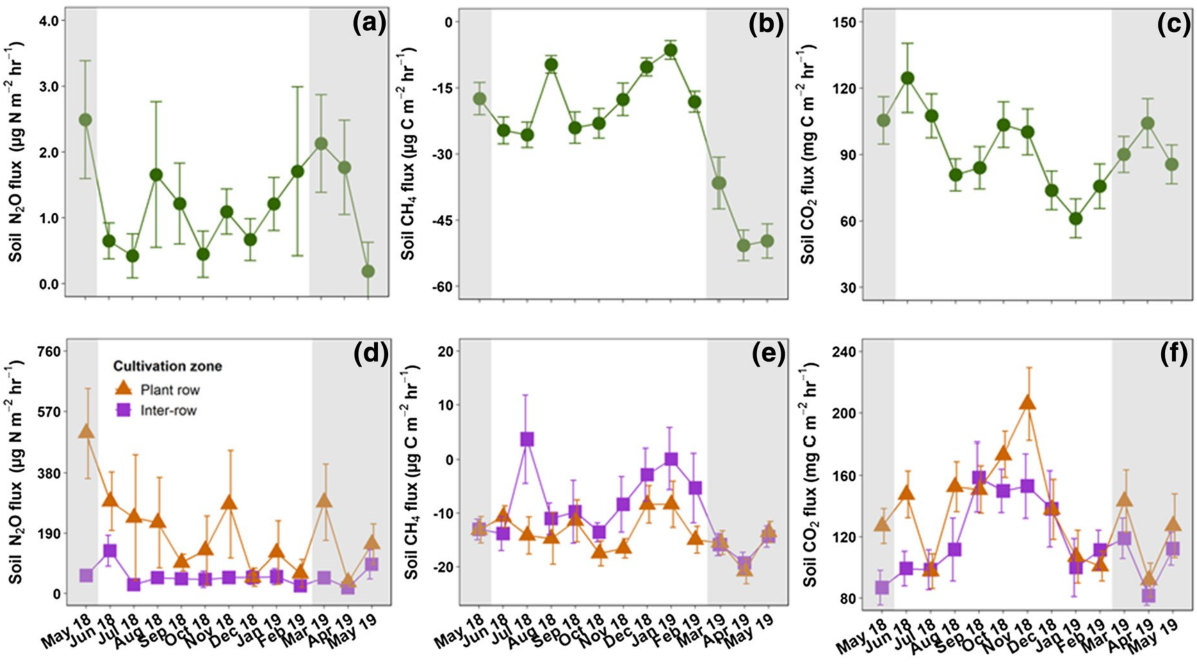

Fig. 1 Mean (± SE) soil N_2O, CH_4 and CO_2 fluxes from the secondary forest (n = 9 plots; a–c) and cultivation zones of small-scale vegetable farms (n = 10 farms; d–f), with commonly varied fertilization regimes (Table 1), on an Andosol soil in Leyte, Philippines. Gray shadings mark the dry season (< 150 mm rainfall $month^{-1}$.)

The annual soil GHG fluxes were remarkably larger than the forest (Table 3), ranging from 7.0 to 34 kg N_2O-N ha^{-1} yr^{-1}, −2.0 to −0.2 kg CH_4-C ha^{-1} yr^{-1}, and 9.0 to 15 Mg CO_2-C ha^{-1} yr^{-1}.

The GWP in the smallholder farms ranged from 18 to 42 Mg CO_2-eq ha^{-1} yr^{-1} across the studied 10 farms (Fig. 4; Electronic supplementary material Table 2). The heterotrophic respiration (i.e., 10 ± 0.6 Mg C ha^{-1} yr^{-1}; represented by the CO_2 flux from the inter-row as crop roots were absent from this zone) was larger than the combined C inputs from NPP (1.2 ± 0.2 Mg C ha^{-1} yr^{-1}) and chicken manure

Table 4 Soil variables (top 0.05 m) in the secondary forest (mean ± SE, n = 9 plots) and small-scale vegetable farms (mean ± SE, n = 10 farms), with monthly measurements from May 2018 to May 2019, on an Andosol soil in Leyte, Philippines. For mineral N, values in parenthesis are expressed in mg N m^{-2} (top 0.05 m), calculated using the soil bulk density of the reference land use

Land use	Cultivation zone	NH_4^+ (mg N kg^{-1})	NO_3^- (mg N kg^{-1})	Water–filled pore space (%)	Soil temp.(°C)
Secondary forest		7.4 ± 0.3A	0.2 ± 0.1B	60.0 ± 1.6A	21.8 ± 0.1B
		(121.3 ± 5.4)	(3.4 ± 1.0)		
Small-scale vegetable farm		11.4 ± 1.6A	17.3 ± 1.9A	55.2 ± 1.3A	24.4 ± 0.2A
		(187.4 ± 25.7)	(285.0 ± 32.0)		
	Plant row	13.8 ± 2.1a	20.4 ± 2.4a	55.8 ± 1.3a	24.4 ± 0.2a
	Inter-row	5.3 ± 0.3b	10.1 ± 1.2a	54.3 ± 1.4a	24.5 ± 0.2a

Means followed by different capital letters indicate significant difference between land uses, and different lowercase letters display significant difference between cultivation zones of vegetable farms (linear mixed-effects models with Fisher's LSD test at $P \leq 0.05$)

(2.6 ± 0.3 Mg C ha^{-1} yr^{-1}), signifying the farms as a net source of CO_2 (Fig. 4; Electronic supplementary material Table 2). On average, NEP comprised 81% of the GWP, soil N_2O emissions accounted 19%, whereas soil CH_4 uptake counteracted only ~ 0.1%.

Differences in soil variables between land uses and their relationships with greenhouse gas fluxes

Soil NO_3^- and temperature were higher in the vegetable farms than in the forest ($P < 0.01$; Table 4) whereas soil NH_4^+ and WFPS did not differ between land uses ($P \geq 0.62$; Table 4). In the vegetable farms, NO_3^- was the dominant form of extractable N (Tables 4, 5). Soil NO_3^- and WFPS did not differ between cultivation zones ($P \geq 0.09$; Table 4) but differed between seasons: NO_3^- was higher while WFPS was lower ($P \leq 0.01$) in the dry than the wet season. Soil NH_4^+ in the vegetable farms was higher in the plant row ($P < 0.01$; Table 4) than the inter-row but did not vary between seasons ($P = 0.41$). The vegetables farms also showed large stocks of NO_3^- in the entire 1 m depth followed by extractable organic N and NH_4^+, and these were larger in the wet than in the dry season ($P < 0.01$; Table 5). Soil temperature in the vegetable farms neither differed between cultivation zones nor between seasons (Table 4).

Soil CH_4 fluxes from the forest showed a positive relationship with WFPS, illustrating a decrease in CH_4 uptake with an increase in soil moisture (Fig. 2a). Soil CO_2 emissions from the forest displayed positive correlation with soil temperature (Fig. 2b). In the vegetable farms, soil N_2O fluxes were positively correlated with total mineral N (NH_4^++NO_3^-) (Fig. 2c). Conversely, soil CH_4 fluxes from the vegetable farms exhibited negative relationship with NO_3^- (Fig. 2d).

Table 5 Mean (± SE) stocks of soil extractable N within 1-m depth in small-scale vegetable farms (n = 10 farms) during the dry and wet seasons of 2019, on an Andosol soil in Leyte, Philippines

Extractable N	Small-scale vegetable farms			
	Top 0–0.5 m soil depth		0.5–1.0 m soil depth	
	Dry	Wet	Dry	Wet
NH_4^+ (mg N m^{-2})	644 ± 62a	872 ± 115a	252 ± 27b	590 ± 99a
NO_3^- (mg N m^{-2})	6520 ± 2584b	12,844 ± 3354a	7423 ± 3915b	15,088 ± 8084a
Organic N (mg N m^{-2})	8301 ± 814b	9431 ± 654a	3506 ± 421b	4608 ± 479a

Means followed by different lowercase letters within a row and soil depth indicate significant difference between cropping seasons (Paired *T* test or Wilcoxon signed-rank test at $P \leq 0.05$). Values of extractable N for each replicate plot in the top 50 cm are cumulative N stocks from the sampled soil depth intervals 0–0.1, 0.1–0.3, and 0.3–0.5 m

Considering both land uses, soil CO_2 emissions showed a parabolic relationship with WFPS, which was not revealed when only either forest or vegetable farm was considered; soil CO_2 emissions increased within 30–45% WFPS, were optimum at approximately > 55–65% WFPS, and decreased at > 70% WFPS (Fig. 3). There were no other significant relationships observed between soil GHG fluxes and measured soil controlling variables.

Discussion

Changes in soil properties between forest and vegetable farms

The SOC stock in our forest soil (Table 2) was comparable to other forests on Andosol soils (e.g., 12–14 kg C m^{-2} in the top 0.5 m, Veldkamp 1994; de Koning et al. 2003; 17 kg C m^{-2} in the top 1.0 m, Anda and Dahlgren 2020). Changes in soil properties after forest conversion to croplands can be substantial, persistent (e.g., detectable even after > 25 years),

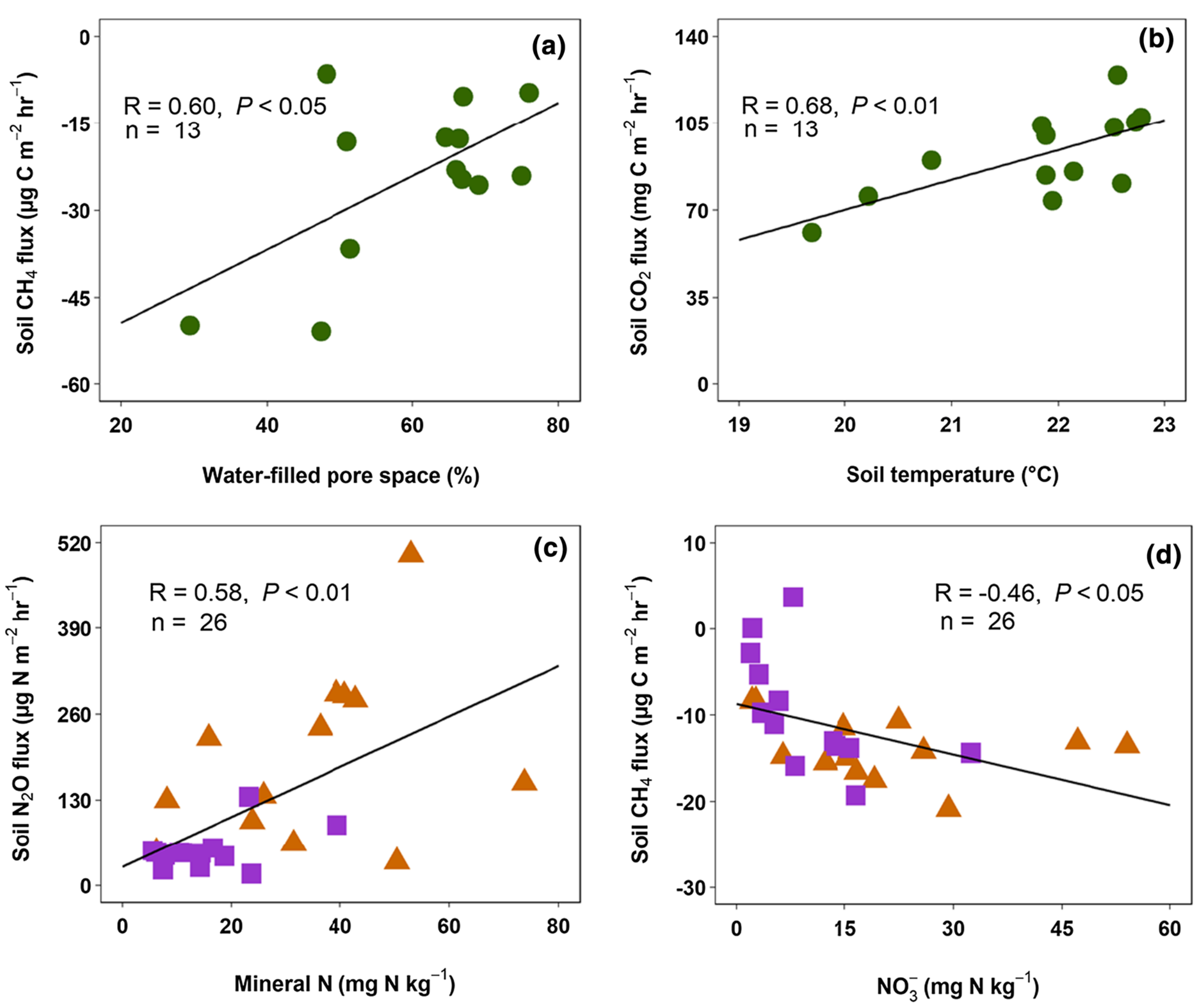

Fig. 2 Pearson's correlations between soil greenhouse gas fluxes (top 0.05 m) and soil factors in the secondary forest (a, b) and small-scale vegetable farms (c, d; plant row: ▲, inter-row: ■) on an Andosol soil in Leyte, Philippines. Each data point is the average of nine plots in the secondary forest and of 10 farms, on each sampling day during monthly measurements from May 2018 to May 2019

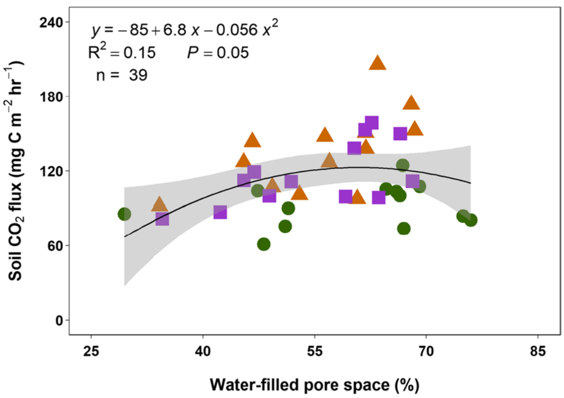

Fig. 3 Regression between soil CO_2 fluxes and water-filled pore space (top 0.05 m) in the secondary forest (●) and small-scale vegetable farms (plant row: ▲, inter-row: ■) on an Andosol soil in Leyte, Philippines. Each data point is the average of nine plots in the secondary forest and of 10 farms, on each sampling day during monthly measurements from May 2018 to May 2019

and extensive (e.g., affecting depths > 0.5 m) (Veldkamp et al. 2020). At our study area, the increased soil bulk density and reduced SOC and total N stocks down to 1-m depth in the vegetable farms (Table 2) had resulted from soil compaction brought about by foot traffic of farmers' management operations, reduction in input of organic materials (e.g., harvest export) (Koga et al. 2020), and low stability of SOC derived from the present land use as observed for Andosol soils (Paul et al. 2008). The decrease in SOC stock in the vegetable farms (Table 2) was at the upper range of those reported for SOC losses from tropical forest-to-cropland conversion (17–58% in the top 0.1 m and 35% in depths > 0.5 m, Veldkamp et al. 2020; 30–66% in the top 0.3–0.4 m, Beheshti et al. 2012; Powers et al. 2011). The decreased soil C:N ratio in the vegetable farms (Table 2) was indicative of highly decomposed organic matter (Murty et al. 2002; Piñeiro et al. 2006; Veldkamp et al. 2020),

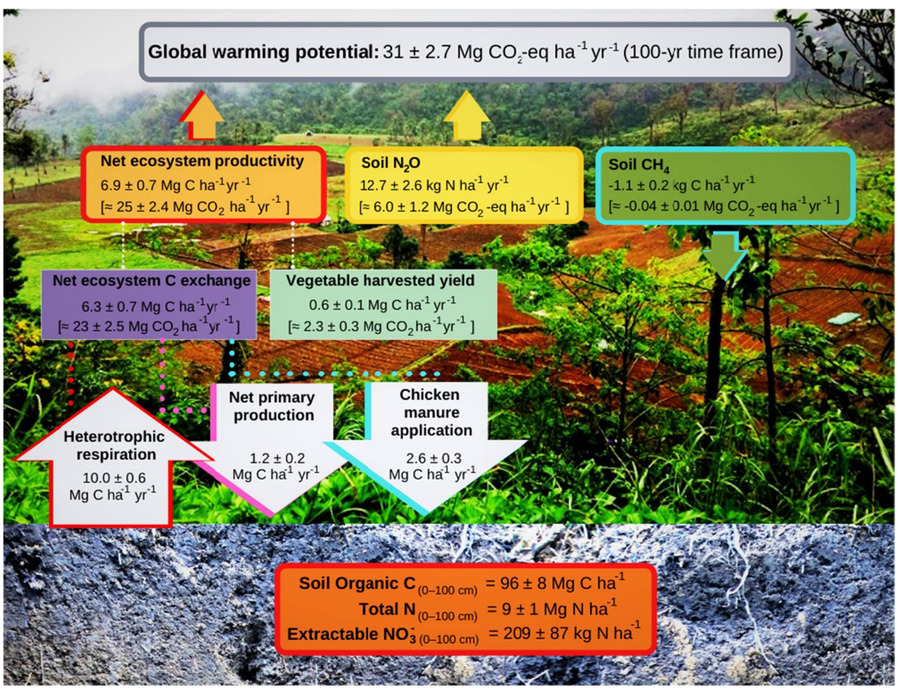

Fig. 4 Soil greenhouse gas balance in vegetable production system on an Andosol soil of smallholders in Leyte, Philippines. Net primary production (NPP) is the sum of harvested yield (measured in the field), herbage production (estimated using the harvest index) and root production (estimated using the root:shoot ratio) for each vegetable type (Electronic supplementary material Table 2). Heterotrophic respiration was represented by the soil CO_2 emission from the inter-row, as this cultivation zone did not have plant roots. Net ecosystem C exchange (NEE) = heterotrophic respiration-(NPP + chicken manure) (Malhi et al. 1999) and net ecosystem productivity (NEP) = NEE + harvested yield exported from the farm (Meijide et al. 2020). Global warming potential (GWP) is the sum of NEP, N_2O and CH_4 in CO_2-equivalent within a 100 year time frame, for which soil N_2O, and CH_4 were multiplied by 298 and 25, respectively (IPCC Guidelines for National Greenhouse Gas Inventories 2006)

which may have resulted from improved decomposition and respiration losses of labile organic matter due to regular plowing and long-term high N fertilization (e.g., Cusack et al. 2011).

Moreover, the reduced organic matter (i.e., SOC and total N) in the vegetable farms had also resulted in reduced ECEC (Table 2), which is largely contributed by organic matter on the variable charge of soils like Andosols (Dahlgren et al. 2004; Dechert et al. 2004; Iwasaki et al. 2017; Veldkamp et al. 2020). Occasional liming as well as chicken manure application (see "Experimental design and farm management practices" subsection above) in vegetable farms had increased the base saturation and decreased Al saturation as compared to the forest (Table 2). Also, the effect of ash deposition (from biomass burning during forest conversion) on increasing base saturation could last one to two decades after forest conversion to croplands (Veldkamp et al. 2020). In the lower depth (0.5–1.0 m), the increased Mg, Ca, and K stocks in the vegetable farms (Table 2) may have reflected the legacy of soil amendments (lime, chicken manure, K fertilizer) and base-containing ashes that have been translocated downwards with time (Andriesse and Schelhaas 1987; van Breemen et al. 1983; Veldkamp et al. 2020). In sum, soil properties related to organic matter loss had decreased (SOC, total N, and ECEC), whereas those associated with soil amendment practices have increased (base cation stocks at deeper depths) in these vegetable farms.

Soil greenhouse gas fluxes from the forest

The annual soil N_2O emission from the forest (Table 3) was lower than those reported for forests in Southeast Asia (see reviewed studies in Electronic supplementary material Table 3). We associated the low soil N_2O emissions from our forest site to the low mineral N levels with the dominance of NH_4^+ over NO_3^- (Table 3), which suggests limited nitrification and thus also low substrate availability for denitrification (Ishizuka et al. 2002; Veldkamp et al. 1998). The low mineral N levels at our forest site were comparable to those forests in Puerto Rico, frequently affected by severe tropical storms (Cusack et al. 2016). Months after storm episodes, mineral N may be elevated but eventually returns to low level as the forest recovers from storm disturbances (Silver et al. 1996). In addition, the low soil N_2O emissions from this forest on Andosol soil may have been a consequence of its high soil porosity throughout the 1-m depth (i.e., low bulk density; Table 2), which may render infrequent or only brief anaerobic conditions (Figs. 2a, 3) despite high rainfall months (see "Site description" subsection above). Thus, the low mineral N levels and the medium-range WFPS (Table 2) indicated the effects of the two main regulating factors (i.e., N availability and aeration status; Davidson et al. 2000) on the low soil N_2O emissions from this forest on Andosol soil.

The soil CH_4 fluxes, displaying the dominance of CH_4 consumption over CH_4 production (Table 3; Figs. 1b, 2a), concurred with other Southeast Asian forests on mineral soils that act as net CH_4 sinks (see reviewed studies in Electronic supplementary material Table 3). Our annual soil CH_4 uptake, however, was at the lower end of those reported for montane forests on Cambisol and Regosol soils and for a lowland forest on Acrisol soil in Indonesia (Hassler et al. 2015; Purbopuspito et al. 2006). We observed a decrease in soil CH_4 uptake with increase in WFPS (Fig. 2a), suggesting the concurrent reduction in diffusion of atmospheric CH_4 into the soil that can limit methanotrophic activity and possible CH_4 production in anaerobic microsites during the wet season (Hassler et al. 2015; Itoh et al. 2012; Tate et al. 2007; Zhao et al. 2019). The overall low CH_4 uptake in our forested volcanic soil was probably also influenced by its low soil mineral N levels (Table 2). Limitation by N availability on soil CH_4 consumption had been reported for forests in Indonesia (Hassler et al. 2015), Panama (Matson et al. 2017; Veldkamp et al. 2013), Ecuador (Wolf et al. 2012), and Cameroon (Iddris et al. 2021) with Andosol, Cambisol and highly weathered Acrisol and Ferralsol soils.

The soil CO_2 efflux, encompassing both microbial and autotrophic respiration (Table 3; Figs. 1c, 2b), was lower than values reported for tropical forests in Southeast Asia on Acrisol and Nitisol soils but slightly higher than a lowland forest on Ferralsol soil (see reviewed studies in Electronic supplementary material Table 3). Our secondary forest had been disturbed by a super typhoon six years prior to our investigation (see "Experimental design and farm management practices" subsection above). The low soil CO_2 efflux from this forest could be due to less autotrophic contribution (which could account for 30% of soil respiration; van Straaten et al. 2011) as well as low heterotrophic activity, if the litter input had not yet recovered to levels prior to this natural disturbance. Although soil CO_2 efflux increased with increase in soil temperature, this effect may also have been confounded by soil moisture, as those low soil CO_2 emissions at the low end of soil temperature (19–20 °C; Fig. 2b) also had low WFPS (29–50%; Fig. 4). Confounding effects of soil temperature and WFPS were commonly observed in tropical forests on a wide range of soil types (Koehler et al. 2009a; Matson et al. 2017). The curvilinear relationship between soil CO_2 emissions and soil moisture in both land uses (Fig. 3) signified root and microbial respiration towards optimum soil moisture condition (WFPS between 55 and 65%), beyond which high soil moisture can hamper gas diffusion as well as limit soil respiration (Koehler et al. 2009a; Matson et al. 2017; Sotta et al. 2007; Tchiofo Lontsi et al. 2020; van Straaten et al. 2011).

Larger soil greenhouse gas fluxes from the vegetable farms than the secondary forest

Soil N_2O emissions from the vegetable farms (Table 3; Figs. 1d) were higher than those reported for Andosol soils in Japan planted to Chinese cabbage but lower than those reported for Andosol soils in Costa Rica utilized for banana plantation and pasture (see reviewed studies in Electronic supplementary material Table 4). In terms of annual soil N_2O emissions (Table 3), our values were similar with those reported for tea plantations (11–19 kg N ha^{-1} yr^{-1}, Hirono and Nonaka 2012; Hou et al. 2015) and 30-year corn field (13–15 kg N ha^{-1} yr^{-1}, Mukumbuta et al. 2017a; 2017b), two of the long-term land uses on Andosol soils in Japan. The large soil N_2O emissions from our vegetable farms had clearly resulted from long-term high N fertilization rates, as evident from the large stocks of mineral N even down to 1-m depth (Tables 4, 5) and as supported by the positive correlation between soil N_2O emissions and mineral N (Figs. 1d, 2c). Although the mineral N values used in the correlation test was measured routinely only at the top 0.05 m (due to logistical limitations) during the year-round measurement, the N_2O emitted at the soil surface was likely contributed by the entire soil depth, where substrates (NH_4^+, NO_3^-, extractable organic N; Table 5) for nitrification and denitrification remained high, similar to the findings of elevated N_2O concentrations throughout the 2-m soil depth under long-term N fertilization to tropical forests (Corre et al. 2014; Koehler et al. 2012). Furthermore, the large soil N_2O emissions from the plant row, with recurring N fertilization and chicken manure application (providing available organic C), also lent support to the primary control of N and C availability, as exemplified by the HIP model (Davidson et al. 2000). The dominance of NO_3^- over NH_4^+ combined with available organic C and mid-range WFPS (Tables 4, 5) resulted in large soil N_2O emissions during the dry season (Fig. 1e) as reduction of NO_3^- to N_2O in such microaerophilic condition (sufficient oxygen condition in relative sense) is energetically more favorable than further reduction to N_2 (Maier 2009; Stolk et al. 2011). Furthermore, the high NO_3^- stocks with depths (Table 5) attested to the anion adsorption capacity of this variable charge Andosol soil. This adsorbed NO_3^- onto the soil's exchange sites is only temporary and is exchangeable when other anions become dominant (e.g., Cl^- and SO_4^{2-} from K-based fertilizers and atmospheric deposition, Formaglio et al. 2020; Kurniawan et al. 2018). This suggests that not only large N_2O emissions but also susceptibility to large NO_3^- leaching (e.g., Formaglio et al. 2020) are the deleterious environmental impacts from this high fertilizer-input vegetable production system on a highly porous Andosol soil under a humid climate.

Soil CH_4 consumption in the vegetable farms (Table 3; Fig. 1e) were within the range of those reported for corn fields in Japan and pastures in Costa Rica all on Andosol soils (see reviewed studies in Electronic supplementary material Table 4). The net CH_4 flux from the soil surface is a net effect of CH_4 production and consumption within the soil. Thus, the lower soil CH_4 uptake in the vegetable farms than in the forest (Table 3) suggests the effect of methanogenic activity that may partly offset methanotrophic activity in the vegetable farms. This was demonstrated by the occasional near-zero soil CH_4 fluxes from the inter-row during the wet season (Fig. 1e) that concurred to the increased soil bulk density in the vegetable farms (Table 2), which may have favored methanogenic activity in anaerobic microsites. Additionally, during the wet season gas diffusion from the atmosphere to the soil could decrease as soil moisture content increased (Fig. 1e), which may limit CH_4 availability to microsites where concurrent methanotrophic activity occur (Iddris et al. 2021; Matson et al. 2017; Tchiofo Lontsi et al. 2020;

Veldkamp et al. 2013). Additionally, the increasing CH_4 uptake with increasing NO_3^- levels (Fig. 2d) in these vegetable farms, reflected the inhibition effect of large NO_3^- levels (Tables 4, 5) on CH_4 production, as NO_3^- is the preferred electron acceptor over bicarbonate (Schlesinger and Bernhardt 2013). The low NO_3^- levels in the inter-rows (Fig. 2d; Table 4) may only have low effect on inhibiting CH_4 production during the wet season, resulting in low net CH_4 uptake, whereas the high NO_3^- levels in the plant-rows may have large impact on dampening CH_4 production, resulting in high net CH_4 uptake. Therefore, on one hand the increased soil bulk density in these vegetable farms may increase occurrence of anaerobic microsites in the inter-rows, but on the other hand, the large NO_3^- levels may dampen CH_4 production in the plant rows, resulting in moderate decrease (although statistically significant) in soil CH_4 uptake in the vegetable farms compared to that in the forest.

The soil CO_2 efflux from the vegetable farms (Table 3) was within the range of those reported for corn fields on Andosol soils in Japan (see reviewed studies in Electronic supplementary material Table 4). While the autotrophic contribution, i.e., on the plant row where roots occurred (Fig. 1f), to soil respiration was possibly low owing to low root biomass (Raich and Tufekcioglu 2000) of shallow-rooted vegetables (Table 1), heterotrophic respiration may have been enhanced by the increased soil temperature (Table 4), frequent deep plowing during two to three cropping periods per year as well as regular chicken manure application (Electronic supplementary material Table 2). Heterotrophic respiration of labile organic matter in the vegetable farms may also been enhanced by the large N fertilization rates (e.g., Cusack et al. 2011).

Consequently, heterotrophic respiration was the most important process contributing to the greenhouse gas budget of these vegetable farms (Fig. 4; Electronic supplementary material Table 2). The heterotrophic respiration from the inter-rows was consistent to that reported for a 30 year corn field on an Andosol soil ($7.8 \pm 1.1 - 10.2 \pm 0.7$ Mg C h a^{-1} yr^{-1}, Mukumbuta et al. 2017a; 2017b). These authors reported GWP (100-yr time frame) of $5.5 \pm 4.8 - 18.4 \pm 1.5$ Mg CO_2-eq ha^{-1} yr^{-1}, which is lower than our estimated GWP from vegetable farms, partly because of the large NPP of corn that offsets the heterotrophic respiration (Mukumbuta et al. 2017a). Similarly, the GWP (100-yr time frame) of 3.1–4.2 Mg CO_2-eq ha^{-1} yr^{-1} in vegetable fields on the same soil type in Japan were lower than our values, likely ascribed to conventional tillage alongside lower N fertilization rate (130 kg N ha^{-1}) as cultivation practices (Yagioka et al. 2015). However, our GWP budget of the vegetable farms was in agreement to that from a newly established oil palm plantation on Acrisol soil in Jambi, Indonesia (38 Mg CO_2-eq ha^{-1} yr^{-1}; Meijide et al. 2020). These other studies as well as our present estimate of GHG footprint of smallholder vegetable production revealed the need for converted land uses to employ sustainable management practices in order to prevent soil degradation and thus avoiding further forest conversion. As microbial decomposition of soil organic matter was the largest contributor to the GHG footprint of our vegetable farms and C losses were substantial (as illustrated by the NEE in Fig. 4), the intensively managed soils currently continue to lose large amount of C every year. It is likely that these losses will become less with time, when a new steady state between C input and output will be reached. However, this will probably come at the costs of serious soil degradation, which will likely also affect the productivity of these soils. Management practices should thus be tailored towards increasing organic matter storage, which in turn enhances nutrient recycling (thereby minimizing dependency on large fertilizer inputs; Formaglio et al. 2021), erosion resistance (through aggregate stabilization), and water filtration (through recycling of excess nutrients) (Veldkamp et al. 2020).

Conclusions

The findings support our hypothesis that soil N_2O and CO_2 emissions increased while soil CH_4 uptake decreased in vegetable farms compared to the secondary forest. These soil GHG patterns were attributed to large N fertilization rates, increased soil temperature, and deep soil tillage in the vegetable farms, favoring increased soil CO_2 and N_2O emissions. Additionally, increased soil bulk density in the vegetable farms may promote anaerobic microsites during the wet season and limit CH_4 diffusion from the atmosphere to the soil, resulting in reduced soil CH_4 uptake. The large stock of mineral N down to 1-m depth, dominated by NO_3^-, also suggests susceptibility to leaching; this, along with excessive pesticide applications, can have deleterious impact on water quality-an environmental effect that remains uninvestigated in our study area. There is an urgent need for knowledge support to these vegetable farmers on sustainable management practices, which should be geared towards conserving soil fertility and minimizing environmental impact in order to prevent further forest conversion. The supports may include field-based trials for improving soil organic matter storage and efficient cycling of nutrients: e.g., mulching (which also reduces herbicide use; Formaglio et al. 2021), perennial buffer strips (to intercept leached nutrients; McKergow et al. 2004), reduce soil tillage, and crop rotations between deep- and shallow-rooted vegetables or between high and low nutrient-demanding vegetables. These management practices, overall, can serve as tools to offset the current estimated GWP in small-scale vegetable farms. In addition, qualitative soil analysis must be accessible to farmers as an integrative approach in fertilizer use management (e.g., as reference for synchronized split applications of fertilizers). Most importantly, our findings served as primary contribution to the data gaps and inconsitencies of available data in the agriculture sector's GHG inventory of the Philippines (Kawanishi et al. 2019). Lastly, we would like to stress that strategies and policies to reduce soil GHG emissions should aim at incentivizing smallholders in employing sustainable crop management practices.

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Authors' contributions CMOQ, EV, and MDC designed the study, interpreted the data, and wrote the manuscript. CMOQ performed the field measurements, laboratory works, and material preparation. SBL, MJMB, and AOA assisted in extensive field works. CMOQ and MDC conducted the data analyses. All authors approved the final manuscript.

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Data availability The data of this study are deposited in the institutional data repository of Goettingen University (https://doi.org/10.25625/RHF7BU).

Code availability Data analysis has been conducted using the free statistical software R as described in the "Statistical Analysis" section.

Declarations

Conflicts of interest The authors declare no conflicts of interest.

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