Environmental and economic trade-offs of using composted or stockpiled manure as partial substitute for synthetic fertilizer

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Abstract
Manure generated from livestock production could represent an important source of plant nutrients in substitution of synthetic fertilizer. To evaluate the sustainability of partially substituting synthetic fertilizer with soil organic amendments (OAs) in horticulture, an economic and greenhouse gas (GHG) budget was developed. The boundary for analysis included manure processing (stockpiling vs. composting) and transport and spreading of manure and compost (feedlot and chicken) in intensively cultivated horticultural fields. The OA field application rates were calculated based on the nitrogen supplied by OAs. The GHG budget based on directly measured emissions indicates that the application of composted manure, in combination with reduced fertilizer rate, was always superior to stockpiled manures. Compost treatments showed from 9 to 90% less GHG emissions than stockpiled manure treatments. However, higher costs associated with the purchase and transport of composted manure (three times higher) generated a greater economic burden compared with stockpiled manure and synthetic fertilizer application. The plant nutrient replacement value of the OAs was considered only for the first year of application, and if long-term nutrient release from OAs is taken into account, additional savings are possible. Because the income from soil carbon sequestration initiatives in response to OA application is unlikely to bridge this financial gap, particularly in the short term, this study proposes that future policy should develop methodologies for avoided GHG emissions from OA application. The combined income from soil carbon sequestration and potentially avoided GHG initiatives could incentivize farmers to adopt OAs as a substitute for synthetic fertilizers, thereby promoting more sustainable farming practices.

Abbreviations: CCM, composted chicken manure; CFM, composted beef feedlot manure; CM, chicken manure; dw, dry weight; EF, emission factor; FM, beef feedlot manure; fw, fresh weight; GHG, greenhouse gas; GREET, Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation; IPCC, Intergovernmental Panel on Climate Change; OA, organic amendment; PAN, plant available nitrogen.
1 | INTRODUCTION

Before the introduction of synthetic fertilizer, the application of organic amendments (OAs) such as animal manures to agricultural soils was the traditional way of fertilizing crops because between 55 and 95% of the nitrogen (N) and about 70% of the phosphorus (P) ingested by livestock are excreted through urine or feces (Menzi et al., 2010). The intensification of global meat consumption has resulted in an exponential increase in manure production, contributing to the accumulation of nutrient-rich manure around areas dedicated to livestock feeding operations (Spiegal et al., 2020). These manure-nutrient rich “hotspots” pose several environmental challenges, such as the emissions of greenhouse gases (GHGs) and nutrients leaching to groundwater and their subsequent movement into rivers and estuaries, causing eutrophication of the terrestrial water body ecosystem (Mottet et al., 2017). Relocating surplus manure nutrients from nutrient hotspots to nutrient-deficit cropland can alleviate the environmental burden associated with livestock production operations while reducing the need for fertilizer imports (Powers et al., 2019; Spiegal et al., 2020). Nevertheless, sustainable relocation and land application of OAs require several economic and environmental considerations.

There are indisputable benefits of OA application as a means of sustaining soil physical and chemical fertility (Diacono & Montemurro, 2010). However, these benefits are often associated with application rates many times larger than what is necessary to supplement synthetic fertilizer (Quilty & Cattle, 2011). The over-application of OAs might exacerbate soil nitrous oxide (N\textsubscript{2}O), a potent GHG, due to soil inorganic N levels exceeding crop N demand (De Rosa et al., 2018; VandezAag et al., 2011). The careful consideration of OA types and application rates is crucial to maximizing its benefits while minimizing the associated environmental impacts. De Rosa et al. (2018) compared the application of composted and stockpiled manure and demonstrated that the balanced application of composted manure with reduced synthetic fertilizer to match crop needs reduces soil N\textsubscript{2}O. This reduction in GHG emissions was attributed to the higher stability of organic matter, the larger content of recalcitrant material, and the lower mineral N availability for nitrifying and denitrifying microorganisms in composted rather than raw manures.

The composting process has also been demonstrated to reduce pathogens and weed seeds and to reduce material weight and volume (Eghball, 2000) and thus is a preferred way to improve export potential for dry feedlot manure (Spiegel et al., 2020). However, manure composting also emits GHGs, and estimation of these emissions is currently based on default emission factors (EFs) recommended by the Intergovernmental Panel on Climate Change (IPCC). The default EFs for composting have been largely questioned, and literature values vary widely (Ba et al., 2020) from 0.004 g CH\textsubscript{4}−C kg\textsuperscript{−1} dry matter and 0.01 g N\textsubscript{2}O−N kg\textsuperscript{−1} total N (Biala et al., 2019) to 0.85 g CH\textsubscript{4}−C kg\textsuperscript{−1} dry matter and 10.40 g N\textsubscript{2}O−N kg total N (Fillingham et al., 2017). Therefore, although there are numerous benefits to OA application, it is still uncertain whether the life cycle GHG emissions generated from transport (especially long-haul) and the composting process offset the reduction in soil GHGs observed on application of composted OAs.

In addition to the environmental impact, the development of sustainable farming systems should ensure the economic viability of OA application because this will ultimately determine the adoption of this practice by farmers. Farmers would only purchase OAs as a substitute or in combination with synthetic fertilizer if the total cost associated with the use of OAs is lower than or comparable to that of synthetic fertilizer. Although the substitution of synthetic fertilizers with OAs could reduce the environmental impact associated with the industrial production of synthetic fertilizer, in circumstances where the economic revenue derived from OAs is lower than synthetic fertilizer, economic support may be needed to incentivize adoption. However, the economic revenue derived solely from the nutrient replacement value of OAs is difficult to quantify because it depends on geographical context and local regulation policies (Leip et al., 2019). Therefore, the environmental and economic costs and benefits of the use of OAs as a substitute for synthetic fertilizer cannot be understood unless the entire supply chain is considered.

The objective of this study was to quantify the environmental and economic trade-offs associated with using composted versus stockpiled manure as a partial substitute for synthetic fertilizer. Greenhouse gas and economic budgets were developed for an OA supply chain case study including measurements of total GHG emissions and economic cost-benefit generated from the manure processing operations (i.e., composting and stockpiling), transport to the field, and manure spreading in a horticultural crop rotation.

2 | MATERIALS AND METHODS

To evaluate the economic and environmental trade-offs of applying composted or stockpiled manures to agricultural
soils, total GHG emissions along with the associated costs were estimated from the composting and stockpiling process, transport of the OA from processing site to field site, and field application.

2.1 Horticultural field experiment

In this case study, the intensively cultivated agricultural soil for vegetable crop production was chosen as potentially representing the upper end of field GHG emissions (Rezaei Rashti et al., 2015). Intensively cultivated agricultural soils for vegetable crop production are characterized by high N-fertilizer application rates often combined with OA applications, leading to annual total N application rates ranging from 220 to 1,145 kg N ha\(^{-1}\) yr\(^{-1}\) (De Rosa et al., 2018; Porter et al., 2017; Scheer et al., 2017).

The details of the experimental design and plant nutrients (N, P, and K) management strategies can be found in De Rosa et al. (2016). Briefly, a field experiment was conducted from September 2013 to September 2014 at Gatton Research Station in the Lockyer Valley, a major vegetable producing region in southeast Queensland, Australia (27°32′56″ S, 152°19′39″ E; 100 m asl). The crop rotation under investigation comprised a succession of three vegetable crops: green beans (Phaseolus vulgaris L.), broccoli (Brassica oleracea var. italica) and lettuce (Lactuca sativa L.). Sorghum [Sorghum bicolor (L.) Moench] was used as a cover crop to reduce N losses during the summer fallow period between green beans and broccoli.

Ten fertilizer treatments were arranged in a randomized block design with four replicates (1.5 m by 10 m with 1.5-m buffer):

1. A conventional N-fertilizer rate (CONV, 310 kg of N ha\(^{-1}\)) and 65 kg of P ha\(^{-1}\) and 178 kg of K ha\(^{-1}\) for the entire rotation based on local farm management.
2. Zero N input treatment (0N) was used to account for background soil GHG emissions.

Eight treatments of OA were derived from the factorial combination between (a) four organic amendments: conventionally stockpiled beef feedlot manure (FM) and chicken manure (CM) and aerated turned composted chicken manure (CCM) and composted beef feedlot manure (CFM) and (b) two levels of synthetic N-fertilizer: reduced (+Nd) and a conventional (+\(N_{\text{CONV}}\)) N-fertilizer rate. The factorial combination between OA and different N-fertilizer rates included:

3. Composted chicken manure plus conventional N-fertilizer rate (CCM+\(N_{\text{CONV}}\)).
4. Composted chicken manure plus reduced N-fertilizer rate (CCM+\(N_{\text{Nd}}\)).
5. Stockpiled chicken manure plus conventional N-fertilizer rate (CM+\(N_{\text{CONV}}\)).
6. Stockpiled chicken manure plus reduced N-fertilizer rate (CM+\(N_{\text{Nd}}\)).
7. Composted feedlot manure plus conventional N-fertilizer rate (CFM+\(N_{\text{CONV}}\)).
8. Composted feedlot manure plus reduced N-fertilizer rate (CFM+\(N_{\text{Nd}}\)).
9. Stockpiled feedlot manure plus conventional N-fertilizer rate (FM+\(N_{\text{CONV}}\)).
10. Stockpiled feedlot manure plus reduced N-fertilizer rate (FM+\(N_{\text{Nd}}\)).

The OAs were added at the start of the crop cycle on 2 Sept. 2013 and incorporated with a rotary hoe to a depth of 0.2 m. The OA application rates were determined to match the basal N-fertilizer application rate in the CONV treatment (35 kg N ha\(^{-1}\)) considering the fraction of plant available N (PAN, NO\(_3^-\) + NH\(_4^+\)) content in the OAs. The amount of OA applied and the respective concentrations of N, P, and K provided at application are reported in Table 1.

The +Nd N-fertilizer rate was calculated by subtracting the estimated PAN delivered by the OA from the CONV N-fertilizer application rate. The PAN delivered by OA was estimated by multiplying the amount of organic N supplied by OA with mineralization rate coefficients (Table 1) taken from the literature (Eghball et al., 2002; Hartz et al., 2000) divided into quartiles (3 mo) for the entire duration of the crop rotation using Equation 2 described in De Rosa et al. (2017). The application of OAs provided 29, 278, 299, and 157 kg P ha\(^{-1}\) for CM, CCM, FM, and CFM, respectively, and 91, 431, 995, and 498 kg K ha\(^{-1}\) for CM, CCM, FM, and CFM, respectively (Table 1). For the P and K fertilization strategies, only the CONV and 0N treatments received supplemental P and K synthetic fertilizer (67 kg P ha\(^{-1}\) and 179 kg K ha\(^{-1}\)), whereas OA treatments did not receive any P and K synthetic fertilizer supplements.

The yearly amount of fertilizer and estimated N available from OAs for the crop rotation are listed in Table 2.

Plots were irrigated at least once a week using an overhead sprinkler irrigation system following standard farming practice. Crop residues were incorporated at the end of each crop growing phase with a rotary hoe to 0.2 m depth. Permanent crop beds (1.5 m by 10 m with 1.5-m buffer) that accommodated two plant rows were reformed following the incorporation of crop residues.

2.1.1 Plant and soil analysis

At each harvest, crop yield and total biomass production were measured by harvesting all plants (15 m\(^2\)) within each experimental plot. Representative plant samples (four per plot) were
TABLE 1 Composition and application rates of chicken (CM) and feedlot (FM) manures and composted chicken (CCM) and feedlot (CFM) manures on dry weight basis at Gatton Research Facility, Queensland, Australia (2013–2014)

| Composition  | CM | CCM | FM | CFM |
|--------------|----|-----|----|-----|
| H2O, %       | 60 | 34  | 16 | 30  |
| Organic C, % | 24.2 | 23 | 20 | 17 |
| Total N, %   | 7.2 | 1.9 | 1.9 | 2.7 |
| C/N ratio    | 3.3 | 12 | 10.5 | 6.3 |
| NO3−–N, mg kg⁻¹ | 115 | 93.9 | 87 | 32 |
| NH4+-N, mg kg⁻¹ | 2.4845 | 2.819 | 829 | 1,403 |
| P, %         | 2.4 | 2.2 | 0.85 | 0.9 |
| K, %         | 6 | 3.5 | 2.9 | 2.9 |
| S, %         | 0.54 | 0.64 | 0.65 | 0.6 |
| Field application, Sept. 2013 |
| OA application, t dry wt. ha⁻¹ | 1.4 | 12.5 | 35.3 | 17.5 |
| Total N, kg N ha⁻¹ | 102 | 240 | 699 | 466 |
| Organic N, kg N ha⁻¹ | 67 | 205 | 660 | 430 |
| PAN, kg N ha⁻¹ | 35 | 35 | 35 | 35 |
| P, kg P ha⁻¹ | 29 | 278 | 299 | 157 |
| K, kg K ha⁻¹ | 91 | 431 | 995 | 498 |
| S, kg S ha⁻¹ | 8 | 79 | 230 | 106 |
| Organic C, kg C ha⁻¹ | 340 | 2,884 | 7,059 | 2,882 |
| Annual mineralized organic N |
| MR, % | 33 | 12 | 23 | 9 |
| Mineralized PAN, kg N ha⁻¹ yr⁻¹ | 22 | 25 | 157 | 39 |

Note: MR, estimated annual mineralization rate (De Rosa et al., 2017; Egghall, 2000; Hartz et al., 2000); OA, organic amendment; PAN, plant available N (NO3− + NH4+).

oven-dried for 24–48 h at 70 °C and subsequently ground and analyzed for total N and C content using a Leco Trumac CNS Analyzer (LECO Corp.). Soil organic C (SOC) was measured from each plot prior to OA application in September 2013 and at the end of the field experiment in September 2014 by collecting four subsamples of soil per plot (0–0.15 m) and analyzed using Leco Trumac CNS Analyzer.

2.1.2 GHG measurements from the field experiment

For the horticultural field experiment, high temporal resolution measurements of soil N2O and CH4 fluxes from each experimental plot were collected with an automated chamber sampling system as described in De Rosa et al. (2016) from the treatments that received CM, CCM, CONV, and 0N. Nitrous oxide and CH4 emissions were also measured from the treatments that received FM and CFM, using the manual closed chamber method.

The automated chamber sampling system used in the horticultural field experiment consists of transparent acrylic static chambers (0.5 by 0.5 by 0.15 m) fixed on stainless steel bases inserted 0.1 m into the soil and equipped with pneumatically operated lids. The chambers were linked to a computerized sampling unit and an in situ gas chromatograph (SRI GC 8610C) equipped with a 63Ni electron capture detector for N2O, a flame ionization detector for CH4, and an infrared gas analyzer (LI-820, LI-COR) for CO2. During closure (1 h), each chamber was sampled every 15 min with a known calibration standard every fifth measurement, obtaining eight fluxes per day from each chamber. Chambers were opened during irrigation events, and a tipping bucket rain gauge connected to the system facilitated the automatic opening of the lids to ensure rainfall entered the chambers.

In the FM and CFM treatments, the manual gas samples were taken between 10 AM and 12 PM every 2–3 d during the first 2 wk after fertilization and incorporation of OAs and crop residues and weekly for the rest of the time. The polyethylene manual chambers were the same dimensions as the automated chambers. Chamber headspace gas samples (20 ml) were collected 0, 30, and 60 min after closure by connecting a syringe to a two-way Luer lock tap installed in the lid of the chamber and then injected into a pre-evacuated 12-ml glass vial (Friedl et al., 2017; Scheer et al., 2017). Manual gas samples were analyzed for N2O and CH4 by laboratory-based gas chromatography (GC-2014, Shimadzu). The N2O and CH4 fluxes from the automated chambers were calculated from the slope of the linear concentration increase of the four and three measurements for the automated and manual chambers, respectively, taken over the 60-min chamber closure time (Scheer et al., 2014). Mean daily fluxes from the automated system (g N2O·N ha⁻¹ d⁻¹ and g CH4·C ha⁻¹ d⁻¹) were obtained by averaging sub-daily fluxes over a 24-h period from each chamber. Data gaps were filled using linear interpolation by chamber across the missing day (Dorich et al., 2020). Cumulative N2O fluxes (g N2O–N ha⁻¹ and g CH4–C ha⁻¹) were calculated by summing the daily average of each individual chamber over the 1-yr crop rotation.

Two different approaches were used to calculate the annual N2O EfS from the horticultural field experiment for the treatments that received OAs; both used 0N treatment to correct for soil background emissions. The first approach (EF) took into account the yearly total N applied with OAs plus N-fertilizer following the Tier 1 methodology of the IPCC (Kroeze et al., 1997); the second approach (EF0), proposed by De Rosa et al. (2016), only accounted for the estimated N mineralized from OA (cumulative PAN) plus the N-fertilizer (Table 2). The Benjamini and Hochberg procedure (Benjamini & Hochberg, 1995) was performed to assess significant differences.
TABLE 2  Annual rate of N application, calculated by summing the N-fertilizer and N supplied from organic amendments over the entire crop cycle at Gatton Research Facility (2013–2014)

| Treatment | N-fertilizer | N OA | PAN | N-fertilizer + N OA | N-fertilizer + PAN | Fertilizer reduction in Rd from CONV |
|-----------|--------------|------|-----|---------------------|-------------------|-------------------------------------|
| CONV      | 310          | –    | –   | 310                 | 310               |                                     |
| CM+N CONV | 310          | 102  | 57  | 412                 | 367               |                                     |
| CM+N Rd   | 252          | 102  | 57  | 355                 | 310               | 19                                  |
| CCM+N CONV| 310          | 240  | 60  | 550                 | 370               |                                     |
| CCM+N Rd  | 249          | 240  | 60  | 490                 | 310               | 20                                  |
| FM+N CONV | 310          | 698  | 192 | 1,008               | 502               |                                     |
| FM+N Rd   | 118          | 698  | 192 | 816                 | 310               | 62                                  |
| CFM+N CONV| 310          | 466  | 74  | 776                 | 384               |                                     |
| CFM+N Rd  | 236          | 466  | 74  | 702                 | 310               | 24                                  |

* CCM, composted chicken manure; CFM, composted feedlot manure; CM, stockpiled chicken manure; CONV, conventional N, P, and K application rate; FM, stockpiled feedlot manure; +N CONV, conventional N fertilizer rate; +N Rd, reduced N fertilizer rate.

**Total N supplied with the application of organic amendments.

***Mineral N supplied with the application of organic amendments as the sum of the mineral N content at the time of organic amendments application (September 2013) and the estimated N mineralized over time.

\[ \text{differences (} p < .05 \text{) in cumulative } N_2O \text{ and } CH_4 \text{ emissions and on EF.} \]

2.2 | Composting and stockpiling experiments

The GHG measurements during stockpiling and composting of cattle feedlot manure and layer chicken manure were taken at two separate facilities near Toowoomba, Queensland, Australia (27°34′04.6″ S, 151°56′10.1″ E). Manure processing at the two sites was carried out as follows.

2.2.1 | Beef cattle feedlot

The manure stockpile (866 t fresh weight [fw]) was established by compacting manure collected from feed pens with a front-end loader as is typical practice for the industry. The stockpile remained untouched for the duration of the GHG measuring period (153 d). Manure (211 t fw) was also composted for 104 d, using windrow composting turned with a front-end loader 30, 60, and 90 d after establishment. Water was supplied with a hand-held hose each time the windrow was turned.

2.2.2 | Layer chicken manure composting facility

The manure stockpile (78 t fw) was established by tipping manure from the delivery truck, which then remained untouched for the duration of the monitoring period (154 d). Manure was also composted for 161 d after blending it with sawdust at a 4:1 (v/v) ratio, which equates to approximately 10:1 on a weight basis. The windrow comprised around 201 t (fw) manure and 20.6 t (fw) sawdust and was turned frequently (14, 25, 33, 40, 57, 71, 90, 117, and 124 d after windrow establishment) with a self-propelled straddle windrow turner. Water was supplied via the windrow turner and added as needed.

2.2.3 | GHG measurements during the composting and stockpiling experiments

Five round greenhouse gas sampling chambers (diameter, 0.235 m; height, 0.3 m with 0.2 m headspace) were placed equidistant on each of the stockpiles and the windrows. Gas sampling occurred twice per week for the first 6 wk of the trial and then at weekly intervals. Sampling from each chamber involved taking three samples at 2-min intervals, starting 2 min after the lid of the chamber was closed. Samples were drawn from the chamber via a three-way valve by extracting 20 ml of gas with a syringe and injecting 20 ml into a 12-ml evacuated vial. Samples were analyzed for N₂O and CH₄ at the Queensland University of Technology laboratory using a Shimadzu GC-2014 gas chromatograph. Fluxes were calculated as described for the horticultural field experiment. The emissions for periods between sampling events were estimated by means of linear interpolation, providing daily and cumulated emission values for each chamber for the entire trial period. Total emissions were calculated by multiplying emissions per gas chamber surface area (0.055 m²) by the
flat top surface area of stockpiles and windrows. Windrows had a trapezoidal shape, with the top surface area being measured after establishment and each turning. Emissions were measured only from the flat top surface area because it was shown that 100% of CH$_4$ and 91% of N$_2$O is emitted via the top of large windrows (Andersen et al., 2010). Emissions per ton of manure and composting feedstock (chicken manure) were calculated as cumulative emissions in relation to the mass of manure used for establishing the stockpiles and windrows. Estimated carbon losses during stockpiling (feedlot manure: 2.4%; chicken manure: 0.4%) and composting (feedlot manure: 34.0%; chicken manure: 39.2%) were used to convert input-based (per t dry weight [dw] manure) to output-based (per t dw manure or compost applied) emissions.

### 2.3 Calculation of economic and GHG budget for OA treatments

The economic budget of different OA treatments was calculated based on the local purchase price of OAs and synthetic fertilizers (i.e., N, P, and K) plus the costs associated with the transport of the materials to farm (110 km traveling distance from manure-nutrient hotspots to nutrient-deficit cropland) and the handling and spreading of OAs and synthetic fertilizers on field. The purchase prices of CM and CCM used in our analysis were US$18.5 and US$44.6 fw t$^{-1}$; those for FM and CFM were US$8.9 and US$38.6 fw t$^{-1}$, respectively. The purchase prices of synthetic fertilizers were considered on an element basis, including US$1.4 kg N, US$6.2 kg$^{-1}$ P, and US$2.4 kg$^{-1}$ K. For both OAs and fertilizers, we used a transport cost of US$0.11 t$^{-1}$ km$^{-1}$ and an application (spreading) cost of US$3.1 t$^{-1}$. The information regarding prices of products (OAs and fertilizers) and service (transport and spreading) were obtained from local producer and national consultants.

The GHG emission budget used a cradle-to-field boundary and considered the emissions from different sources, including the production, transport, and field spreading of OAs and synthetic fertilizers and soil GHG emissions from the horticultural field over a 1-yr timeframe. The emissions associated with the production of OAs (i.e., stockpiling and composting processes) and soil emissions were captured with the chamber measurements described in Sections 2.2.1 and 2.2.2. Because CO$_2$ emissions from biological systems are considered to rapidly cycle and occur anyway in natural systems during the breakdown of organic residues, these biogenic emissions are considered to have no net global warming effect (Christensen et al., 2009). Due to only a portion of the OAs organic C being mineralized during the first-year application, the possible changes in soil C content observed in the OAs treatments are considered transient and therefore are not included in the environmental budget.

Methane and N$_2$O emission values were converted to carbon dioxide equivalents (CO$_2$-eq) using global warming potential factors of 28 for CH$_4$ and 265 for N$_2$O (IPCC, 2019).

To calculate the GHG emissions associated with the production of synthetic fertilizer as well as the transport and application of OAs and fertilizers, we used the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model (GREET) (Argonne National Laboratory, 2019; Wang et al., 2020). The GREET model’s default emission coefficients were used for the manufacturing and for the transport and distribution of fertilizers, which are 3.87, 0.65, and 0.53 kg CO$_2$-eq kg$^{-1}$ of N, P, and K, respectively. An emission of 4.5 kg CO$_2$-eq ha$^{-1}$ was used for field spreading of fertilizers, as reported by Hanna (2005).

For our case study, composts and manures were transported directly from the processing facilities to the study sites for application. We assumed these organic materials were transported using an industrial truck with a capacity of 32 tons per load. The fuel consumption of this truck was estimated under different road conditions (i.e., urban vs. rural) for a round trip from the manufacturing facility to the application site (loaded) and nonbackloading return (unloaded) (ROU-UNSW, 2006). The transport distances in urban and rural areas were 10 and 100 km, respectively. Composts and manures were applied to the field using an OA spreader with a fuel consumption rate of 23.5 L h$^{-1}$ fuel and a handling rate of 200 t d$^{-1}$ (∼9 working hours) (ROU-UNSW, 2006). The fuel consumptions were converted to GHG emissions using GREET’s coefficient of 3.220 g CO$_2$-eq L$^{-1}$ of diesel combusted (Argonne National Laboratory, 2019). These calculations resulted in emission rates of 3.3 kg CO$_2$-eq t$^{-1}$ km$^{-1}$ and 3.4 kg CO$_2$-eq t$^{-1}$ for OA transport to field and on-field spreading, respectively.

### 3 RESULTS AND DISCUSSION

#### 3.1 Economic budget

The application of OAs in combination with either the reduced fertilizer rate (+N$_{Rd}$) and the conventional N fertilizer rate (+N$_{CONV}$) showed no significant positive or negative effect on crop production in comparison to CONV (Table 3). Total yields were 4.1 ± 0.5 to 5.1 ± 0.8 t dw ha$^{-1}$ for FM+N$_{Rd}$ and CFM+N$_{CONV}$, respectively. This yield level was achieved despite reducing synthetic fertilizers of up to 62% of N and 100% of P and K compared with the CONV treatment. Although yield was similar among OA treatments, our economic budget calculations revealed that some treatments were not economically beneficial when compared to the CONV treatment (Figure 1). On average, the costs of using CFM and CCM (at either +N$_{Rd}$ and +N$_{CONV}$ fertilizer rate) were 33% higher (US$440 ha$^{-1}$) (Table 4) than CONV. Only treatments that received CM and FM+N$_{Rd}$ rates had lower
TABLE 3 Annual yield, total $\text{N}_2\text{O}$ emissions and changes in soil organic C at the field experiment at Gatton Research Facility (2013–2014)

| Treatments | CONV | CM+$\text{N}_{\text{conv}}$ | CCM+$\text{N}_{\text{conv}}$ | CM+$\text{N}_{\text{Rd}}$ | CCM+$\text{N}_{\text{Rd}}$ | FM+$\text{N}_{\text{conv}}$ | CFM+$\text{N}_{\text{conv}}$ | FM+$\text{N}_{\text{Rd}}$ | CFM+$\text{N}_{\text{Rd}}$ | 0N |
|-----------|------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|------|
| Crop production | | | | | | | | | | |
| **Horticultural field experiment** | | | | | | | | | | |
| Yield, t dry wt. ha$^{-1}$ | 4.4 ± 0.4a | 4.2 ± 0.6a | 4.4 ± 0.6a | 4.1 ± 0.3a | 4.5 ± 0.5a | 4.5 ± 6a | 5.1 ± 0.8a | 4.1 ± 0.5a | 4.6 ± 0.2a | 2.0 ± 1.2b |
| N removed, kg N ha$^{-1}$ | 196.4 ± 4.1a | 181.2 ± 16.3a | 189.6 ± 11.4a | 172.8 ± 12.6a | 200.7 ± 5.4a | 187.1 ± 18.5a | 195.1 ± 25.8a | 152.9 ± 1.3a | 188.6 ± 5.6a | 65 ± 33b |
| $\text{N}_2\text{O}$ emissions | | | | | | | | | | |
| Total flux, g $\text{N}_2\text{O}$–N ha$^{-1}$ yr$^{-1}$ | 1,179 ± 213cde | 1,523 ± 379bcd | 1,127 ± 207de | 1,748 ± 284bc | 1,670 ± 309bcd | 1,984 ± 377b | 1,876 ± 683b | 1,984 ± 377b | 1,670 ± 309bcd | 862 ± 10e |
| Annual $\text{N}_2\text{O}$ emission factors, % | | | | | | | | | | |
| Total N applied (EF) | 0.10 ± 0.01ab | 0.16 ± 0.009ab | 0.05 ± 0.03b | 0.25 ± 0.08a | 0.02 ± 0.06b | 0.23 ± 0.05a | 0.13 ± 0.08ab | 0.14 ± 0.04ab | 0.11 ± 0.08 ab | – |
| Mineralized+N-Fer (EF$_{\text{OA}}$) | 0.10 ± 0.01bc | 0.18 ± 0.01abc | 0.07 ± 0.05c | 0.29 ± 0.09abc | 0.04 ± 0.09c | 0.46 ± 0.10a | 0.27 ± 0.18abc | 0.40 ± 0.13ab | 0.26 ± 0.18abc | – |
| CH$_4$ emissions | | | | | | | | | | |
| Total flux, g CH$_4$–C ha$^{-1}$ yr$^{-1}$ | 195.9 ± 246.5 | –182.1 ± 110.3 | –87.4 ± 107.2 | –108.4 ± 201.5 | –174.4 ± 199 | –102.2 ± 183.4 | –128.6 ± 147.8 | –132.9 ± 72.5 | –165.12 ± 72.7 | – |
| Soil organic C | | | | | | | | | | |
| SOC, t C ha$^{-1}$ | 25.9 ± 1.12c | 27.5 ± 1.30bc | 28.3 ± 1.90abc | 26.5 ± 1.30bc | 28.7 ± 1.90abc | 30.9 ± 1.9a | 28.2 ± 1.8abc | 29.9 ± 2.1ab | 28.8 ± 1.9abc | – |
| Difference in SOC, Sept. 2013–Sept. 2014, % | –4.3 ± 4.1c | –1.9 ± 4.1bc | 4.1 ± 6.5abc | –2.3 ± 4.2bc | 5.5 ± 6.5abc | 12.22 ± 4.2a | 3.9 ± 6.4abc | 9.6 ± 5.9ab | 5.9 ± 3.4abc | – |
| Composting and stockpiling experiments | | | | | | | | | | |
| GHG emissions | | | | | | | | | | |
| g CH$_4$–C t$^{-1}$ fw OA | 179.0.03 | 102.46 | 49.3 | 93.31 | 0.18 | 3.6 | 25.40 | 4.84 | 2.9 | 8.55 | 16.07 |
| g $\text{N}_2\text{O}$–N t$^{-1}$ fw OA | | | | | | | | | | |
| kg CO$_2$-eq t$^{-1}$ fw OA (N$_2$O+CH$_4$) | | | | | | | | | | |

Note: Total CH$_4$, N$_2$O, and CO$_2$ from the composting and stockpiling experiment. CCM, composted chicken manure; CFM, composted feedlot manure; CM, stockpiled chicken manure; CONV, conventional N, P, and K application rate; FM, stockpiled feedlot manure; fw, fresh weight; OA, organic amendment; +N$_{\text{conv}}$, conventional N fertilizer rate; +N$_{\text{Rd}}$, reduced N fertilizer rate; SOC, soil organic carbon. Means denoted by different letters indicate significant differences between treatments ($p < 0.05$).
costs (up to 60% or ~US$800 ha$^{-1}$) (Table 4) in comparison to CONV. However, the CM treatments received a reduced amount of P and K. For the CM treatments, the need for supplemental P and K fertilizer could offset the observed economic advantages in comparison to CONV.

Stockpiled manures were more economically competitive than composted manures due to the lower costs associated with material purchase. The ratio between the fertilizer replacement value (the reduced cost due to lower input of synthetic fertilizer) and the total cost of field application of composted OAs was 0.8 for both CCM+$N_{Rd}$ and CFM+$N_{Rd}$, whereas those for stockpiled OAs were 7.9 and 1.3 for CM+$N_{Rd}$ and FM+$N_{Rd}$, respectively. Compost purchase and transport amounted to 80% of the total cost of CCM+$N_{Rd}$ and CFM+$N_{Rd}$, representing a significant economic barrier to the use of composted manure by farmers. For this study, the rates at which OAs were applied were selected focusing on replacing PAN. Given the low N availability and low N/P ratio of OAs, calculating the OA application rates focusing on replacing PAN could lead to relatively high OA application rates, resulting in potentially excessive P application and high costs when using composted OA. An alternative way to reduce the economic burden associated with the use of composted OAs could be to reduce the OA application rate with a marginal increase of supplemental N fertilizer. Also, the high application rates of AOAs led to a high cost associated with the transport that was up to US$475 ha$^{-1}$ for treatments that received FM. A possible way to further reduce the cost associated with the use of OAs could be to incentivize the cultivation of high-value crops with high plant nutrients requirement close to large manure/compost producers.

This economic assessment did not account for the additional environmental and economic benefits associated with composted OAs. The value of OAs may extend beyond nutrient replacement value and includes enhancement of agronomic and biological soil properties. These benefits are difficult to quantify economically but include improvements in chemical, physical, and biological soil quality (D’Hose et al., 2016; Diacono & Montemurro, 2010). For instance, D’Hose et al. (2016) showed that the repeated application of composted material increased disease suppressiveness against Botrytis cinerea on lettuce. This effect could have been economically quantified by considering the savings associated with the non- or reduced application of disease control chemicals or by considering the reduced losses, but this would have been an extremely case-specific parameter and therefore was not included in the economic calculation. For this study, the quantification of the crop nutrient supply with the OA application, however, was only considered for a relatively short time frame (1 yr). Because only about 25–35% of the organic N added to soil is released in the first year after application (De Rosa et al., 2017), it could be argued that if the long-term nutrient release is considered in the economic budget, the additional cost calculated for the composted materials as well as stockpiled would be reduced. Indeed, considering the residual effect of the first-year OA application on P and K availability (70% of P and 100% K added with OA application could be available for crops; Eghball et al. [2002]), in the second-year crop rotation it will be possible to further save up to ~US$800 ha$^{-1}$. Therefore, to estimate the real value of soil OAs, nutrient replacement value should be considered over longer time frames rather than a single crop cycle application. On the other hand, the continuous application of CM would require the supplemental addition of P and K fertilizers to match the crop needs and hence increase the environmental and economic costs associated with the application of CM.

In this case study, we analyzed the economic budgets of different OA supply chains for the production of high-value horticultural crops characterized by large plant nutrient requirements. For grain cropping systems where the plant nutrient requirements and marginal net return are lower than horticultural crops, it could be argued that to obtain a positive economic budget it will be required to reduce the OA application rates in favor of synthetic fertilizers. This could be achieved by calculating the OA application rate by matching the crop P requirements rather than N because N-based OA application rates could overapply P relative to crop demand (Spiegel et al., 2020).
| Treatments | CONV | CM+N<sub>conv</sub> | CCM+N<sub>conv</sub> | CM+N<sub>Rd</sub> | CCM+N<sub>Rd</sub> | FM+N<sub>conv</sub> | CFM+N<sub>conv</sub> | FM+N<sub>Rd</sub> | CFM+N<sub>Rd</sub> |
|------------|------|---------------------|---------------------|-----------------|-----------------|-------------------|-------------------|-----------------|-------------------|
| Fertilizer |      |                     |                     |                 |                 |                   |                   |                 |                   |
| N          | 444.18 | 444.18              | 444.18              | 361.07          | 358.21          | 444.18            | 444.18            | 169.07          | 338.15            |
| P          | 403.30 |                     |                     |                 |                 |                   |                   |                 |                   |
| K          | 434.70 |                     |                     |                 |                 |                   |                   |                 |                   |
| Transport and spreading | 41.05 | 27.90               | 27.90               | 26.48           | 26.43           | 27.90             | 27.90             | 23.18           | 26.08             |
| Organic amendments |      |                     |                     |                 |                 |                   |                   |                 |                   |
| Purchase   | 64.84  | 1,037.40            | 64.84               | 1,037.40        | 373.46          | 965.25            | 373.46            | 965.25          | 1,192.06          |
| Transport and spreading | 70.79 | 294.45              | 70.79               | 294.45          | 626.34          | 381.03            | 626.34            | 381.03          | 1,710.51          |
| Total cost | 1,323.22 | 607.70            | 1,803.93            | 523.17          | 1,716.48        | 1,471.88          | 1,818.36          | 1,192.06        | 1,710.51          |
| Difference from CONV, % | −54.07 | 36.33               | −60.46              | 29.72           | 11.23           | 37.42             | −9.91             | 29.27           |                   |
| GHG budgets, kg CO<sub>2</sub>-eq ha<sup>−1</sup> |      |                     |                     |                 |                 |                   |                   |                 |                   |
| Fertilizer |      |                     |                     |                 |                 |                   |                   |                 |                   |
| N          | 1,199.7 | 1,199.7             | 1,199.7             | 975.24          | 967.5           | 1,199.7           | 1,199.7           | 456.66          | 913.32            |
| P          | 47.71  |                     |                     |                 |                 |                   |                   |                 |                   |
| K          | 93.05  |                     |                     |                 |                 |                   |                   |                 |                   |
| Transport and spreading | 11 | 7                    | 7                    | 6               | 6               | 7                 | 7                 | 5               | 6                 |
| Organic amendments |      |                     |                     |                 |                 |                   |                   |                 |                   |
| Processing (i.e., stockpiling or composting) | 10.37 | 133.17              | 10.37               | 133.17          | 2,072.18        | 88.77             | 2,072.18          | 88.77            |                   |
| Transport and spreading | 23.48 | 127.48              | 23.48               | 127.48          | 281.79          | 167.73            | 281.79            | 167.73          |                   |
| Field emissions | 546.65 | 708.10              | 526.96              | 815.53          | 447.95          | 1,468.50          | 874.90            | 928.16          | 777.41            |
| Total GHG, t CO<sub>2</sub> ha<sup>−1</sup> | 1.89 | 1.95                 | 1.99                | 1.83            | 1.68            | 5.03              | 2.34              | 3.74            | 1.95              |
| Difference from CONV, % | 2.97 | 5.29                | −3.17               | −11.11          | 166.14          | 23.81             | 97.88             | 3.17            |                   |

Environmental credits scheme—avoided fertilizer manufacturing GHG derived from the reduction of fertilizers use

| Avoided, t CO<sub>2</sub>-eq ha<sup>−1</sup> | 0.13 | 0.13 | 0.36 | 0.37 | 0.13 | 0.13 | 0.88 | 0.42 |
| Net balance, t CO<sub>2</sub>-eq ha<sup>−1</sup> | 1.89 | 1.82 | 1.86 | 1.47 | 1.31 | 4.90 | 2.21 | 2.86 | 1.53 |
| Difference from CONV, % | −3.70 | −1.59 | −22.22 | −30.69 | 159.26 | 16.93 | 51.32 | −19.05 |

Note. CCM, composted chicken manure; CFM, composted feedlot manure; CM, stockpiled chicken manure; CONV, conventional N, P, and K application rate; FM, stockpiled feedlot manure; GHG, greenhouse gas; +N<sub>conv</sub>, conventional N fertilizer rate; +N<sub>Rd</sub>, reduced N fertilizer rate.

### 3.2 GHG budget

The GHG budget highlighted that the treatments that received FM emitted the highest CO<sub>2</sub>-eq among all treatments. The FM+ N<sub>conv</sub> and FM+ N<sub>Rd</sub> emitted 5.03 and 3.74 t CO<sub>2</sub>-eq ha<sup>−1</sup> (Table 4). On average, the total GHG emissions from the FM treatments were higher than those from CFM treatments by 90 and 115% for +N<sub>Rd</sub> and +N<sub>conv</sub>, respectively. Combining CFM with reduced N fertilizer rate resulted in a similar emission level to that of CONV (1.89 t CO<sub>2</sub>-eq ha<sup>−1</sup>). Both CM and CCM in combination with the reduced fertilizer rate decreased emissions up to 11% compared with CONV, corresponding to 1.68 and 1.83 t CO<sub>2</sub>-eq ha<sup>−1</sup> for CCM+N<sub>Rd</sub> and CM+N<sub>Rd</sub>, respectively (Table 4; Figure 1).

The emissions generated from the transport and field application of OAs accounted for only a small proportion (1.2 and 8% for CM and FM, respectively) of the total CO<sub>2</sub>-eq of the OA supply chain. These emissions ranged from 23.5 to 281.8 kg CO<sub>2</sub>-eq ha<sup>−1</sup> (Table 4) for treatments that received CM and FM, respectively. However, these values only account for tailpipe emissions (over the 110-km travel distance) and do not account for the possible direct emissions from OAs themselves during transportation. It can be hypothesized that transporting OAs over longer distances will increase not only the vehicular emissions but also the exposure time of OAs, thereby increasing the direct emissions from OAs. Given the lower stability of stockpiled OAs than composted OAs, it is expected that over a long haul, direct
emissions from stockpiled OAs will be higher than composted OAs.

Manure processing (i.e., stockpiling and composting) was responsible for 55.2% of the total direct emissions from FM+N_Rd and only 4.5% for CFM+N_Rd, whereas it accounted for 7.9 and 0.6% of the total emissions for CCM+N_Rd and CM+N_Rd, respectively. The lower field emissions of treatments that received CCM in comparison to CM at the same rate of N fertilizers compensated the higher processing emissions of CCM (7.1 kg CO_2-eq t^-1 fw OA) than CM (2.9 kg CO_2-eq t^-1 fw OA) (Table 3). However, although the composting process lowered N_2O and CH_4 emissions (Table 3) due to the increased aeration as a consequence of turning operations, the composting process could potentially increase other N losses, such as via ammonia volatilization (Amon et al., 2001), reducing the environmental benefit observed from the field application.

In the field experiment, the application of composted manure in combination with reduced fertilizer rate did not increase N_2O emissions (CCM+N_Rd 967 ± 309 g N_2O-N ha^-1 yr^-1 and CM+N_Rd 1,670 ± 309 g N_2O-N ha^-1 yr^-1) in comparison to CONV (1,179 ± 213 g N_2O-N ha^-1 yr^-1) (Table 3), whereas stockpiled feedlot manure with a standard fertilizer rate (FM+N_CONV) resulted in the highest N_2O emissions (3,142 ± 542 g N_2O-N ha^-1 yr^-1) (Table 3). Reducing N fertilizer rate in the FM treatment (FM+N_Rd) lowered N_2O emissions by 37% (1,984 ± 377 g N_2O-N ha^-1 yr^-1) as compared to FM+N_CONV. The lowest N_2O emissions among the fertilized treatments were recorded in the CCM+N_Rd (967 ± 309 g N_2O-N ha^-1 yr^-1) and were not significantly different from the emissions measured from the unfertilized treatment (0N, 862 ± 10 g N_2O-N ha^-1 yr^-1) (Table 3). Generally, the application of composted OAs in combination with reduced fertilizer rate lowered soil N_2O emissions in comparison to stockpiled OAs under the same fertilizer strategy. This N_2O emission reduction can be attributed to N and C limitation for soil microorganisms. Composted OAs, when compared to raw OAs, generally contain less easily degradable organic matter and a higher percentage of recalcitrant material (high stability) that favors N immobilization due to the lower C/N ratio (Bernai et al., 1998). Easily degradable organic matter also serves as an O_2 sink as well as a C source for heterotrophic denitrification.

The calculated EFs following the IPCC methodology that took into account the yearly total N applied with OAs plus N-fertilizer ranged from 0.02% for CCM+N_Rd to 0.25% for CM+N_Rd (Table 3) and were lower than the EP proposed by the current IPCC methodology, which considers 1% for total N applied lost as N_2O. Furthermore, considering only the estimated PAN released from OAs plus the total N-fertilizer applied, the EF_OA values were still lower than 1% and ranged from 0.04% for CCM+N_Rd to 0.46% for FM+N_CONV (Table 3). However, other studies using OAs in combination with synthetic fertilizer report EFs substantially higher (1.7–2.9%) than the IPCC default of 1% (Charles et al., 2017; Liyanage et al., 2020). These results highlight the uncertainty associated with the use of the standard EF when estimating N_2O losses following the application of OAs.

The field-measured CH_4 fluxes ranged from −195.99 ± 246.55 g CH_4-C ha^-1 yr^-1 for CONV to −87.42 ± 107.2 g CH_4-C ha^-1 yr^-1 for CCM+N_CONV and the application of OAs did not increase annual soil CH_4 (Table 4). The calculation of the cumulative fluxes highlighted a net uptake of CH_4 among all treatments (Table 3) most likely driven by the nonanoxic conditions of the soil at the experimental site because aerated soils are generally a net sink for CH_4 (van Delden et al., 2018).

Another positive effect observed with the application of OAs is the increase of SOC content after only a single annual application, though the changes in SOC were significantly higher than CONV only for treatments that received FM (Table 3). The highest SOC of 30.9 ± 1.9 t C ha^-1 (0–0.3 m) was observed in FM+N_CONV, followed by treatments that received composted OAs (CMF and CCM) (Table 3). The lowest levels of SOC content were observed in CM+N_CONV and CONV, with 26.5 ± 1.30 and 25.9 ± 1.12 t C ha^-1 (Table 3). With the annual application of OAs, the FM treatments received a total of 7 t C ha^-1, whereas for CCM and CM an average of 2.9 t C ha^-1 was applied (Table 1). Because the net difference in soil C content at the end of the annual crop rotation between CONV and FM treatments was only from 4 to 5 t C ha^-1 and for CCM and CFM the average differences were 2.8 and 2.6 t C ha^-1, respectively, it can be argued that the majority of C applied with the composted material was retained in the soil. This can be attributed to the higher stability of the composted products in comparison to raw OAs. Therefore, the field application of composted OAs might increase SOC stocks while reducing reactive N pollution, but the latter is only achievable if the plant nutrients released from OAs are accounted for in the crop-rotation fertilization strategy.
expensive than those of stockpiled manures, but if long-term nutrient release from OAs is taken into account, especially K and P considering their relatively low mobility in the soil, additional savings are possible. If only considering the nutrient replacement value of OAs, farmers will only purchase OAs in substitution of synthetic fertilizer if the price of OAs is lower or equal to the equivalent nutrient value of synthetic fertilizer or if the environmental benefits provided with the OA application exceed the extra cost associated with the use of OAs (Leip et al., 2019). Because income from soil carbon sequestration from OAs alone is unlikely to bridge this financial gap, particularly in the short-term, this study proposes that future policy should develop methodologies to include avoided GHG emissions from OA application. Our case study showed that the avoided emissions due to synthetic fertilizer reduction (calculated as the difference of total GHG generated from N, P, and K fertilizer manufacturing between CONV and OA treatments) were 0.37 and 0.42 t CO₂-eq ha⁻¹ for CCM+Ν₉Rd and CFM+Ν₉Rd, respectively (Table 4). Considering these avoided emissions in the GHG budgeting indicated that our Ν₉Rd treatments resulted in a 30.4% reduction in GHG emissions at a similar crop yield level when compared to the control treatment (CONV). Considering an average price for a Carbon Credit Unit of US$11.2 t⁻¹ CO₂-e (Commonwealth of Australia, 2021), the additional economic return for avoided emissions for CCM+Ν₉Rd and CFM+Ν₉Rd would be US$4.07 ha⁻¹ and US$4.62 ha⁻¹, respectively.

Our case study demonstrated that the N₂O EFs of synthetic N fertilizer and OA supplementation could be well below 1%, which is the default EF by the IPCC (Kroeze et al., 1997). The EFs of OA and synthetic N fertilizers were 0.23 and 0.14, averaged across all treatments, respectively (Table 3). This high discrepancy observed between default EFs and actual measured field and manure processing emissions highlights the importance of using actual measured GHG emissions to accurately estimate the environmental impact of the manure supply chain. Therefore, to incentivize the use of OAs as an environmentally and economically sustainable substitute for synthetic fertilizers, the reduction in GHG emissions associated with the use of OAs should be taken into account into actual C sequestration programs with a refined N₂O emission factor used for GHG emission accounting.

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AUTHOR CONTRIBUTIONS
Daniele De Rosa: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Project administration; Resources; Software; Supervision; Validation; Visualization; Writing-original draft; Writing-review & editing. Johannes Biala: Conceptualization; Data curation; Formal analysis; Funding acquisition; Investigation; Methodology; Project administration; Resources; Supervision; Validation; Writing-original draft; Writing-review & editing. Trung H. Nguyen: Data curation; Investigation; Methodology; Software; Writing-original draft; Writing-review & editing. Elaine Mitchell: Conceptualization; Investigation; Methodology; Writing-review & editing. Johannes Friedl: Conceptualization; Investigation; Methodology; Writing-review & editing. Clemens Scheer: Data curation; Funding acquisition; Investigation; Methodology; Project administration; Resources; Validation; Writing-review & editing. Peter R. Grace: Conceptualization; Funding acquisition; Methodology; Project administration; Resources; Validation; Writing-review & editing. David W. Rowlings: Conceptualization; Data curation; Formal analysis; Funding acquisition; Investigation; Methodology; Project administration; Resources; Writing-review & editing.

CONFLICT OF INTEREST
The authors declare no conflict of interest.

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REFERENCES
Argonne National Laboratory. (2019). GREET® model: The greenhouse gases, regulated emissions, and energy use in transportation model. https://greet.es.anl.gov/greet.models
Amon, B., Amon, T., Boxberger, J., & Alt, C. (2001). Emissions of NH₃, N₂O and CH₄ from dairy cows housed in a farmyard manure tying stall (housing, manure storage, manure spreading). Nutrient Cycling in Agroecosystems, 60, 103–111. https://doi.org/10.1023/A:1012649028772
Andersen, J. K., Boldrin, A., Christensen, T. H., & Scheutz, C. (2010). Greenhouse gas emissions from home composting of organic household waste. Waste Management, 30, 2475–2482. https://doi.org/10.1016/j.wasman.2010.07.004
Ba, S., Qu, Q., Zhang, K., & Groot, J. C. J. (2020). Meta-analysis of greenhouse gas and ammonia emissions from dairy manure composting. Biosystems Engineering, 193, 126–137. https://doi.org/10.1016/jbiosystemseng.2020.02.015
Benjamini, Y., & Hochberg, Y. (1995). Controlling the false discovery rate: A practical and powerful approach to multiple testing. Journal of the Royal Statistical Society: Series B (Methodological), 57, 289–300. https://doi.org/10.2307/2346101
Bernaí, M., Paredes, C., Sanchez-Monedero, M., & Cegarra, J. (1998). Maturity and stability parameters of composts prepared with a wide range of organic wastes. *Bioresource Technology*, 63, 91–99.

Biala, J., Lovrick, N., Rowlings, D., & Grace, P. (2016). Greenhouse-gas emissions from stockpiled and composted dairy-manure residues and consideration of associated emission factors. *Animal Production Science*, 56, 1432–1441. https://doi.org/10.1071/AN16009

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*, 236, 88–98. https://doi.org/10.1016/j.agee.2016.11.021

Christensen, T. H., Gentil, E., Boldrin, A., Larsen, A. W., Weidema, B. P., & Hauschild, M. (2009). C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. *Waste Management & Research*, 27, 707–711. https://doi.org/10.1177/0734242x08096304

Commonwealth of Australia. (2021). *Clean energy regulator*. http://www.cleanenergyregulator.gov.au

D’Hose, T., Ruyschstraet, G., Vlaene, N., Debode, J., Vanden Nest, T., Van Vaerenbergh, J., Cornelis, W., Willekens, K., & Vandecasteele, B. (2016). Farm compost amendment and non-inversion tillage improve soil quality without increasing the risk for N and P leaching. *Agriculture, Ecosystems & Environment*, 225, 126–139. https://doi.org/10.1016/j.agee.2016.03.035

De Rosa, D., Basso, B., Rowlings, D. W., Scheer, C., Biala, J., & Grace, P. R. (2017). Can organic amendments support sustainable vegetable production? *Agronomy Journal*, 109, 1856–1869. https://doi.org/10.2134/agronj2016.12.0739

De Rosa, D., Rowlings, D. W., Biala, J., Scheer, C., Basso, B., & Grace, P. R. (2018). N₂O and CO₂ emissions following repeated application of organic and mineral N fertiliser from a vegetable crop rotation. *Science of the Total Environment*, 637–638, 813–824. https://doi.org/10.1016/j.scitotenv.2018.05.046

De Rosa, D., Rowlings, D. W., Biala, J., Scheer, C., Basso, B., McGree, J., & Grace, P. R. (2016). Effect of organic and mineral N fertilizers on N₂O emissions from an intensive vegetable rotation. *Biology and Fertility of Soils*, 52, 895–908. https://doi.org/10.1007/s00374-016-1117-5

Diacono, M., & Montemurro, F. (2010). Long-term effects of organic amendments on soil fertility. A review. *Agronomy for Sustainable Development*, 30, 401–422. https://doi.org/10.1051/agro/2009040

Dorich, C. D., De Rosa, D., Barton, L., Grace, P., Rowlings, D., Migliorati, M. D. A., Wagner-Riddle, C., Key, C., Wang, D., Fehr, B., & Conant, R. T. (2020). Global Research Alliance N₂O methodology guidelines: Guidelines for gap-filling missing measurements. *Journal of Environmental Quality*, 49, 1186–1202. https://doi.org/10.1002/jeq2.20138

Eghball, B. (2000). Nitrogen mineralization from field-applied beef cattle feedlot manure or compost. *Soil Science Society of America Journal*, 64, 2024–2030. https://doi.org/10.2136/sssaj2000.6462024x

Eghball, B., Wienenhold, B. J., Gilley, J. E., & Eigenberg, R. A. (2002). Mineralization of manure nutrients. *Journal of Soil and Water Conservation*, 57, 470–473. http://www.jswconline.org/content/57/6/470. abstract

Fillingham, M. A., VanderZaag, A. C., Burtt, S., Baldé, H., Ngwabie, N. M., Smith, W., Hakami, A., Wagner-Riddle, C., Bittman, S., & MacDonald, D. (2017). Greenhouse gas and ammonia emissions from production of compost bedding on a dairy farm. *Waste Management*, 70, 45–52. https://doi.org/10.1016/j.wasman.2017.09.013

Friedl, J., Scheer, C., Rowlings, D. W., Mumford, M. T., & Grace, P. R. (2017). The nitrification inhibitor DMPP (3,4-dimethylpyrazole phosphate) reduces N₂ emissions from intensively managed pastures in subtropical Australia. *Soil Biology and Biochemistry*, 108, 55–64. https://doi.org/10.1016/j.soilbio.2017.01.016

Hartz, T. K., Mitchell, J. P., & Giannini, C. (2000). Nitrogen and carbon mineralization dynamics of manures and composts. *HortScience*, 35, 209–212. http://hortsci.ashspublications.org/content/35/2/209. abstract

IPCC. (2019). Summary for policymakers. In P. R. Shukla, J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H.-O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E. Huntley, K. Kissick, M. Belkacemi, & J. Malley (Eds.). *Climate change and land: An IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems*. IPCC. https://www.ipcc.ch/srccl/chapter/summary-for-policymakers/

Kroeze, C., Mosier, A., Nevison, C., Oenema, O., Seitzinger, S., van Cleemput, O., Conrad, R., Mitra, A., HU, N., & Sass, R. (1997). Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC/OECD/IEA. http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.html

Leip, A., Ledgard, S., Uwizey, A., Palhares, J. C. P., Aller, M. F., Amom, B., Binder, M., Cordovil, C. M.dS., De Camillis, C., Dong, H., Fusit, A., Helin, J., Hörtenhuber, S., Hristov, A. N., Koelsch, R. Liu, C., Masso, C., Nkongolo, N. V., Patra, A. K., … Wang, Y. (2019). The value of manure: Manure as co-product in life cycle assessment. *Journal of Environmental Management*, 241, 293–304. https://doi.org/10.1016/j.jenvman.2019.03.059

Liyanage, A., Grace, P. R., Scheer, C., de Rosa, D., Ranwala, S., & Rowlings, D. W. (2020). Carbon limits non-linear response of nitrous oxide (N₂O) to increasing N inputs in a highly-weathered tropical soil in Sri Lanka. *Agriculture, Ecosystems & Environment*, 292, 106808. https://doi.org/10.1016/j.agee.2019.106808

Menz, H., Oenema, O., Burton, C., Shipin, O., & Gerber, P. J., (2010). Impacts of intensive livestock production and manure management on the environment. In H. Steinfeld, H. A. Mooney, F. Schneider, & L. E. Neville (Eds.), *Livestock in a changing landscape* (Vol. 1, pp. 139–164). Island Press.

Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C., & Gerber, P. (2017). Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. *Global Food Security*, 14, 1–8. https://doi.org/10.1016/j.gfs.2017.01.001

Porter, I., Riches, D., & Scheer, C. (2017). Benchmarking and mitigation of nitrous oxide emissions from manures and fertilisers used in temperate vegetable crops in Australia. *Soil Research*, 55, 534–546. https://doi.org/10.1071/SR17043

Powers, S. M., Chowdhury, R. B., MacDonald, G. K., Metson, G. S., Beusen, A. H. W., Bouwman, A. F., Hampton, S. E., Mayer, B. K., McCrackin, M. L., & Vaccari, D. A. (2019). Global opportunities to increase agricultural independence through phosphorus recycling. *Earth’s Future*, 7, 370–383. https://doi.org/10.1029/2018EF001097

Quilty, J. R., & Cattle, S. R. (2011). Use and understanding of organic amendments in Australian agriculture: A review. *Soil Research*, 49, 1–26. https://doi.org/10.1071/SR10059
Rezaei Rashti, M., Wang, W., Moody, P., Chen, C., & Ghadiri, H. (2015). Fertiliser-induced nitrous oxide emissions from vegetable production in the world and the regulating factors: A review. Atmospheric Environment, 112, 225–233. https://doi.org/10.1016/j.atmosenv.2015.04.036

Scheer, C., Rowlings, D., Firrell, M., Deuter, P., Morris, S., Riches, D., Porter, I., & Grace, P. (2017). Nitrification inhibitors can increase post-harvest nitrous oxide emissions in an intensive vegetable production system. Scientific Reports, 7, 43677. https://doi.org/10.1038/srep43677

Scheer, C., Rowlings, D. W., Firrel, M., Deuter, P., Morris, S., & Grace, P. R. (2014). Impact of nitrification inhibitor (DMPP) on soil nitrous oxide emissions from an intensive broccoli production system in sub-tropical Australia. Soil Biology and Biochemistry, 77, 243–251. https://doi.org/10.1016/j.soilbio.2014.07.006

Spiegal, S., Kleinman, P. J. A., Endale, D. M., Bryant, R. B., Dell, C., Goslee, S., Meinen, R. J., Flynn, K. C., Baker, J. M., Browning, D. M., McCarty, G., Bittman, S., Carter, J., Cavigelli, M., Duncan, E., Gowda, P., Li, X., Ponce-Campos, G. E., Cibin, R., … Yang Q. (2020). Manuresheds: Advancing nutrient recycling in US agriculture. Agricultural Systems, 182, 102813. https://doi.org/10.1016/j.agsy.2020.102813

van Delden, L., Rowlings, D. W., Scheer, C., De Rosa, D., & Grace, P. R. (2018). Effect of urbanization on soil methane and nitrous oxide fluxes in subtropical Australia. Global Change Biology, 24, 5695–5707. https://doi.org/10.1111/gcb.14444

VanderZaag, A. C., Jayasundara, S., & Wagner-Riddle, C., (2011). Strategies to mitigate nitrous oxide emissions from land applied manure. Animal Feed Science and Technology, 166–167, 464–479. https://doi.org/10.1016/j.anifeedsci.2011.04.034

Wang, M., Elgowainy, A., Lee, U., Bafana, A., Benavides, P. T., Burnham, A., Cai, H., Dai, Q., Gracida-Alvarez, U. R., Hawkins, T. R., Jaquez, P. V., Kelly, J. C., Kwong, H., Lu, Z., Liu, X., Ou, L., Sun, P., Winjobi, O., Xu, H., … Zang, G. (2020). Summary of expansions and updates in GREET® 2020 (No. ANL/ESD-209). Argonne National Lab. (ANL). https://doi.org/10.2172/1671788

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