Greenhouse Gas Emissions from Swine Effluent Applied to Soil by Different Methods

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Greenhouse gas (CO₂, CH₄, and N₂O) emissions were measured in a field experiment evaluating preplant swine effluent application methods for no-till corn (Zea mays L.) grain production. The treatments included a control, an inorganic fertilizer treatment receiving 179 kg N ha⁻¹ as urea–NH₄NO₃ (UAN), and three effluent application methods that received a target rate of 200 kg N ha⁻¹. The effluent treatments included surface application, direct injection, and application in combination with soil aeration. Gas emission measurements were initiated after application and collected throughout the 2007 and 2008 growing seasons using a vented chamber technique. There were no significant differences in CO₂ losses, which averaged 738 and 718 g CO₂ m⁻² in 2007 and 2008, respectively. Placement of effluent below the soil surface by injection or aeration resulted in elevated CH₄ emissions compared with the control. Injection emitted significantly more CH₄ than surface applications, with emissions of 0.26 and 0.80 g CH₄ m⁻² from the injection treatments in 2007 and 2008, respectively. In 2007, N₂O emissions were similar for the UAN, surface effluent, and aeration effluent treatments, emitting an average of 0.72 g N₂O m⁻². In contrast, the injection treatment emitted 0.47 g N₂O m⁻². In 2008, this trend was reversed, with the injection treatment emitting 0.82 g N₂O m⁻² and the remaining N source treatments emitting an average of 0.36 g N₂O m⁻². These differences between years probably resulted from differences in rainfall distribution. These results demonstrate that climatic conditions and application method need consideration when evaluating the impact of liquid manure management on greenhouse gas emissions.

Abbreviations: GHG, greenhouse gas; UAN, urea–ammonium nitrate.

Because of continuing concerns associated with the impact of anthropogenic greenhouse gas (GHG) production on global climate change, it is important to assess the magnitude of greenhouse gas emissions from various human activities. Manure management has been identified as a significant contributor to atmospheric GHG concentrations (USEPA, 2009). The USEPA (2009) has estimated that manure management is responsible for 24% of the CH₄ and 5% of the N₂O emitted by U.S. agricultural activities. Variations in GHG emissions resulting from different land application methods are not sufficiently understood, however, to include in current estimates of GHG emissions from manure management.

In general, efficient utilization of manure N requires that the manure be placed below the soil surface through direct injection or incorporation, through tillage, after application; however, no-till production systems require that manure not be incorporated using conventional tillage practices. Therefore injection of manures such as swine (Sus scrofa) effluent has been recommended because it limits the loss of N through NH₃ volatilization. In fact, Misselbrook et al. (1996) found that when cattle manure slurry was injected into grassland soils to a depth of 6 cm, NH₃ losses were reduced by 40 to 79% compared with surface applications. Mattila and Joki-Tokola (2003) found that injection of swine effluent completely eliminated NH₃ volatilization from grasslands. Liquid manure injection is also efficient at reducing runoff transport of nutrients; for example, Daverede et al. (2004) found
that direct injection of liquid swine effluent reduced runoff P losses by 94 to 99% compared with surface applications.

These environmental benefits are generally well understood; however, the impact of liquid manure application techniques on GHG emissions is very much uncertain. A laboratory incubation conducted by Flessa and Beese (2000) found that simulated injection of cattle slurry resulted in N2O emissions equal to 3.3% of the slurry N added. Surface application emitted only 0.2% of the added N as N2O. These researchers also found that 2 and 39 g CH4-C ha−1 was emitted from surface and injection treatments. An additional laboratory experiment conducted by Velthof et al. (2003) found that band placement of swine effluent at 5 cm below the soil surface maximized N2O emissions compared with surface applications or homogeneous mixing of the effluent with the soil. Field research conducted by Perala et al. (2006) also found that injection increased N2O emission compared with surface application followed by incorporation; however, the results from this field experiment showed that injection treatments served as net sinks for CH4. A laboratory experiment conducted by Dendooven et al. (1998) concluded that injection of pig slurries did not significantly impact N2O emissions. In contrast, Wulf et al. (2002) found that injection into both arable land and grassland maximized CH4 and N2O emissions compared with various surface applications and tillage incorporation. Few field studies have been conducted to compare the impact of application method on GHG emissions from liquid manures. Of these, no studies have been conducted to evaluate application methods in soil under long-term continuous no-till crop production.

There are numerous chemical, physical, and biological differences between cultivated soils and no-till soils that may influence GHG emissions from different swine effluent application techniques. The surface of no-till soils generally have higher levels of organic C (Ding et al., 2002), lower temperatures, increased moisture, and altered microbial communities (Doran, 1980) than tilled soils. In addition, incorporation of swine effluent into no-till soils is limited to methods that minimize surface disturbance, such as direct injection or aeration.

In fact, a new commercial application technology has recently been developed where liquid manure can be applied in combination with soil aeration. This allows the effluent to rapidly move below the soil surface and potentially decreases NH3 volatilization, thereby improving N utilization compared with surface applications without incorporation. This aeration application technique was compared with injection and surface application methods in grasslands by Chen et al. (2001). Their study found that aeration application of swine effluent resulted in NH3 concentrations at the soil surface that were intermediately between injection and surface application. Grass forage yields were also comparable among the three application types.

There simply are no data available to evaluate the impact of swine effluent application techniques on GHG emissions from no-till cropland. Therefore, the objective of this field experiment was to compare GHG emissions from swine effluent that was injected, surface applied, or applied in combination with aeration in a no-till corn grain production system. The injection and aeration methods utilized in this field experiment were chosen because they minimize surface soil disturbance and NH3 volatilization compared with surface applications. They are also a commercially available option for no-till crop producers.

**MATERIALS AND METHODS**

A field experiment was conducted in 2007 in which field corn was grown for grain. The site had been cropped in a no-till corn–soybean (Glycine max [L.] Merr.) rotation for approximately 10 yr on a Nicholson silt loam (a fine-silty, mixed, active, mesic Oxyaquic Fragiudalf). Greenhouse gas emissions were measured from five treatments. These treatments included a control, which received no preplant fertilizer, and an inorganic fertilizer treatment that received 179 kg N ha−1 as UAN liquid fertilizer (28–0–0) that was broadcast applied. Three swine effluent application treatments were also included. These swine effluent treatments were applied at a target rate of 179 kg available N ha−1; however, there was a difference between the target rate and actual rate, which is due to the total N concentration of the effluent and the available N (85–90% of the total N) in the effluent. These swine effluent applications were made using the following methods: (i) a surface application, (ii) an injection application, and (iii) an aeration application. These preplant treatments were applied on 27 Apr. 2007 and 19 Apr. 2008. The resulting total nutrient application rates are presented in Table 1. Corn (Wfells Hybrids, W9602) was planted after treatment application, with 76 cm between each corn row at a seeding rate of 69,000 kernels ha−1.

The aeration implement (Aerway, SSD Manure Management System, Wylie, TX) creates holes that are 7.5 cm long and 1.5 cm wide at the soil surface and tapered to a point at a depth of 20 cm. The aeration holes are placed every 15 cm from center to center in the direction of travel and 19 cm from center to center across the width of the implement. Effluent is delivered to each row of aeration holes with hoses that place the effluent into the aeration holes. The injection implement (Dietrich slurry injection system, DSI Inc., Goodfield, IL) places effluent below the soil surface behind injection shanks that are placed at a depth of 20 cm. The injection shanks are 102 cm apart across the width of the implement and injection rows were parallel with corn rows planted after effluent application. These two effluent application implements were fitted to a Balzer 2250 Magnum vacuum tank (Balzer Inc., Mountain Lake, MN) that was pulled by a John Deere 8130 tractor (Deere and Co., Moline, IL). The surface application treatment was applied using the aeration implement without the aerating mechanism placed in the ground. This method of surface application was chosen because it resulted in a uniform application.

The experiment was arranged in a randomized complete block with three replicates. Plots were placed...
in the field so that each would contain 16 rows of corn measuring 46 m long. Before treatment applications, composite soil samples (0–15 cm) were collected from the experimental area to assess the initial soil nutrient concentrations (Table 2).

Soil samples collected before treatment application were dried at room temperature and ground to pass a 2.0-mm sieve. These samples were analyzed for NO$_3$–N and NH$_4$–N after extraction with 2 mol L$^{-1}$ KCl (1:10 soil/KCl extraction ratio) using flow injection analysis (QuickChem FIA+, Lachat Instruments, Milwaukee, WI). Soil pH was measured on a 1:1 soil/0.05 mol L$^{-1}$ CaCl$_2$ solution using a combination electrode (Accuphast electrode, Fisher Scientific, Pittsburgh, PA). Soil plant nutrient availability was also assessed using inductively coupled plasma–optical emissions spectroscopy (ICP–OES) (Varian Vista Pro; Varian Analytical Instruments, Walnut Creek, CA) analysis of Mehlich 3 soil extracts. Total N and C in the soil were measured using a Vario Max CN analyzer (Elementar Americas, Mt. Laurel, NJ).

The total N and C contents of the effluent was also measured using the Vario Max CN analyzer. The remaining elements were measured using ICP–OES after microwave digestion with HNO$_3$ and HCl.

The fluxes of CO$_2$, CH$_4$, and N$_2$O were measured using vented chambers (Mosier and Mack, 1980; Hutchinson and Mosier, 1981; Mosier et al., 1991). The chambers used were made of Al and measured 10 cm tall. At each flux measurement time, the chambers were placed in a water channel on fixed anchors (38 cm wide and 102 cm long). After treatment applications, one anchor was forced into the ground to a depth of 15 cm in each plot such that they were flush with the soil surface. The anchors were placed such that the 102-cm length was perpendicular to the corn rows and injection rows. At planting, the anchors were removed and then reestablished in a new location in each plot. When corn plants emerged inside the measurement area, the anchors were removed. At each flux sampling period, a small 12-V fan powered by a 9-V battery was placed on a platform inside the chamber to mix the headspace air. Gas samples (20 mL) from inside the chambers were collected by syringe at 0, 15, and 30 min after placement on the anchors. The gas samples were injected into 20-mL evacuated vials that were sealed with gray butyl rubber septa. The samples were analyzed with a gas chromatograph (CP-3800, Varion, Palo Alto, CA) equipped with a thermocconductivity detector, a flame ionization detector, and an electron capture detector for quantification of CO$_2$, CH$_4$, and N$_2$O, respectively (Mosier and Mack, 1980; Mosier et al., 1991). These analyses were achieved using a two-point calibration curve for each gas, with quality control standards analyzed every 25 samples during the analysis of unknowns.

Linear regression analysis of the relationship between headspace gas concentration and time was used to calculate gas fluxes. The cumulative gas emissions were estimated through linear extrapolation between sampling periods.

Meteorological data including rainfall, air temperature, wind speed, and relative humidity were collected throughout the growing season. In addition, soil temperature and volumetric soil moisture content were measured in each treatment at each gas flux measurement period.

Analysis of variance was performed using the SAS PROC GLM procedure (SAS Institute, 2001) to determine treatment effects on cumulative GHG emissions. Fisher’s protected LSD was used to separate treatment means.

**RESULTS AND DISCUSSION**

During the 2007 growing season, CO$_2$ fluxes were initially elevated and then declined as soil moisture declined (Fig. 1 and 2). On 7 July, following a period of rain, CO$_2$ fluxes increased to approximately 450 mg m$^{-2}$ h$^{-1}$. The CO$_2$ flux rates remained elevated until the last 2 wk of the 2007 growing season. The CO$_2$ flux rates were again elevated at the beginning of the 2008 growing season, with a similar increase during July (Fig. 3 and 4) corresponding to increases in soil moisture (Fig. 3). Calculations of cumulative CO$_2$ emissions found that N source applications did not significantly increase CO$_2$ emissions compared with the control treatment during the 2007 and 2008 growing seasons (Tables 3 and 4).

Evaluation of cumulative CO$_2$ emitted during the first month after application shows that the CO$_2$ emissions shortly after effluent application were consistently elevated compared with the control. The impact of effluent application was significant only in 2008, however, not in 2007 (Table 4).

In contrast to CO$_2$ emissions, the cumulative CH$_4$ emissions from soils after effluent application were much more responsive to treatment (Tables 3 and 4). In 2007, each effluent treatment resulted in elevated CH$_4$ flux rates compared with the control for 3 to 5 d after application (Fig. 2). In 2008, the injection treatment was the only treatment with cumulative CH$_4$ emissions significantly greater than the control (Table 4). In 2008, the N content of the effluent used was higher than that of the effluent used in 2007; therefore, a lower liquid volume was applied. This resulted in lower soil moisture values in 2008 after application (Fig. 1 and 3). This may explain why CH$_4$ emissions from the surface and aeration applications were not significantly elevated compared with the control. The data do show that injection optimizes CH$_4$ losses and that emission can occur for up to 11 d after application. These field data are consistent with the results from a recent laboratory study in which effluent injection was simulated. That laboratory study found that CH$_4$ could be emitted for up 12 d after effluent injection and that CH$_4$ emission would spike upward after water application during this time period (Jarecki et al., 2008).

The flux of N$_2$O from the four N source treatments was elevated compared with the control during the first 30 d after application in 2007 and 2008 (Fig. 2 and 4). A secondary spike in N$_2$O flux occurred on 6 July 2007. This spike corresponded to a 6.6-cm rainfall event, which occurred the day before this gas measurement. In 2007, the largest N$_2$O fluxes were measured from the injection treatment, which was the only treatment with cumulative CH$_4$ emissions significant compared with the control treatment during the 2008 growing season.

**Table 2. Select chemical characteristics of soils collected to a depth of 0 to 15 cm before treatment application on 3 Apr. 2007 and 2 Apr. 2008.**

<table>
<thead>
<tr>
<th>Year</th>
<th>pH</th>
<th>Total N</th>
<th>Total C</th>
<th>NH$_4$–N</th>
<th>NO$_3$–N</th>
<th>Ca</th>
<th>K</th>
<th>Mg</th>
<th>Mn</th>
<th>P</th>
<th>Zn</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>mg kg$^{-1}$</td>
<td>mg kg$^{-1}$</td>
<td>mg kg$^{-1}$</td>
<td>mg kg$^{-1}$</td>
<td>mg kg$^{-1}$</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>6.2</td>
<td>1241</td>
<td>12,306</td>
<td>7</td>
<td>6</td>
<td>1312</td>
<td>258</td>
<td>183</td>
<td>292</td>
<td>253</td>
<td>17.0</td>
</tr>
<tr>
<td>2008</td>
<td>6.3</td>
<td>1048</td>
<td>10,370</td>
<td>6</td>
<td>2</td>
<td>1429</td>
<td>166</td>
<td>58</td>
<td>199</td>
<td>110</td>
<td>2.6</td>
</tr>
</tbody>
</table>
Fig. 1. Daily rainfall totals measured throughout the measurement period, and soil moisture and volumetric soil moisture content measured in each treatment at each sampling time during the 2007 growing season (error bars represent the standard deviation).

Fig. 2. The flux of CO₂, N₂O, and CH₄ at each measurement time throughout the 141-d sampling period of the 2007 growing season (error bars represent the standard deviation).
from the UAN fertilizer and surface effluent treatments 5 d after application. These treatments were also responsible for the largest fluxes on 6 July 2007. In 2008, the N₂O flux from the UAN fertilizer treatment increased to its highest level 2 d after application and then experienced a steady decline. The effluent treatments responded differently. The surface effluent application resulted in maximum N₂O flux at 6 d after application. The N₂O flux from the aeration and injection treatments followed the same trend as the surface treatment; however, the N₂O flux from the aeration and injection treatments continued to increase and was maximized at 18 d after application, with flux rates of 0.91 and 1.89 mg N₂O m⁻² h⁻¹, respectively. The delayed flux of N₂O from the effluent treatments may result from the amount of NH₄⁻N applied in 2008. The NH₄⁻N content of the effluent applied in 2008 was only 25% of the total N applied (Table 1). Perhaps the delayed N₂O emissions from the effluent treatments resulted because the organic N was mineralized and then became available for denitrification and nitrification. This delayed N₂O emissions from swine effluent was also observed by Velthof et al. (2003).

Each N source application resulted in cumulative N₂O emissions that were greater than the control (Tables 3 and 4). Among the four N source application treatments, the UAN, surface effluent, and aeration effluent applications were consistently similar. In contrast, the relative response of N₂O emissions from the injection treatment was inconsistent. Table 3 shows that the injection treatment resulted in the lowest emission of N₂O among N sources in 2007. In 2008, this treatment had the highest cumulative N₂O emission among the N sources (Table 4). The elevated N₂O emissions from the injection treatment in 2008 is consistent with previous research showing that injection maximizes N₂O losses (Wulf et al., 2002; Perala et al., 2006; Velthof et al., 2003). The significant difference in N₂O emissions among N sources in 2007 is the result of differential emissions rates on 6 July 2007. Notice that the N₂O emissions during the first 31 d of the 2007 growing season were not significantly different among N source treatments. On 6 July 2007, N₂O emission from the injection treatment was similar to that of the control (Fig. 4). The lack of response of this treatment may have occurred because the 6.6-cm rainfall may not have sufficiently wet the soil at the injection depth of 20 cm. In a laboratory incubation, Velthof et al. (2003) found that N₂O fluxes where lower when swine effluent was placed at 10 cm than when it was placed at 5 cm. This effect was most prevalent after water was applied to the soil surface on Day 57 of the incubation. In 2008, the rainfall patterns were not such that a secondary flush of N₂O was measured.

In 2007, treatment did not significantly influence the cumulative global warming potential (GWP) as measured throughout the growing season. Of course, when calculated for the first 31 d, the GWP from each of the effluent treatments was elevated compared with the control (Table 5). This shows that evaluating GHG emissions during a short period after application can overestimate the significance of their impact on the GWP. In 2008, the GWP of the injection treatment was significantly larger than each of the other treatments except the aeration treatment (Table 6). This elevated GWP resulted from the CH₄ and N₂O emissions from the injection treatment, which separated this treatment from the remaining treatments during the first 31 d of the experiment.

**CONCLUSIONS**

The method by which swine effluent is applied for no-till corn grain production can have a significant influence on greenhouse gas emissions after application. The data suggest that effluent incorporation through aeration or injection application will increase the amount of CH₄ emitted after application compared with surface application. This increased CH₄ emission presumably results because of prolonged methanogenic activity in soils after incorporation. All N source applications resulted in significantly larger N₂O emissions compared with the control; however, differences among the N source application methods were inconsistent. In 2007, a secondary flush of N₂O at 70 d after effluent application resulted in significantly lower cumulative N₂O emissions from the injected effluent compared

| N source Tillage† Total N applied CO₂ CH₄ N₂O GWP‡ | kg N ha⁻¹ | g m⁻² | g m⁻² |
|---|---|---|---|---|
| Control | 0 | 777 | 0.06 | 0.24 | 853 |
| Urea–NH₄NO₃ no-till | 179 | 638 | 0.08 | 0.74 | 869 |
| Effluent no-till | 200 | 690 | 0.12 | 0.73 | 919 |
| Effluent aeration | 200 | 737 | 0.21 | 0.69 | 955 |
| Effluent injector | 200 | 847 | 0.26 | 0.47 | 998 |
| LSD (P < 0.1) | 180 | 0.05 | 0.23 | NS§ |

† For no-till, effluent and liquid urea–NH₄NO₃ was surface applied before planting without incorporation; for injection, effluent was injected; for aeration, effluent was applied with the aeration implement.

‡ GWP = the total global warming potential of trace gases emitted, where 1 g CO₂ m⁻² = 1 g GWP m⁻², 1 g CH₄ m⁻² = 21 g GWP m⁻², and 1 g N₂O m⁻² = 310 g GWP m⁻².

§ NS = not significant at the 0.1 probability level.

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|---|---|---|---|---|
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| Effluent aeration | 200 | 737 | 0.21 | 0.69 | 955 |
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§ NS = not significant at the 0.1 probability level.
Fig. 3. Daily rainfall totals measured throughout the measurement period, and soil moisture and volumetric soil moisture content measured in each treatment at each sampling time during the 2008 growing season (error bars represent the standard deviation).

Fig. 4. The flux of CO$_2$, N$_2$O, and CH$_4$ at each measurement time throughout the 158-d sampling period of the 2008 growing season (error bars represent the standard error).
with the remaining N source treatments. The rainfall event (6.6 cm) that occurred 1 d before this flush of N\textsubscript{2}O appears to have not wet the soil to the injection depth sufficiently to cause denitrification. In contrast to 2007, the injection treatment emitted the largest amount of N\textsubscript{2}O. These data show that both environmental conditions and application method must be considered when evaluating the impact of effluent management on N\textsubscript{2}O emissions.

The method of application did not influence the emissions from the effluent treatments. This indicates that the level of tillage used to incorporate the effluent through injection or aeration will not significantly increase CO\textsubscript{2} emissions from these otherwise no-till soils.

In 2007, there were no significant differences in GWP among treatments. In 2008, however, the GWP of the effluent injection treatment was significantly greater than that measured for the remaining treatments. This shows that effluent application technique can impact the GWP of the effluent when applied to no-till cropland. Specifically, effluent injection can optimize N\textsubscript{2}O and CH\textsubscript{4} emissions. Therefore, the application method needs to be considered when evaluating the impact of liquid manure management on GHG emission when it is utilized for crop production.

REFERENCES


Table 5. Greenhouse gas emission during the first 33 d of measurement in 2007 (27 Apr.–30 May).

<table>
<thead>
<tr>
<th>N source</th>
<th>Tillage†</th>
<th>Total N applied</th>
<th>CO\textsubscript{2}</th>
<th>CH\textsubscript{4}</th>
<th>N\textsubscript{2}O</th>
<th>GWP‡</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td></td>
<td>0</td>
<td>95</td>
<td>0.07</td>
<td>0.02</td>
<td>103</td>
</tr>
<tr>
<td>Urea–NH\textsubscript{4}NO\textsubscript{3} no-till</td>
<td>179</td>
<td>92</td>
<td>0.04</td>
<td>0.21</td>
<td>158</td>
<td></td>
</tr>
<tr>
<td>Effluent no-till</td>
<td>200</td>
<td>106</td>
<td>0.11</td>
<td>0.36</td>
<td>220</td>
<td></td>
</tr>
<tr>
<td>Effluent aeration</td>
<td>200</td>
<td>107</td>
<td>0.19</td>
<td>0.36</td>
<td>223</td>
<td></td>
</tr>
<tr>
<td>Effluent injector</td>
<td>200</td>
<td>118</td>
<td>0.22</td>
<td>0.30</td>
<td>216</td>
<td></td>
</tr>
<tr>
<td>LSD (P &lt; 0.1)</td>
<td>NS§</td>
<td>0.04</td>
<td>0.13</td>
<td>0.60</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

† For no-till, effluent and liquid urea–NH\textsubscript{4}NO\textsubscript{3} was surface applied before planting without incorporation; for injection, effluent was injected; for aeration, effluent was applied with the aeration implement.

‡ GWP = the total global warming potential of trace gases emitted, where 1 g CO\textsubscript{2} m\textsuperscript{−2} = 1 g GWP m\textsuperscript{−2}, 1 g CH\textsubscript{4} m\textsuperscript{−2} = 21 g GWP m\textsuperscript{−2}, and 1 g N\textsubscript{2}O m\textsuperscript{−2} = 310 g GWP m\textsuperscript{−2}.

§ NS = not significant at the 0.1 probability level.

Table 6. Greenhouse gas emission during the first 31 d of measurement in 2008 (19 Apr.–20 May).

<table>
<thead>
<tr>
<th>N source</th>
<th>Tillage†</th>
<th>Total N applied</th>
<th>CO\textsubscript{2}</th>
<th>CH\textsubscript{4}</th>
<th>N\textsubscript{2}O</th>
<th>GWP‡</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td></td>
<td>0</td>
<td>119</td>
<td>0.01</td>
<td>0.02</td>
<td>125</td>
</tr>
<tr>
<td>Urea–NH\textsubscript{4}NO\textsubscript{3} no-till</td>
<td>179</td>
<td>145</td>
<td>0.01</td>
<td>0.24</td>
<td>248</td>
<td></td>
</tr>
<tr>
<td>Effluent no-till</td>
<td>200</td>
<td>185</td>
<td>0.03</td>
<td>0.20</td>
<td>221</td>
<td></td>
</tr>
<tr>
<td>Effluent aeration</td>
<td>200</td>
<td>157</td>
<td>0.09</td>
<td>0.38</td>
<td>278</td>
<td></td>
</tr>
<tr>
<td>Effluent injector</td>
<td>200</td>
<td>168</td>
<td>0.80</td>
<td>0.69</td>
<td>397</td>
<td></td>
</tr>
<tr>
<td>LSD (P &lt; 0.1)</td>
<td>NS§</td>
<td>35</td>
<td>0.30</td>
<td>0.21</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

† For no-till, effluent and liquid urea–NH\textsubscript{4}NO\textsubscript{3} was surface applied before planting without incorporation; for injection, effluent was injected; for aeration, effluent was applied with the aeration implement.

‡ GWP = the total global warming potential of trace gases emitted, where 1 g CO\textsubscript{2} m\textsuperscript{−2} = 1 g GWP m\textsuperscript{−2}, 1 g CH\textsubscript{4} m\textsuperscript{−2} = 21 g GWP m\textsuperscript{−2}, and 1 g N\textsubscript{2}O m\textsuperscript{−2} = 310 g GWP m\textsuperscript{−2}.

§ NS = not significant at the 0.1 probability level.