Carbon Transformations in the Land Areas Receiving Organic Wastes in Relation to Nonpoint Source Pollution: A Conceptual Model

K. R. REDDY, R. KHALEEL, AND M. R. OVERCASH

ABSTRACT

A simple conceptual model based on current literature data was developed to describe organic carbon (C) loss from land areas receiving organic wastes. The model considers the decomposition of substrate C as represented by the evolution of CO₂. Decomposition of wastes was described in two or three phases, assuming first-order kinetics at each phase. The fraction decomposed at each phase was determined graphically. Rapid decomposition of an easily decomposable C fraction in Phase I and II was followed by the decomposition of more resistant C fraction(s). For plant residues, the amount of C decomposed in Phase I was significantly related to the log C/N ratio of the residue, whereas, a similar relationship was not observed for animal wastes. Decomposition rates were about 6 to 10 times faster during Phase I and II decomposition, as compared to Phase III decomposition. Kinetic rate constants, calculated at each phase of decomposition were adjusted for the soil temperature, soil moisture, and method of application. For plant residues, simulated results were in close agreement with observed results. No field data are available to test the complete model for animal waste decomposition. Transport of soluble C (easily decomposable C fraction) in leaching and surface runoff was discussed in the paper. Equations were presented to calculate the transport of waste particles (resistant C fraction) and sediment-associated C in the runoff water.

Future research needs identified include (i) a better understanding of C transformations; (ii) decomposition of individual C species, such as water-soluble C, cellulose, and lignin; (iii) a relationship between soluble and particulate C fractions in runoff water; (iv) mechanisms involved in the mode of C transport along with percolating water or in runoff water; and (v) extensive testing of existing models.

Additional Index Words: decomposition, plant residues, animal wastes, water quality.


Organic carbon (C) subjected to microbial decomposition in soil is derived from sources such as animal waste (land application sites and animal defection); plant residues (after harvest of corn, wheat, soybeans, and others); and applied sewage effluents, sludges, and industrial wastes. Animal wastes have been traditionally applied with an objective to supply plant nutrients, but before rainfall-runoff, increased microbial activity reduces the potential for runoff transport of C. During the period after waste application, the land application of organic waste does, however, increase the potential for runoff transport of C. The land application of organic wastes is organic material from animal digestion of feeds. Animal rations consist largely of carbohydrates (sugars, starches, hemicellulose, cellulose, pectins, gums, and mucilages); proteins, amino acids, and amines; fats, oils, waxes, and resins; alcohols, aldehydes, and ketones; organic acids; lignin; cellulose, and lignin. The composition of organic wastes with respect to these C species is shown in Table 1 and the composition based on total C is shown in Table 2. Roughly 90% of the dry matter in animal wastes is organic material from animal digestion of feeds. Animal rations consist largely of carbohydrates (sugars, starches, hemicelluloses, and celluloses); some proteins; fats; small amounts of lignin; and numerous inorganic nutrients, such as nitrogen, phosphorus, potassium, and micronutrients, which finally reflect on the composition of animal waste.

The decomposition of animal wastes or plant residues every growing season. The application of wastes for any purpose evidently increases soil C and improves the physical characteristics of the soil, thus, improving the infiltration capacity of the soil and reducing the amount of runoff water. The land application of organic waste does, however, increase the potential for runoff transport of C. During the period after waste application, the land application of organic waste decreases the potential pollution of water bodies. Thus, C-transformation kinetics are essential in assessing the nonpoint source pollutional impact of wastes applied to land (Fig. 1).

Decomposition of wastes applied to the soil depends on the nature of the wastes and on several soil and environmental factors. The main components of the waste materials include: (i) carbohydrates (sugars, starches, hemicelluloses, celluloses, pectins, gums, and mucilages); (ii) proteins, amino acids, and amines; (iii) fats, oils, waxes, and resins; (iv) alcohols, aldehydes, and ketones; (v) organic acids; (vi) lignin; (vii) cyclic or ring structure compounds (phenols, tannins, and cyclohydrocarbons); (viii) alkaloids and organic bases; and (ix) important miscellaneous substances present in very small amounts (antibiotics, auxins, vitamins, enzymes, and pigments) (Allison, 1973). Based on decreasing biodegradability, the waste C species can be grouped into (i) readily oxidizable soluble organic C, (ii) proteins, (iii) hemicellulose, (iv) cellulose, and (v) lignin. The composition of organic wastes with respect to these C species is shown in Table 1 and the composition based on total C is shown in Table 2. Roughly 90% of the dry matter in animal wastes is organic material from animal digestion of feeds. Animal rations consist largely of carbohydrates (sugars, starches, hemicelluloses, and celluloses); some proteins; fats; small amounts of lignin; and numerous inorganic nutrients, such as nitrogen, phosphorus, potassium, and micronutrients, which finally reflect on the composition of animal waste.

The decomposition of animal wastes or plant residues

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depends on the rate of degradation of each C species. However, sufficient data are not available in the literature to develop a simulation model based on the decomposition of individual C species. Recently, Parnas et al. (1975), Hunt (1977), Jenkinson and Rayner (1977) have presented simulation models to describe the decomposition of added substrate C in the soils. These models present a good basic understanding of C transformations, but are complex in nature and were developed to describe the decomposition of plant residues. Gilmour et al. (1977) presented a simple model to describe C transformations in soils treated with animal wastes. However, this model does not describe the decomposition of easily degradable fractions of the wastes. The models reported so far are not based on the current literature data, do not include the effect of management practices on rates of decomposition, and were not developed with an objective of measuring the influence of decomposition on surface runoff water quality. The increased interest in water quality modeling makes it necessary to develop a simple submodel to describe the C transformations. Such a submodel can be incorporated into a hydrologic model to describe water quality from land areas receiving organic wastes. The objectives of this research were (i) to review the available literature on C transformations in soils treated with organic wastes; (ii) to develop a conceptual model to describe the fate of C in the soil-waste system, (iii) to determine the effects of soil, environmental, and management factors on the rate of C decomposition; and (iv) to discuss the impact of C decomposition on the runoff transport of C and on water quality.

Table 1—Chemical composition of wastes and residues.

<table>
<thead>
<tr>
<th>Type of waste or residue</th>
<th>Water soluble C</th>
<th>Ether soluble substance</th>
<th>Cellulose</th>
<th>Hemicellulose</th>
<th>Lignin</th>
<th>Total N</th>
<th>Ash</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>% dry wt</td>
<td>% dry wt</td>
<td>% dry wt</td>
<td>% dry wt</td>
<td>% dry wt</td>
<td>% dry wt</td>
<td>% dry wt</td>
<td></td>
</tr>
<tr>
<td>Sheep manure</td>
<td>24.92</td>
<td>2.83</td>
<td>18.72</td>
<td>18.46</td>
<td>20.68</td>
<td>4.08</td>
<td>17.21</td>
<td>Waksman (1938)</td>
</tr>
<tr>
<td>Horse manure</td>
<td>5.58</td>
<td>1.89</td>
<td>27.46</td>
<td>23.52</td>
<td>14.23</td>
<td>1.09</td>
<td>9.11</td>
<td></td>
</tr>
<tr>
<td>Cow manure</td>
<td>19.34</td>
<td>2.77</td>
<td>25.43</td>
<td>18.57</td>
<td>20.21</td>
<td>2.38</td>
<td>12.95</td>
<td></td>
</tr>
<tr>
<td>Oat straw</td>
<td>5.83</td>
<td>2.26</td>
<td>39.10</td>
<td>19.27</td>
<td>15.79</td>
<td>0.42</td>
<td>4.20</td>
<td></td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>-</td>
<td>-</td>
<td>3.80</td>
<td>3.20</td>
<td>5.80</td>
<td>4.50</td>
<td>-</td>
<td>Rodulf and Ghem (1942)</td>
</tr>
</tbody>
</table>

Table 2—Total carbon concentration of the selected organic wastes.

<table>
<thead>
<tr>
<th>Type of waste</th>
<th>Mean</th>
<th>Min.</th>
<th>Max.</th>
<th>SD +</th>
<th>C.V.</th>
<th>Number of observations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Animal waste</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Beef</td>
<td>36.3</td>
<td>29.0</td>
<td>48.8</td>
<td>6.1</td>
<td>16.8</td>
<td>9</td>
</tr>
<tr>
<td>Dairy</td>
<td>43.1</td>
<td>41.0</td>
<td>44.6</td>
<td>1.7</td>
<td>4.0</td>
<td>4</td>
</tr>
<tr>
<td>Poultry</td>
<td>32.9</td>
<td>22.4</td>
<td>37.8</td>
<td>5.8</td>
<td>17.7</td>
<td>6</td>
</tr>
<tr>
<td>Swine</td>
<td>38.3</td>
<td>30.0</td>
<td>44.6</td>
<td>6.6</td>
<td>17.3</td>
<td>6</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anaerobic</td>
<td>27.6</td>
<td>18.0</td>
<td>39.0</td>
<td>-</td>
<td>-</td>
<td>31</td>
</tr>
<tr>
<td>Aerobic</td>
<td>31.7</td>
<td>27.0</td>
<td>37.0</td>
<td>-</td>
<td>-</td>
<td>10</td>
</tr>
<tr>
<td>Other sludges</td>
<td>32.6</td>
<td>6.5</td>
<td>48.0</td>
<td>-</td>
<td>-</td>
<td>60</td>
</tr>
<tr>
<td>Crop residues</td>
<td>40.1</td>
<td>29.8</td>
<td>48.0</td>
<td>4.4</td>
<td>10.8</td>
<td>11</td>
</tr>
</tbody>
</table>

† Data compiled by Sommers (1977).
‡ Includes lagooned, primary, tertiary, and unspecified sludges.

MODEL DESCRIPTION

The decomposition of organic matter from three sources, (i) animal waste, (ii) plant residues, and (iii) native soil organic matter, are considered in this model. Decomposition of organic C in the soil-waste system serves two functions for the microflora, providing energy for growth, and supplying C to form new cell material. The C content of most microbial cells is about 40 to 50%. Under aerobic decomposition about 20 to 40% of the substrate C is assimilated and the remainder is released as CO₂. During the decomposition of added substrate C, assuming that the aerobic population contains 50% C and 5% N and assimilates one-third of substrate C, then one unit of available N incorporated into cell material will allow for the assimilation of 10 units of cell C, but will be accompanied by 20 units of CO₂-C (Alexander, 1977). In quantifying the decomposition of added C, measuring CO₂ loss underestimates the actual substrate breakdown, because only 67% of the decomposed substrate appears as CO₂, while 33% becomes new cell C. For surface water quality, C remaining in the soil-waste system is of greater importance than C assimilated by microflora. Assimilated C remains in the system and is available for transport during rainfall-runoff events or upon death of microflora, it is subsequently decomposed to CO₂.

The biological breakdown or degradation of added waste or plant residue depends upon the different rate of degradation of each of the C species. Overall C decomposition in the soil-waste system is controlled by an individual reaction with a different reaction rate constant for each waste or plant residue fraction. During degradation, the more complex C compounds undergo initial degradation and eventual solubilization via reactions mediated by specific extracellular enzymes. The soluble C substances are used as an energy source by microflora and are oxidized to CO₂ through intracellular enzymes or are temporarily incorporated into cellular material (Alexander, 1977; Gilmour et al., 1977). The overall C decomposition cannot be described by simple kinetic equations. Recently, Gilmour et al. (1977) and Hunt (1977) used first-order kinetics to describe substrate C decomposition in two to three phases. In this model, decomposition of organic wastes was also assumed to occur in two to three phases depending on the ease of degradation of various C species. To simplify the model, the decomposition of added C was assumed to follow first-order kinetics at each phase of decomposition. The soluble and rapidly solubilized C substrates are mineralized during initial phases of decomposition followed by more complex substrate C decomposed at slower rates. The experimental data reported by several workers (see Tables 3 and 5 for references) validate the assumption of two to three phase decomposition of animal wastes and plant residues. The decomposition sequence of a complex substrate C can be described by the following equations:

\[ \frac{dC}{dt} = k_i C_i \]  

where subscript \( i \) refers to phase \( i = 1, II, \) and III.

Integrating Eq. [1] one obtains:

\[ C_i = C_i \exp(-k_i t) \]  

where

\[ C_i = \text{organic C present at the beginning of Phase I decomposition;} \]

\[ C_i = \text{organic C remaining at the end of Phase I decomposition;} \]
Composed at each phase of decomposition depends on the type and chemical nature of the waste.

**Animal Wastes**

The percent of applied C decomposed at each phase for different wastes was determined graphically (Fig. 2) and the data are presented in Table 3. The amount of C decomposed at each phase was represented as $D_1$ (Phase I), $D_2$ (Phase II), and $D_3$ (Phase III). The $D_1$ and $D_3$ represent easily decomposable fraction (labile), whereas $D_2$ represents more resistant residual fraction. Carbon decomposition in soils treated with beef wastes occurred in three phases, with average values of 24, 9, and 67% of applied C representing $D_1$, $D_2$, and $D_3$, respectively (Table 3). A large fraction of poultry waste C was present in the $D_1$ (65% of applied C) followed by $D_3$ (17% of applied C), and $D_2$ (18% of applied C). The waste material derived from beef and dairy animals contains more fibrous material because of the type of ration (such as roughages) fed to these animals. Wastes derived from poultry consist of less fibrous material, rich in N content, thus, leaving a small residual resistant fraction. Attempts were made to relate the fractions $D_1$, $D_2$, and $D_3$ to the total N or C/N ratio of the waste. No significant relationship was observed among these parameters. More data are needed on decomposition of other animal wastes, such as swine and dairy waste to obtain reliable estimates of these fractions. Average values of $D_1$, $D_2$, and $D_3$ presented in Table 4, can be used in simulation of C decomposition of the wastes.

First-order kinetic rate coefficients corresponding to each phase of decomposition were estimated using Eq. [2] and the data are presented in Tables 3 and 4. The values of $D_1$, $D_2$, and $D_3$ determine the amount of C to be decomposed in a particular phase. For example, Phase I decomposition will be terminated when the amount of C decomposed

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**Table 3—First-order constants for the decomposition of carbon in the soils treated with wastes.**

<table>
<thead>
<tr>
<th>Type of waste</th>
<th>Temperature, °C</th>
<th>Rate constant, day$^{-1}$</th>
<th>Percent of added C decomposed at each phase</th>
<th>Half-life of C remaining, days</th>
<th>Remarks</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>$k_1$</td>
<td>$k_2$</td>
<td>$k_3$</td>
<td>$D_1$</td>
<td>$D_2$</td>
</tr>
<tr>
<td><strong>Feedlot waste</strong></td>
<td>27</td>
<td>0.0248</td>
<td>0.0120</td>
<td>0.0058</td>
<td>22</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>30°F</td>
<td>0.0304</td>
<td>0.0147</td>
<td>0.0071</td>
<td>27</td>
<td>15</td>
</tr>
<tr>
<td><strong>Feedlot waste</strong></td>
<td>27</td>
<td>0.0375</td>
<td>0.0114</td>
<td>0.0031</td>
<td>27</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>30</td>
<td>0.0459</td>
<td>0.0139</td>
<td>0.0038</td>
<td>37</td>
<td>12</td>
</tr>
<tr>
<td><strong>Dairymanure</strong></td>
<td>27</td>
<td>0.0062</td>
<td>0.0021</td>
<td>0.0007</td>
<td>36</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>30</td>
<td>0.0212</td>
<td>0.0078</td>
<td>0.0030</td>
<td>30</td>
<td>17</td>
</tr>
</tbody>
</table>

---

**Beef cattle waste at different moisture levels**

<table>
<thead>
<tr>
<th>Moisture level</th>
<th>Temperature, °C</th>
<th>$k_1$</th>
<th>$k_2$</th>
<th>$k_3$</th>
<th>$D_1$</th>
<th>$D_2$</th>
<th>$D_3$</th>
<th>Remarks</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 bar</td>
<td>25</td>
<td>0.0044</td>
<td>0.0020</td>
<td>0.0018</td>
<td>9</td>
<td>4</td>
<td>87</td>
<td>267 Laboratory study at different loading rates and moisture levels, 60-day decomposition period. Weld sandy loam.</td>
<td>Ruehr (1976)</td>
</tr>
<tr>
<td>0.1 bar</td>
<td>25</td>
<td>0.0096</td>
<td>0.0060</td>
<td>0.0041</td>
<td>18</td>
<td>10</td>
<td>72</td>
<td>116 Laboratory study with high loading rate, 90-day decomposition period. Pullman clay loam.</td>
<td>Mathers and Stewart (1970)</td>
</tr>
<tr>
<td>0.33 bar</td>
<td>25</td>
<td>0.0141</td>
<td>0.0088</td>
<td>0.0060</td>
<td>18</td>
<td>8</td>
<td>74</td>
<td>277 Laboratory study with medium loading rate, 90-day decomposition period. Pullman clay loam.</td>
<td>Mathers and Stewart (1970)</td>
</tr>
<tr>
<td>1.0 bar</td>
<td>25</td>
<td>0.0214</td>
<td>0.0086</td>
<td>0.0025</td>
<td>15</td>
<td>4</td>
<td>81</td>
<td>462 Laboratory study with low loading rate of waste, 90-day decomposition period.</td>
<td>Mathers and Stewart (1970)</td>
</tr>
<tr>
<td>5.0 bar</td>
<td>25</td>
<td>0.0073</td>
<td>0.0019</td>
<td>0.0007</td>
<td>14</td>
<td>3</td>
<td>83</td>
<td>693 Laboratory study with low loading rate of waste, 90-day decomposition period.</td>
<td>Mathers and Stewart (1970)</td>
</tr>
</tbody>
</table>

---

**Poultry waste**

| 0.1 bar | 25 | 0.0212 | 0.0041 | 9      | 4      | 87     | 267 Laboratory study at different loading rates and moisture levels, 60-day decomposition period. Weld sandy loam. | Ruehr (1976)  |
| 0.33 bar | 25 | 0.0067 | 0.0014 | 0.0002 | 25     | 75     | 25     | 148 Laboratory study with manure only. | Hashimoto (1974)  |
| 1.0 bar | 25 | 0.0220 | 0.0047 | 0.0007 | 60     | 17     | 23     | 990 Laboratory study with 2.5% loading rate. | Finsen (1972)  |
| 5.0 bar | 25 | 0.0080 | 0.0019 | 0.0009 | 70     | 16     | 14     | 231 Laboratory study with 1.0% loading rate. | Finsen (1972)  |

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**Sewage sludge**

| 21 | 0.0092 | 0.0052 | 30 | 70 | 71 Laboratory study. Short term incubation study (54 days). | Youch (1976) |

---

1 Rate constants were adjusted using Eq. [6], for relative comparison of the data.
is equal to the \( D_2 \) fraction \((100 - C_2)\) (see Fig. 2). The calculated \( k \) values were measured at different temperatures, but under optimum moisture conditions. For relative comparison, these values were further adjusted to a constant temperature of 30°C using an equation presented in the later part of the paper. The decomposition of feedlot waste at Phase I, II, and III occurred at a rate of 0.0295, 0.0098, and 0.0036 day\(^{-1}\), respectively. Even though poultry wastes contained a large fraction of added C in the \( D_2 \) fraction, decomposition rates were approximately the same as the feedlot wastes. No significant relationship was observed between \( k \) values and corresponding amounts decomposed at each phase.

**Plant Residues**

The decomposition of plant residues in the soil system is fairly well understood compared to animal wastes. Plant residue decomposition in the soil system occurs in two phases. The amount of C decomposed in Phase I, represented as \( D_1 \) (labile fraction, easily decomposable) and Phase III, represented as \( D_3 \) (residual fraction, resistant and slowly decomposable). No phase II (\( D_2 \) fraction) decomposition observed for plant residues. These fractions were determined graphically as shown in Fig. 2 and results are presented in Table 5. A significant relationship was obtained between the \( D_1 \) values (percent of added C decomposed at Phase I) and the C/N ratio of the plant residue (Pinck et al., 1950; Reddy and Patrick, 1975).

\[
D_1 = 86.64 - 13.95 \ln(\frac{C}{N}) \quad (r = 0.97**; n = 10) \tag{3}
\]

where

\[
D_1 = \frac{\text{percent of added C decomposed at Phase I}}{C/N} \quad (\text{carbon-nitrogen ratio of the plant residue},
\]

**Table 4**—Average values for \( D_1, D_2, \) and \( D_3 \) with respective \( k_1, k_2, \) and \( k_3 \) values as averaged from Table 2.

<table>
<thead>
<tr>
<th>Type of waste</th>
<th>( D_1 )</th>
<th>( D_2 )</th>
<th>( D_3 )</th>
<th>( k_1 )</th>
<th>( k_2 )</th>
<th>( k_3 )</th>
<th>No. of sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Feedlot waste</td>
<td>24</td>
<td>9</td>
<td>67</td>
<td>0.0296</td>
<td>0.0098</td>
<td>0.0036</td>
<td>10</td>
</tr>
<tr>
<td>Poultry waste</td>
<td>65</td>
<td>17</td>
<td>18</td>
<td>0.0264</td>
<td>0.0056</td>
<td>0.0019</td>
<td>2</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>30</td>
<td>70</td>
<td></td>
<td></td>
<td>0.0169</td>
<td></td>
<td>1</td>
</tr>
</tbody>
</table>

\[
D_3 = 100 - D_1 \tag{4}
\]

where \( D_3 = \frac{\text{percent of added C decomposed in Phase III}}{\text{percent of added C not decomposed}} \). The kinetic rate coefficients calculated for each phase decomposition are presented in Table 5. The \( k \) values estimated are based on the

**Table 5**—First-order rate constants for the decomposition of carbon in the soils treated with plant residues.

<table>
<thead>
<tr>
<th>Type of plant residues</th>
<th>Temperature, °C</th>
<th>Rate constant/day</th>
<th>Percent of added C decomposed at each phase</th>
<th>Half-life of C remaining, days</th>
<th>Remarks</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>( k_1 )</td>
<td>( k_2 )</td>
<td>( D_1 )</td>
<td>( D_2 )</td>
<td></td>
</tr>
<tr>
<td>Barley tops + soil</td>
<td>25</td>
<td>0.0052</td>
<td>0.0010</td>
<td>25</td>
<td>75</td>
<td>495 Field study—surface applied. Laboratory incubation study, 180 days decomposition period.</td>
</tr>
<tr>
<td>Barley roots + soil</td>
<td>25</td>
<td>0.0052</td>
<td>0.0010</td>
<td>25</td>
<td>75</td>
<td>495 Field study—incorporated. Laboratory incubation study, 180 days decomposition period.</td>
</tr>
<tr>
<td>Soil organic matter</td>
<td>25</td>
<td>0.0073</td>
<td>0.0014</td>
<td>36</td>
<td>64</td>
<td>365 Laboratoiy study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Rice straw (aerobic)</td>
<td>30</td>
<td>0.0054</td>
<td>0.0013</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory incubation study, 128 days decomposition. Crowley silt loam soil.</td>
</tr>
<tr>
<td>Cornstalk residues</td>
<td>--</td>
<td>0.0049</td>
<td>--</td>
<td>52</td>
<td>48</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Oat straw</td>
<td>35</td>
<td>0.0011</td>
<td>0.0018</td>
<td>28</td>
<td>72</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Casein (C/N = 3.1)</td>
<td>28</td>
<td>0.0308</td>
<td>0.0016</td>
<td>28</td>
<td>72</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Bluegrass (C/N = 9.7)</td>
<td>28</td>
<td>0.0163</td>
<td>0.0011</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Oats (C/N = 10.6)</td>
<td>28</td>
<td>0.0168</td>
<td>0.0011</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Millet (C/N = 10.7)</td>
<td>28</td>
<td>0.0136</td>
<td>0.0013</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Soybeans (C/N = 15.0)</td>
<td>28</td>
<td>0.0182</td>
<td>0.0014</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Lespedeza (C/N = 18.1)</td>
<td>28</td>
<td>0.0128</td>
<td>0.0010</td>
<td>53</td>
<td>47</td>
<td>363 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Oak leaves (C/N = 64.3)</td>
<td>30</td>
<td>0.0149</td>
<td>0.0012</td>
<td>22</td>
<td>78</td>
<td>462 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Wheat straw (C/N = 73)</td>
<td>28</td>
<td>0.0055</td>
<td>0.0013</td>
<td></td>
<td></td>
<td>578 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Corn stover (C/N = 102)</td>
<td>28</td>
<td>0.0084</td>
<td>0.0015</td>
<td>23</td>
<td>77</td>
<td>495 Laboratory study. Ammonium carbonate was added as energy source.</td>
</tr>
<tr>
<td>Leaf litter (miscanthus sinensis)</td>
<td>--</td>
<td>0.0012</td>
<td>--</td>
<td>100</td>
<td>578 Field study, 5-year decomposition period. No temperature data available.</td>
<td>Yamane (1974)</td>
</tr>
<tr>
<td>Soil organic matter</td>
<td>30</td>
<td>0.0027</td>
<td>--</td>
<td>100</td>
<td>578 Field study, 5-year decomposition period. No temperature data available.</td>
<td>Yamane (1974)</td>
</tr>
</tbody>
</table>


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initial substrate C, but are not based on the amount of C decomposed in Phase I. A significant relationship was observed between $k_i$ (Phase I decomposition rate) and $D_i$ (percent of added C decomposed in Phase I) as presented by the following equation:

$$k_i = 0.00035 D_i - 0.0013 \quad (r = 0.73^{**}; n = 18) \quad [5]$$

where $k_i$ = first-order rate constant for the decomposition of added C at phase $i$, day$^{-1}$. However, no significant relationship was observed between $k_i$ (Phase III decomposition rate) and $D_i$ (percent of added C decomposed at Phase III). The average $k_i$ value obtained for plant residues is $0.0015 \pm 0.0004$ day$^{-1}$ at $30^\circ C$.

The kinetic rate coefficients obtained for animal wastes and plant residues are approximately of the same order of magnitude. However, the decomposition of C in animal wastes occurred in three distinct phases as compared to the two-phase decomposition for plant residues. These differences are due to the nature of the waste and its composition.

**Soil Organic Matter**

Nature of the C material in the soil organic matter represents the $D_i$ fraction. This C decomposes very slowly in the soil. Native soil organic C decomposition can be described by Phase III rate constant, $0.0015 \pm 0.0004$ day$^{-1}$, an average value obtained from several Phase III decomposition constants.

### FACTORS INFLUENCING THE DECOMPOSITION OF CARBON

A number of soil, environmental, and management factors affect the mineralization of added and native organic C. These include temperature, pH, soil moisture, oxygen supply, available nutrients, soil structure and texture, carbon/nitrogen ratio, lignin content, particle size, physical nature of the waste, nature of the indigenuous microflora, biochemical oxygen demand (BOD), type of waste material, and method of application. The most important ones, and for which some experimental data are available, were considered in the model to adjust the rate constants ($k$) for the influence of these factors. These are temperature, soil moisture, and method of application.

**Soil Temperature**

Maximum rates of biological activity occur at temperatures as high as 45 to 65$^\circ C$, but such high temperatures seldom exist in the soil system. Below this optimum temperature, an increase in temperature accelerates the decomposition of C. To make comparisons among different studies, a temperature quotient was calculated, i.e., the ratio of the rate constants of decomposer activity at two temperatures, which is often termed $A_{10}$ (i.e., decomposition rate ratio at an interval of a 10$^\circ C$ rise in temperature). A linear relationship was observed between $Q_{10}$ values and the reciprocal of a 10$^\circ C$ rise in temperature. At low temperature, the reaction rate increased by 2 to 4 times for each 10$^\circ C$-rise in temperature (Waksman and Gerretsen, 1931; Bunt and Rovera, 1955; Waksman and Gerretsen, 1931; Bunt and Rovira, 1955; Drubnik, 1962; Moureaux, 1967; and Volk, 1973). However, the lowest temperature studied by these workers was 5$^\circ C$. The rate of C-decomposition in the model was adjusted for changes in temperature, using a simplified form of the Arrhenius equation. To describe the effect of temperature on the reaction rate constant, the Arrhenius equation is written as follows:

$$d(ln k)/dT = -Ea/RT^2. \quad [6]$$

On integration Eq. [6] becomes

$$k_{i2}/k_{i1} = (Ea/R) [(T_1 - T_i)/(T_2 - T_i)] \quad [7]$$

where

$\phi_{i2}/k_{i1} = (Ea/R) [(T_1 - T_i)/(T_2 - T_i)].$

Equation [8] can also be written as

$$k_{i2} = k_{i1} \exp [\phi (T_2 - T_1)]. \quad [9]$$

Equation [9] can be further simplified, if $\ln \phi = \theta$; then,

$$k_{i2} = k_{i1} \theta (T_2 - T_1) \quad [10]$$

where $\theta$ = temperature correction coefficient.

The $\theta$ values were estimated from the C decomposition rates reported by several workers (see Table 6 for references). The $\theta$ values range from 1.04 to 1.12 with an average value of 1.07 $\pm$ 0.03.

**Soil Moisture**

Many of the experiments reported on C decomposition have expressed soil moisture as the percent of soil

<table>
<thead>
<tr>
<th>Description of the experiment</th>
<th>Temperature range, $^\circ C$</th>
<th>Temperature correction coefficient $\theta$</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil carbon (Histosol soil)</td>
<td>--</td>
<td>1.04</td>
<td>Volk (1973)</td>
</tr>
<tr>
<td>Plant residues</td>
<td>5-45</td>
<td>1.06</td>
<td>Waksman and Gerretsen (1931)</td>
</tr>
<tr>
<td>Soil carbon</td>
<td>18-53</td>
<td>1.10</td>
<td>Moureaux (1967)</td>
</tr>
<tr>
<td>Soil carbon</td>
<td>12-48</td>
<td>1.08</td>
<td>Drubnik (1962)</td>
</tr>
<tr>
<td>Soil carbon</td>
<td>15-53</td>
<td>1.12</td>
<td>Bunt and Rovera (1955)</td>
</tr>
<tr>
<td>Beef feedlot wastes</td>
<td>5-27</td>
<td>1.07</td>
<td>Gilmore et al. (1977)</td>
</tr>
<tr>
<td>Glucose</td>
<td>4-37</td>
<td>1.06</td>
<td>Mayaudon (1971)</td>
</tr>
<tr>
<td>Average</td>
<td>5-53</td>
<td>1.07 $\pm$ 0.03</td>
<td></td>
</tr>
</tbody>
</table>

Table 6—Effects of temperature on decomposition of carbon.
moisture per gram of soil, or as the percent of water-holding capacity (Gilmour et al., 1977; Pal and Broadbent, 1975), or as soil moisture tension (Miller and Johnson, 1964; and Ruehr, 1976b). These different approaches make it difficult to compare results. Since the percent of soil moisture per gram of soil and the percent of water-holding capacity vary for different soils, the best approach is to express the results based on soil moisture tension, which allows a comparison among several experiments. The rate of respiration generally reached a maximum value at tensions ranging from 0.05 to 0.33 bars. The decomposition rate at zero tension (saturated condition) exhibited a 1.1- to 3.1-fold decrease relative to the maximum value. At a soil moisture tension of 3 bars, a 1.1- to 1.5-fold decrease was observed in respiration relative to maximum activity. At tensions >50 bars (air dry soil) a 12- to 13.5-fold decrease in activity occurred. Utilizing the data presented by Miller and Johnson (1964) and Ruehr (1976b), the following equations were obtained to calculate relative rates of decomposition at a given moisture tension:

\[ F_m = 1.223 + 0.201 \ln \Psi; 0.02 \leq \Psi \leq 0.33 \]  
\[ F_m = 0.874 - 0.115 \ln \Psi; 0.33 \leq \Psi \leq 10.0 \]

where \( F_m \) = relative rate of decomposition at a particular moisture tension (ranges from 0 to 1), and \( \Psi \) = moisture tension, bars. Rate constants are adjusted by multiplying with a factor, \( F_m \), obtained for a particular moisture tension.

**Method of Application**

Experimental work has demonstrated that surface-applied wastes or residues decomposed more slowly than soil-incorporated residues (Parker et al., 1957; Parker, 1962; Brown and Dickey, 1970; King, 1973; Sain and Broadbent, 1977). This is more the case for plant residues than animal wastes. Animal wastes with more fibrous material (such as solid beef or dairy wastes) decomposed more slowly when applied on the soil surface, as compared to incorporation, whereas, poultry and swine wastes, which are in semi-liquid form, decomposed at approximately the same rate under both methods of application (Reddy et al., 1980). These conclusions, however, were based on a shallow incorporation of wastes in soil columns incubated under laboratory conditions. The influence of method of application was expressed by obtaining relative rates of decomposition to the optimum level where waste is incorporated (Table 7).

The rate constant at each phase decomposition shown in Eq. [2] was adjusted using the following equation:

\[ k_{i2} = k_{i1} \theta(T_i - T) F_m * F_{ma} \]  

where \( k_{i2} \) = rate constant for phase \( i \) adjusted to soil temperature, soil moisture, and method of application;

\[ k_{i1} = \text{rate constant measured at phase } i \text{ adjusted to temperature, } T, \text{ and at optimum level of other factors}; \]

\[ T_i = \text{temperature, °C, of the system to which rate constant needs to be adjusted}; \]

\[ T = \text{temperature, °C, at which } k_{i1} \text{ was measured}; \]

\[ F_m = \text{correction factor for changes due to soil moisture tension}; \]

\[ F_{ma} = \text{correction factor for changes due to method of application}. \]

**MODEL VALIDATION**

No data are available to test the model for the decomposition of animal waste under field conditions. However, data on plant residue decomposition under field conditions were used to test the validity of the model. The rate of decomposition of blue grama sods labeled with \(^14\)C, was presented by Nyhan (1975) for a period of 1 year under field conditions (Pawnee Site, Colo.). The blue grama had a C/N ratio of 29.7. Utilizing Eq. [3] and [4], the percents of added C decomposed at each phase were 39.3 and 60.7% for \( D \), and \( D_3 \), fractions, respectively. The k value estimated for Phase I decomposition was 0.0125 day\(^{-1}\) at 30°C (using Eq. [5]). For Phase III decomposition, an average steady state rate constant of (0.0015 day\(^{-1}\) at 30°C) was used. The C remaining in the soil system was simulated for 11 Feb. 1971, and 20 May 1971, (when ground blue grama herbage was incorporated into the soil) applications. The average monthly soil temperature and soil water content, as presented by Nyhan (1975), were used to adjust \( k \) values at each phase using Eq. [13]. The simulated values and the experimental values are presented in Table 8. Phase I decomposition (easily decomposable fraction) occurred up to 126 days for the 11 Feb. 1971 application and 50 days for the 20 May 1971 application followed by Phase III decomposition (slowly decomposable fraction). The longer period required for Phase I decomposition was due to cooler

### Table 7—Relative rate of organic matter decomposition as influenced by method of application.

<table>
<thead>
<tr>
<th>Type of waste of plant residue</th>
<th>Incorporated</th>
<th>Surface applied</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beef waste</td>
<td>1.00</td>
<td>0.70</td>
<td>Reddy et al. (1980)</td>
</tr>
<tr>
<td>Poultry waste</td>
<td>1.00</td>
<td>0.93</td>
<td></td>
</tr>
<tr>
<td>Swine waste</td>
<td>1.00</td>
<td>1.05</td>
<td></td>
</tr>
<tr>
<td>Liquid sewage sludge</td>
<td>1.00</td>
<td>0.53</td>
<td>King (1973)</td>
</tr>
<tr>
<td>Corn stalks</td>
<td>1.00</td>
<td>0.81</td>
<td>Parker (1962)</td>
</tr>
<tr>
<td>Wheat straw</td>
<td>1.00</td>
<td>0.37</td>
<td>Brown and Dickey (1970)</td>
</tr>
<tr>
<td>Rice straw</td>
<td>1.00</td>
<td>0.63</td>
<td>Sain and Broadbent (1977)</td>
</tr>
</tbody>
</table>

\[ k_{i1} = \text{rate constant measured at phase } i \text{ adjusted to } \]

\[ T = \text{temperature, °C, of the system to which rate constant needs to be adjusted}; \]

\[ T_i = \text{temperature, °C, at which } k_{i1} \text{ was measured}; \]

\[ F_m = \text{correction factor for changes due to soil moisture tension}; \]

\[ F_{ma} = \text{correction factor for changes due to method of application}. \]

### Table 8—Relative comparison of the simulated results with the experimental values for the soil treated with blue grass herbage.

<table>
<thead>
<tr>
<th>Time of ground herbage application</th>
<th>Phase I decomposition, days</th>
<th>Percent of added C decomposed in Phase I</th>
<th>Percent of added C remaining after 250 days</th>
</tr>
</thead>
<tbody>
<tr>
<td>11 Feb. 1971</td>
<td>126</td>
<td>50</td>
<td>47</td>
</tr>
<tr>
<td>20 May 1971</td>
<td>50</td>
<td>30</td>
<td>60</td>
</tr>
</tbody>
</table>

temperatures. During this period, microbial activity was probably low. During warmer months (20 May 1971 application) microbial activity was probably high, resulting in a shorter period for Phase I decomposition. Considering the assumptions made in estimating the \( D_1, D_2, k_1, \) and \( k_2 \) values, agreement between simulated and observed values was satisfactory. The \( k \) values used in the simulation were obtained from independent experiments. Further testing of the model is necessary to derive more reliable conclusions. It is still not clear to what extent environmental and soil variables affect the rate of waste decomposition under field conditions.

**COMPARISON OF LONG-TERM DECOMPOSITION OF ANIMAL WASTES WITH THE MODEL**

The application of animal wastes or plant residues increases the C content of the soil. The rate of waste C accumulation in a soil system is dependent on the rate of decomposition of waste and the influence of soil and environmental factors. Literature data (Gupta et al., 1977; Kladivako, 1977; Mathers and Stewart, 1974; Mays et al., 1973; Tiarks et al., 1974; Unger and Stewart, 1974) were obtained relating the effect of the waste loading rate to the net increase in C content of the surface 15-cm soil layer. A simple regression equation was obtained from the literature data as

\[
C_{inc} = 0.037 \times WLR \quad (r = 0.93** \quad n = 28) \quad [14]
\]

where

\( C_{inc} = \) carbon content increase in the soil system (g kg soil\(^{-1}\) yr\(^{-1}\)) at the end of each year after application;

\( WLR = \) waste loading rate, metric tons ha\(^{-1}\) yr\(^{-1}\); and

\(* *= \) significant at 0.01 level of probability.

If the net increase in C content of the soil is the waste C remaining in the soil system at the end of 1 year, then an approximate decomposition rate constant can be estimated assuming first-order kinetics. These data were neither used in developing the model rate constants nor environmental corrections factors, and hence, present a comparison to the model. If a waste is applied at a rate of 200 metric tons ha\(^{-1}\) yr\(^{-1}\), (70,000 kg C ha\(^{-1}\) yr\(^{-1}\)), assuming 35% C in the waste, the net increase in C content at the end of 1 year can be estimated from Eq. [14] as about 5.4 g of C/kg of soil/yr. If we assume the bulk density of the soil after waste application to be 1.4 g/cm\(^3\), the resulting soil weight in the 15-cm plow layer will be 2.1 \times 10^4 kg soil/ha. The total waste C increase at the end of 1 year is about 11,340 kg C/ha, which is the same as the C remaining at the end of 1 year of decomposition. Assuming a gross first-order rate process, an average \( k \) value for all phases of decomposition is given by

\[
k = \ln \left( \frac{70,000}{11,340} \right) / 365. \quad [15]
\]

The \( k \) value obtained from Eq. [15] is 0.005 day\(^{-1}\), which is an approximate steady decomposition rate constant for waste, and agrees with the \( k \) values presented in Table 2. The calculated \( k \) value includes the average effects of soil, environmental, and management factors.

**TRANSPORT OF WASTE CARBON IN RELATION TO WATER QUALITY**

Organic matter transported from land areas receiving wastes serves as a substrate for bacteria when it enters a receiving stream. When the rate of oxygen consumption by bacteria exceeds the respiration rate of the stream, oxygen depletion occurs, thus, creating an adverse ecological impact on water quality. The substrate C content in surface waters is generally expressed as biochemical oxygen demand (BOD), chemical oxygen demand (COD), or total organic carbon (TOC).

The characteristics of the animal wastes with respect to BOD, COD, or TOC are primarily dependent upon the chemical characteristics of the feed processed through the animals. Only a fraction of the feed consumed by any animal is processed into animal tissue. When wastes are characterized in relation to pollutants, the parameters generally used are BOD or COD, whereas, with respect to waste accumulation in the soil system, the parameter used is TOC. When animals (beef and dairy) are fed with roughages containing more complex carbonaceous material, the oxygen demand of the waste produced will be higher compared to the ration containing easily digestable material rich in proteins. Sheep and lamb wastes were shown to have a wider COD/BOD ratio (13.9:1.0), indicating that these wastes contain more biologically resistant carbonaceous material, followed by beef and dairy wastes with a COD/BOD ratio of 5.4:1.0. Swine and poultry wastes fall approximately in the same range of 3.6 to 4.1:1. Plant residue incorporation after the harvest and the dead plant material (from grasses) in pastures also add to the oxygen demand in the surface runoff waters.

The transport of C in the runoff water is also controlled by management practices. The surface application of waste poses a greater problem in relation to water quality compared to the incorporation of waste into the soil. Waste spread on a frozen ground tends to be transported easily during snow melt as compared to summer application. During winter months, biological activity is slow compared to summer months, resulting in less loss of added C as CO\(_2\) in winter.

Only a small fraction of C present in the soil organic C is in water soluble form moving downward through leaching. From the data available for 55 soils (Burford and Bremmer, 1975; Reddy and Patrick, 1979\(^{1}\)), a linear relationship was obtained between water-extractable C and TOC as follows:

\[
WSC = 0.0031 \times (TOC + 24.5) \quad [16]
\]

where

\( WSC = \) water-soluble C, \( \mu g/g \) of soil;

For modeling of C transport in the land areas receiving wastes, the $D_i$ fraction of the applied wastes can be assumed as the water-soluble fraction which is readily available for transport in runoff water. However, further research is needed to establish the relationship between water-extractable C of the soils treated with wastes and residues, and total C of the system. Soluble C is extracted during a rainfall event and will move along with infiltrating water. A fraction of it will also be transported in surface runoff waters.

It is not clear how much of the waste transported is present in soluble form and particulate form. The data relating to the ratio between soluble and particulate fractions are needed in characterizing the runoff water quality. The ratio between particulate C and soluble C will be small in fresh wastes immediately after land application. As the decomposition proceeds, the soluble C fraction ($D_i$ fraction) decreases, thus, widening the ratio between particulate and soluble C fractions. At this stage, the waste particulate C behavior is similar to that of soil organic matter. There is a greater pollution potential with respect to soluble C transport if a rainfall-runoff event occurs during Phase I decomposition.

Another important mode of transport of C is the direct transport of waste particles and sediment-associated C. The mass of waste particles and sediment transported can be estimated from the model described by Khaleel et al. (1979). The concentration of C in the waste particles of the soil-waste system will remain the same, thus, no enrichment occurs. However, C transported along with sediments is enriched approximately 1.5 to 5 times (Neal, 1944; Knoblauch et al., 1942; Massey and Jackson, 1952). From the data reported by these researchers, an average enrichment coefficient of 2.85 ± 1.40 was obtained. The total C transported from a soil-waste system can be estimated from the following equation:

$$\text{TOCL} = [(S_2)(\text{TOC}_s)(EC)] + [(M_2)(\text{TOC}_m)] \quad [17]$$

where:
- $\text{TOCL}$ = total organic C load transported in runoff water, kg/ha;
- $S_2$ = sediment yield from soil source, kg/ha;
- $\text{TOC}_s$ = total organic C concentration in the soil (expressed as fraction) before rainfall-runoff event;
- $EC$ = enrichment coefficient (2.85 ± 1.40);
- $M_2$ = mass of manure transport from manure source, kg/ha; and
- $\text{TOC}_m$ = total organic C concentration (expressed as fraction) of manure before rainfall-runoff event.

The transformation submodel and the transport of C can be incorporated into a hydrologic and sediment transport model to describe the impact of animal waste application on water quality. The subroutine developed for C transformation can also be used to describe the fate of applied C in soils treated with municipal or industrial wastes. More extensive testing of the model is not available at the present, because of limited data availability. It is suggested that further research be directed towards a better understanding of C transformations and transport processes in land areas receiving wastes. Data are needed on the rate of C decomposition in soils treated with different wastes under field conditions. This research should be oriented to develop the relationship between C decomposition rates and soil and environmental variables. Data are also not available on the mode of C transport along with percolating water or in runoff water. Research should be oriented toward developing the relationship between soluble and particulate fractions of C in several types of soils treated with different kinds of wastes. The partition coefficients developed from these data can be used in characterizing the runoff water quality. The decomposition of individual C species of wastes (such as cellulose, lignin, etc.) should be considered in future experimental work. These experiments will generate data for more extensive testing of the models.

**SUMMARY AND CONCLUSIONS**

A simple workable submodel was developed to describe the decomposition of added C. The model considers decomposition of aggregate substrate C as represented by evolution as CO$_2$. Substrate C assimilated by microflora during synthesis of new cell material was not considered separately in the decomposition, since this C still remains in the soil waste system.

Decomposition of animal wastes and plant residues was shown to occur in 2 or 3 phases. The amount of C decomposed at each phase was estimated graphically. For animal wastes no relationship was observed between the amount of C decomposed in Phase I and the C/N ratio or the N content of the waste. For plant residues, the amount of C decomposed in Phase I (easily decomposable fraction) was related significantly to the log of the C/N ratio of the residue. The rate of decomposition at each phase was described assuming first-order kinetics.

The kinetic rate constants calculated at each phase of decomposition were adjusted during simulation for the changes in soil temperature, soil water content, and method of waste application. No field data are available to test the complete model for animal waste decomposition. However, simulated results were compared with the plant residue decomposition data obtained under field conditions by Nyhan (1975). Considering the assumptions made in the model, the agreement between predicted and observed values are good. An independent comparison was also made with the long-term decomposition of animal wastes measured as an increase in soil organic matter and the model gave reasonable predictive results.

A simple linear regression equation was presented to estimate the water-soluble C fractions that can be removed in leaching soil organic C during a rainfall event. Equations were also presented to estimate the transport of C in the runoff water from soil and waste sources as it relates to sediment and manure particle erosion.

**LITERATURE CITED**


Carbon Transformations in the Land Areas Receiving Organic Wastes in Relation to Nonpoint Source Pollution: A Conceptual Model


An error was made in Eq. [8] on page 438 of the above article which appeared in the July-September 1980 issue of the Journal of Environmental Quality. The equation should read:

\[ \ln \left( \frac{k_2}{k_1} \right) = \phi (T_2 - T_1) \]

where \( \phi = \frac{(Ea/R)}{(T_2 T_1)} \).