D4. Nonpoint Source Nitrogen Loading, Sources, and Pathways for Minnesota Surface Waters

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Introduction

Nonpoint source nitrogen (N) loading to Minnesota Surface Waters was estimated for the primary N sources, including cropland, urban/suburban nonpoint sources, forested areas, and feedlots. Pathways for cropland sources were divided into three parts: 1) cropland runoff, 2) tile drainage, and 3) leaching to groundwater which subsequently flows into surface waters. Nitrogen from these sources was estimated for average, wet, and dry precipitation years at the watershed, major basin, and statewide scales.

A cropland soil N balance was also conducted as a separate and distinct element of this study. The cropland balance provided estimates of the N inputs and outputs to the cropland soil. The balance was not used to calculate cropland N sources or delivery to surface waters. Yet certain elements of the N balance, such as fertilizer and manure additions, were also used to estimate N losses to surface waters.

Project goals

- Assess soil N budgets (N additions to soil and losses from soils) for combinations of soils, climates and land uses representative of the most common Minnesota conditions.
- Assess N contributions to Minnesota rivers from each of: a) the primary land use sources (excluding point source municipal and industrial), and b) the primary hydrologic pathways.

Materials and methods

Study area

Minnesota has diverse climatic factors, land use, land cover, soil and geologic materials, and landscapes. In addition, the density of permanent streams, drainage ditches, and lakes varies across the state. This diversity affects water quality and water quantity. It also affects the types of crop and animal production systems and their associated suite of management practices. Mean annual precipitation varies from less than 20 inches in the northwestern part of the state to over 34 inches in the southeastern part of the state. Soil parent material and geologic materials at the land surface include alluvial, outwash, peat, glacial moraine, glacial till, and lacustrine materials. The soils and their associated landscapes range from flat to steep in slope, and from poorly drained to well drained. This combination helps determine the potential for runoff, leaching and the likelihood of artificial drainage and losses of nitrate-N to surface waters.

The diverse range in Minnesota climate, soil and landscapes, and land use/land cover can be broadly described using the concept of agroecoregions (Figure 1), which is defined further in Brezonic et al. (1999). Agroecoregions are units having relatively homogeneous climate, soil and landscapes, and land use/land cover. Agroecoregions can be associated with a specific set of soil and water resource concerns, and with a specific set of management practices to minimize the impact of land use activities on soil and water resource quality.
Land use in Minnesota includes urban areas, forest, forested wetlands, wetlands, agriculture, and barren rock. Land use associations include agriculture, forest, agriculture-forest, forest-wetlands-agriculture, forest-wetlands, and urban-agriculture. Agricultural uses include both crop and animal production. Crop production is diverse, major crops considered for the study include corn, soybeans, wheat, hay, potatoes, sugar beets, oats, and barley. The main cropping production systems include corn-soybeans, corn-soybeans-hay, corn-hay, wheat-hay-mixed, wheat-soybeans-mixed, and hay. Animal production systems include cattle-hogs, cattle-hogs-turkeys-chickens, cattle-poultry, and hogs-cattle, and cattle. If not properly managed, N contained in the manure produced by these animals may pollute the atmosphere, or surface and groundwaters.

Figure 1. Minnesota’s agroecoregions with basin and major watershed boundaries.

Data Sources: The University of Minnesota with support from the Minnesota Department of Agriculture
Methods overview

Two separate methods were used for two distinct purposes within the scope of this project. First, a statewide cropland N budget (or balance) was developed so that specific inputs and outputs to the cropping system could be estimated. Since inputs should roughly equal outputs, comparing the sum of the inputs with the sum of the outputs provides one way to check the estimates. One of the N outputs is an estimate of the amount of cropland N inputs which reach surface waters. This output was determined through the second project objective, and then was also used to complete the N balance.

The second objective was to determine the amount of N that reaches surface waters from all nonpoint sources, including cropland, urban/suburban, septic systems, and forest. The goal was to also break down cropland sources to waters into the three major pathways, tile drainage, groundwater and surface runoff. While some of the information from the cropland N budget was used for the nonpoint sources estimates, most of the information came from information sources separate from the N balance study. The specific methods for each of these two project objectives are described below.

1: Cropland nitrogen balance methods

The approach used to carry out this mass-balance of the state of Minnesota was to compile the information necessary for each component to the balance individually, and then assemble all of these components in a format that would be both easy to interpret, as well as accessible, for changes in the future when updated information becomes available. The fluxes included were chosen based on their implicit importance within the boundaries of the study area, as well as the availability of sufficient information and methods to confidently include them.

An N balance was estimated for the cultivated cropland component of this study. Forest, urban/suburban and septic system inputs and outputs were not considered in the N balance, but N export to surface waters for these sources was considered separately. Ideally, N inputs and outputs should be equal in the N balance. The inputs represented in this balance include mineralization minus immobilization (net mineralization), symbiotic and non-symbiotic fixation, inorganic fertilizer, atmospheric deposition, animal feed, and planted seeds. The outputs include tile drainage and runoff, denitrification, leaching to groundwater, crop senescence, fertilizer volatilization, crop removal, milk, eggs, and animal slaughter. The two fluxes considered internal to this balance are a portion of the harvested crops that are fed to livestock and the livestock manure that is returned to the fields (Figure 2).

Figure 2. Nitrogen balance used to evaluate the N use efficiency in the state of Minnesota (extracted from Stuewe, 2006).
Area in various land uses (forest, urban) and major crops were determined for each agroecoregion based on 2006 National Land Cover Database land use coverages. (NLCD, 2006).

Harvested crop area for each agroecoregion was determined using a five year average (2005-2009) of data from the National Agricultural Statistics Service (USDA-NASS, 2011) for the following crops: corn, soybeans, spring wheat, barley, oats, sugarbeet, potatoes, alfalfa. For corn silage and other hay, a weighted average was reported (USDA-NASS-CDL, 2009). Total cultivated area was the sum of all harvested crop area.

Nitrogen inputs

The estimated N balance and specific inputs and outputs were not used to estimate nitrogen loads to surface waters, except that fertilizer and manure inputs were used for certain elements of the cropland source pathway estimates. The balance provides a framework for understanding the cropland soil N sources and processes, but is not used to attribute N contributions to surface waters.

Planted seeds

Corn and soybean growers in the Midwest annually purchased seeds from seed dealers. This annual purchase represents an input of N into the system that needs to be estimated and included in the N balance. It was assumed that 0.34 kg N ha⁻¹ and 4.5 kg N ha⁻¹ (0.3 lb N ac⁻¹ and 4.0 lb N ac⁻¹) are contained in the seeds planted for corn and soybeans, respectively (Meisinger and Randall, 1991). For barley, oats, spring wheat and potatoes, planted N seed (lb) was calculated as following:

\[ N \text{ seeds (lbs N/ac)} = \text{seeding rate/ac} \times \frac{N \text{ content (lb N/ac)}}{100} \]

Planted seeding rates were 80, 80 and 133 lb ac⁻¹ for barley, oat and spring wheat (MAES, 2006). Nitrogen content was 1.86%, 3.5% and 2.6% for barley, oat, and spring wheat (Sims et al. 2002, Pan and Hopkins, 1991, Hofstetter, 1988). Thus, the estimated planted N seed was 2, 2.3 and 3.9 pounds ac⁻¹ for barley, oat, and spring wheat respectively. Nitrogen content in the potatoes was estimated at 23.4 pounds N ac⁻¹, using N content of 1.608% and 1.648% for tubers and vines (Rosen et al. 1999). Estimates of the N contained in alfalfa seeds, other hay and sugar beets were not included. Total planted N seed was calculated as:

\[ \text{Planted N seed (lb)} = \sum (\text{harvested crop area (ac)} \times \text{N seeds (lb ac}^{-1}) \]

Atmospheric deposition

Atmospheric N deposition comprises both wet and dry depositional processes, and includes all oxidized and reduced forms of N, including NO₃ and NH₄. Total atmospheric deposition rate was area-weighted for each agroecoregion (EPA, 2011; Byun and Schere, 2006), as described in more detail in MPCA, 2012. Atmospheric deposition represents an average over many climatic years.

Symbiotic nitrogen fixation

Symbiotic and non-symbiotic fixations were included in this balance. In symbiotic fixation, specialized root-nodule bacteria attached to leguminous plants and converts N₂-N from the atmosphere into N compounds that are taken up by the plant (Graham, 1998). Non-symbiotic fixation is essentially the same process, but the soil bacteria carrying out the process are free living and unattached to a leguminous host plant (Meisinger and Randall, 1991).
The symbiotic fixation rates used in this balance are reported in Table 1. The total land area over which these fixation rates were applied includes the harvested acres of soybean, alfalfa, and grass/legume crops (USDA-NASS, 2011, USDA-NASS-CDL, 2009).

The non-symbiotic N fixation estimates made for this balance are based on Meisinger and Randall (1991) and a rate of 2.2 kg N ha\(^{-1}\) (2 lb N ac\(^{-1}\)) was applied to all of the harvested cropland area.

### Table 1. Symbiotic fixation rates estimated for soybean, alfalfa and grass/legume.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Symbiotic fixation rates</th>
<th>Symbiotic fixation rates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soybean(^{\dagger})</td>
<td>60.5 kg N ha(^{-1}) yr(^{-1})</td>
<td>50 lb N ac(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Alfalfa(^{\dagger})</td>
<td>22.86 kg N ton(^{-1})</td>
<td>50.4 lb N ton(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Grass/legume(^{\dagger})</td>
<td>19.7 kg N ton(^{-1})</td>
<td>43.5 lb N ton(^{-1}) yr(^{-1})</td>
</tr>
</tbody>
</table>

\(^{\dagger}\)Source: Plants Database, USDA (http://npk.nrcs.usda.gov) reported by MDA, 2005: Reports, publications and fact sheets.

\(^{\ddagger}\)Russelle, M (pers. comm.)

### Mineralization

Mineralizable N was estimated using the same approach presented by Burkart and James (1999a), and reported by Stuewe (2006), with a small modification in the soil elemental N content (from 3.0% to 3.2%). The following equation was used:

\[
Nm = 1000 \times \frac{Db \times Om}{100} \times Vs \times Ne \times Np
\]

where:
- \(Nm\) = Mineralizable nitrogen (lb ac\(^{-1}\))
- \(Db\) = Bulk density of specific soil (Mg/m\(^3\)) (constant=1.471 Mg m\(^{-3}\))
- \(Om\) = Organic matter content of soil (%)
- \(Vs\) = Volume of 30 cm thick soil in one hectare (constant = 3,000 m\(^3\) ha\(^{-1}\))
- \(Ne\) = Elemental nitrogen fraction of soil organic matter (constant = 3.2%)
- \(Np\) = Annual mineralizable portion of soil organic nitrogen (constant = 2%)

The percent organic matter used in these calculations is from SSURGO mapping unit values (USDA NRCS, 1995). Percent organic matter was estimated only in cultivated lands (NLCD, 2006). High anomalous values were removed to maintain data integrity (eg. Anoka Sand Plain average went from 8.4% with anomalous values to 2.02%, a much more appropriate value based on Delin et al. 1994). The bulk density assumed across the entire study area is the commonly used estimate of 1.471 Mg m\(^{-3}\) (2,000,000 pounds ac-1-6 inches deep). The volume of soils considered was the top 30 cm (11.8 inches) of soil, equivalent to 3,000 m\(^3\) ha\(^{-1}\) (Burkart and James, 1999a). The annual mineralizable portion of the soil organic N used was 2% (Schepers and Mosier, 1991).

### Immobilization

The amount of immobilized N (converted from inorganic N to organic N by micro-organisms or plants) was estimated after all volatilization losses were accounted for both the inorganic fertilizer and the manure applied in the study area. This amount of N immobilized should not be considered a complete loss from the system, and should be viewed as N held in the soil organic matter pool, unavailable for immediate plant uptake during the first year of application, but possibly available in subsequent years (Burkart and James, 1999b).
The immobilization rate for all forms of inorganic fertilizer was assumed to be 40% (Burkart and James, 1999a). The immobilization rates for each type of livestock manure are presented in Table 2 (Burkart and James, 1999b, adapted from Elliot and Swanson, 1976; Schepers and Mosier, 1991; reported by Stuewe, 2006).

Table 2. The N immobilization rates assumed for each type of livestock manure applied to cropland (reported by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Animal type</th>
<th>% N immobilized</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beef Cows</td>
<td>70%</td>
</tr>
<tr>
<td>Milk Cows</td>
<td>60%</td>
</tr>
<tr>
<td>Hogs</td>
<td>10%</td>
</tr>
<tr>
<td>Chickens (broilers)</td>
<td>25%</td>
</tr>
<tr>
<td>Turkeys</td>
<td>25%</td>
</tr>
</tbody>
</table>

Inorganic fertilizer

The total amount of inorganic N fertilizer considered in this balance was calculated based on the N fertilizer rate and the cultivated area of each crop.

For crops other than corn, a constant rate was used for all agroecoregions (Table 3). Soybean fertilizer rates were adjusted from 20 to 3 pounds ac\(^{-1}\) since only 15% of soybeans fields are fertilized (NASS, 2002-2004-2006-2008).

Fertilizer N rates for corn in each agroecoregion were determined based on county-level farmer surveys (Figure 3) (Bierman et al., 2011). Nitrogen rates for corn were based on non-manure fields; however rates were adjusted according to manure credit calculations. The Minnesota Pollution Control Agency (MPCA) feedlot registration database was used to determine animal units of different species. These registration numbers are often reported on the high-end of an operation’s potential animal capacity so as to not limit the operation. Since actual animal numbers are often less than reported in this database, animal numbers were corrected downward based on surveyed values from the Minnesota Department of Agriculture (MDA). National Agricultural Statistics Service (NASS) animal statistics were used to cross check this method, and confirm that it accurately represented animal numbers.

Using these adjusted feedlot numbers, available N from manure was calculated using two different methods (Midwest Plan Service MWPS-18 2004; University of Minnesota Extension Service 2001). It was assumed that 50% to 70% of calculated first year N credits would be taken, with no second year credits considered in this calculation, and also 59% of poultry manure would be burned.

Total amounts of N fertilizer were initially estimated as the product of fertilizer N rate times area of each crop planted based on remote sensing data collected for the 2009 CDL. This amount was compared with statewide estimates of N fertilizer sales, excluding sales in urban areas, and found to be slightly low. Initial estimates of corn acres planted were then adjusted upwards based on improved estimates of corn acreage using statistical survey data from NASS. The improved corn acres planted estimate was then multiplied by the surveyed N rates applied to corn (Bierman et al., 2011) and credits for land applied manure were then subtracted to obtain the total amounts of N fertilizer applied to corn.
Table 3. Nitrogen fertilizer rates for each crop

<table>
<thead>
<tr>
<th>Crop</th>
<th>Fertilizer N rate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>lbs ac⁻¹</td>
</tr>
<tr>
<td>Soybeans</td>
<td>3</td>
</tr>
<tr>
<td>Spring wheat</td>
<td>107²</td>
</tr>
<tr>
<td>Barley</td>
<td>66²</td>
</tr>
<tr>
<td>Oats</td>
<td>48³</td>
</tr>
<tr>
<td>Sugarbeet</td>
<td>83⁴</td>
</tr>
<tr>
<td>Potatoes</td>
<td>195⁵</td>
</tr>
<tr>
<td>Alfalfa</td>
<td>10⁶</td>
</tr>
<tr>
<td>Other Hay</td>
<td>10⁶</td>
</tr>
</tbody>
</table>

¹MDA
²NASS, 2003
³NASS, 2005
⁴NASS, 2001 and U of MN recommendations
⁵Weighted average based on U of MN recommendations for irrigated and non-irrigated potatoes
⁶U of MN recommendations

Nitrogen outputs

Crop harvest (grain nitrogen removal)

The total amount of N removed with harvested crops was calculated based on 5- year average yield data and N content in the grain. Average yield data (2005-2009) for the following crops: corn, soybeans, spring wheat, barley, oats, and sugar beets were obtained from USDA-NASS (2011). Potato yield data were provided by Carl Rosen (pers. comm. March 2011). Weighted yield average was used for corn silage and other hay to estimate grain N removal (USDA-NASS-CDL, 2009). The percentage of N in grain and stover for each crop is presented in Table 4.

Figure 3. County N application rates for corn obtained from farmer surveys (Bierman et al. 2011).
Table 4. Percentage of N in grain and stover for each crop used to estimate the grain and stover N removal.

<table>
<thead>
<tr>
<th>Crop</th>
<th>N grain (%)</th>
<th>N stover (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn ^1</td>
<td>1.2</td>
<td>0.70</td>
</tr>
<tr>
<td>Corn silage ^2</td>
<td>6.34</td>
<td>1.18</td>
</tr>
<tr>
<td>Soybeans ^3</td>
<td>2.62</td>
<td>0.55</td>
</tr>
<tr>
<td>Spring wheat ^4</td>
<td>1.86</td>
<td>0.36</td>
</tr>
<tr>
<td>Barley ^5</td>
<td>3.5</td>
<td>0.36</td>
</tr>
<tr>
<td>Oats ^6</td>
<td>1.86</td>
<td>2.28</td>
</tr>
<tr>
<td>Sugarbeet ^7</td>
<td>1.61</td>
<td>1.65</td>
</tr>
<tr>
<td>Potatoes ^8</td>
<td>3.1</td>
<td></td>
</tr>
<tr>
<td>Alfalfa ^9</td>
<td>2.34</td>
<td></td>
</tr>
</tbody>
</table>

The total amount of grain and N removal was calculated as follows:

\[
\text{Total amount of grain N removal (lb)} = N \text{ removal rate (lb ac}^{-1}\text{)} \times \text{Number of acres for each crop}
\]

\[
\text{Total amount of stover removal (lb)} = N \text{ removal rate (lb ac}^{-1}\text{)} \times \text{Number of acres for corn silage}
\]

**Crop senescence**

During senescence, plants will volatilize N into the atmosphere, primarily as NH₃, from the maturing vegetation (Wetselaar and Farquhar, 1980). The rates of N senescence for corn, soybean, alfalfa, and small grains used in this balance are presented in Table 5.
Table 5. Crop N senescence rates estimated for major crops grown within the study area (Burkart and James, 1999a, reported by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Crop</th>
<th>Senescence rate kg N ha(^{-1}) yr(^{-1})</th>
<th>Senescence rate lb N ac(^{-1}) yr(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn</td>
<td>50</td>
<td>44.6</td>
</tr>
<tr>
<td>Soybeans</td>
<td>45</td>
<td>40.1</td>
</tr>
<tr>
<td>Alfalfa</td>
<td>22</td>
<td>19.63</td>
</tr>
<tr>
<td>Small grains†</td>
<td>35</td>
<td>31.2</td>
</tr>
</tbody>
</table>

† Small grains include spring wheat, barley and oats (Burkart and James, 1999a).

**Volatilization of stored manure**

Volatilization of stored manure decreases the amount of N subsequently available for land application. Manure N volatilization rates during storage for each animal species estimated for this study are presented in Table 6 (Purdue, 2001).

Table 6. Manure N volatilization rates during storage used in this balance.

<table>
<thead>
<tr>
<th>Fertilizer type</th>
<th>% N loss</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beef</td>
<td>35%</td>
</tr>
<tr>
<td>Dairy</td>
<td>20%</td>
</tr>
<tr>
<td>Swine</td>
<td>20%</td>
</tr>
<tr>
<td>Chickens</td>
<td>25%</td>
</tr>
<tr>
<td>Turkeys</td>
<td>25%</td>
</tr>
</tbody>
</table>

**Volatilization of land applied fertilizer and manure**

Volatilization losses of N from organic and inorganic fertilizers primarily occur as NH3 during application (Burkart and James, 1999a; Mosier et al., 1998). In this balance, different volatilization rates were assumed to each type of inorganic fertilizer and animal manure applied.

Nitrogen volatilization losses during the application of synthetic fertilizers are based on the estimates described by Stuewe (2006) (Table 7).

Table 7. Percentage of total sales and N volatilization rates for each inorganic fertilizer applied in the study area.

<table>
<thead>
<tr>
<th>Fertilizer type</th>
<th>% of total sold†</th>
<th>% N loss‡</th>
</tr>
</thead>
<tbody>
<tr>
<td>Anhydrous Ammonia (82-0-0)</td>
<td>45.9%</td>
<td>2%</td>
</tr>
<tr>
<td>Urea (46-0-0)</td>
<td>44.8%</td>
<td>5%</td>
</tr>
<tr>
<td>UAN (28-0-0 &amp; 32-0-0)</td>
<td>4.9%</td>
<td>5%</td>
</tr>
<tr>
<td>Custom Blends (all other blends)</td>
<td>4.4%</td>
<td>4%</td>
</tr>
</tbody>
</table>

†Bierman et al. 2011. ‡Stuewe (2006)

Volatilized N losses during the application of livestock manure were estimated for each type of animal manure (Table 8). The manure N considered available for volatilization losses during application is the amount remaining after all storage and burned losses were accounted.
Table 8. Nitrogen volatilization rates during manure application to cropland in the study area used in this balance (Reported by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Fertilizer type</th>
<th>% N loss</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beef</td>
<td>21%</td>
<td>Schmitt, 1999</td>
</tr>
<tr>
<td>Dairy</td>
<td>10%</td>
<td>Written comm. w/ Dr. Gyles Randal &amp; verbal confirmation by Dr. David Mulla (2005)</td>
</tr>
<tr>
<td>Swine</td>
<td>10%</td>
<td>Written comm. w/ Dr. Gyles Randal &amp; verbal confirmation by Dr. David Mulla (2005)</td>
</tr>
<tr>
<td>Chickens</td>
<td>18%</td>
<td>Schmitt, 1999</td>
</tr>
<tr>
<td>Turkeys</td>
<td>18%</td>
<td>Schmitt, 1999</td>
</tr>
</tbody>
</table>

**Denitrification**

Soil denitrification rates were assigned according to soil drainage and other soil characteristics in each agroecoregion. Denitrification rates for each agroecoregion with the described soil characteristics are presented in Table 9. Most tile drained lands were assumed to have half the rate of denitrification of untiled lands in each agroecoregion (Table 9).

Table 9. Denitrification percentages estimated for applied and in situ forms of soil N used in the N balance.

<table>
<thead>
<tr>
<th></th>
<th>No-Tile</th>
<th>Tile</th>
</tr>
</thead>
<tbody>
<tr>
<td>Excessive to well drained (sandy, loam, muck)</td>
<td>3</td>
<td>1</td>
</tr>
<tr>
<td>Somewhat poorly drained (loam)</td>
<td>20</td>
<td>10</td>
</tr>
<tr>
<td>Poorly and very poorly drained</td>
<td>30</td>
<td>15</td>
</tr>
</tbody>
</table>

1 Percentage estimated from Venterea (2011)
2 Percentage estimated from Meisinger and Randall (1991), reported by Stuewe (2006)

**Total denitrification**

Denitrification rates were calculated separately for the amount of N in land applied livestock manure and inorganic fertilizer, in N deposition and in the mineralizable N from soil organic matter. The amount of N in applied manure and inorganic fertilizer available for denitrification is the amount remaining after all volatilization and immobilization losses have been considered. The calculations carried out for each of these sources were combined to come up with an overall estimate of the N escaping from the study area through denitrification. For the N balance, denitrification occurs only at the field scale. To estimate N loadings from groundwater discharge, an additional denitrification factor was applied as discussed below in the section 2 “Methods: Nonpoint Source Nitrogen Loadings to Surface Waters”.

**Cropland nitrogen leaching losses**

A literature review was conducted to determine the N leaching rate for each agroecoregion. Details of estimated N leaching rates are presented in Section 2. Total nitrogen leaching as an output in the agricultural N balance does not account for denitrification losses that occur beyond the edge of field as groundwater travels towards and is discharged to streams.
Cropland nitrogen losses in tile drainage
Total N losses in tile drainage were calculated for dry, average and wet conditions based on growing season precipitation data and N rate applied. Details of tile drainage calculations are presented in Section 2.

Cropland nitrogen losses in surface runoff
Nitrogen losses in surface runoff were calculated as a function of runoff volume and N concentration for each agroecoregion. Details of calculations are presented in Section 2.

Nitrogen exported in milk
Nitrogen exported in milk is based on an assumed average crude protein content of 3.1% and the assumption that 16% of crude protein is N (Ferguson, 2001, reported by Stuewe, 2006). Considering these assumptions, the N content in the milk used in this balance was 0.496%. The quantity of milk considered in these applications is the total amount of milk reported to have been produced within each of agroecoregion (USDA-NASS, 2011, NASS county data weighted average 2005-2009).

Nitrogen exported in eggs
The N content assumed for each egg is 1.00 gram, based on information from the Human Nutrition Information Service (USDA, 1989). The amount of eggs produced in each agroecoregion was estimated assuming 230 eggs per year per layer (NASS, 2010). For this balance, it was assumed that all eggs produced within the study areas (agroecoregions) are sold to customers outside of this area.

Nitrogen exported in meat
The percentage of livestock slaughtered for each agroecoregion was estimated based on the total slaughter counts for the state of Minnesota (Table 10). Total slaughter counts were determined based on the MPCA feedlot registration data developed from 2006 to 2010, which represents the maximum livestock numbers in the feedlot during that time period. Data were adjusted for over-reporting feedlot data using a correction factor of 90% for dairy and swine, 70% for beef, 80% for turkey and 85% for chicken (Wayne Cords, personal communication with D. Wall, MPCA).

The slaughter-weights were estimated for each type of livestock based on the percentage of slaughter count for each agroecoregion and the total slaughter weight for the state of Minnesota (Table 11) (NASS, 2011a, b). Estimates of the live weight percentage of N in each animal type sent to be slaughtered are presented in Table 1.10 (Powers and Van Horn, 2001, reported by Stuewe, 2006). The amount of N contained in livestock sent to be slaughtered is calculated based on the live weight percentage of N and the slaughter-weights for each type of animal.
Table 10. The total slaughter-weights and counts used to estimate the total amount of N removed from the state of Minnesota within slaughtered animals.

<table>
<thead>
<tr>
<th>Animal Type</th>
<th>State-Total Slaughter Count</th>
<th>State-Total Slaughter Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cattle (average of dairy &amp; beef)</td>
<td>2530243.4</td>
<td>127065000</td>
</tr>
<tr>
<td>Hogs</td>
<td>938839.9</td>
<td>269177200</td>
</tr>
<tr>
<td>Chicken (typical 9wk broiler)</td>
<td>13010263.2</td>
<td>248966000</td>
</tr>
<tr>
<td>Turkey (2002 12 month average)</td>
<td>21177624.8</td>
<td>112713900</td>
</tr>
</tbody>
</table>

1 For total slaughter count: MPCA Feedlot registration data (2006-2010) with corrections for over-reporting of feedlot data

Table 11. Whole body live weight percent N content used to estimate the N in livestock sent to be slaughtered (Powers and Van Horn, 2001; reported by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Animal Type</th>
<th>Whole Body % N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cattle (average of dairy &amp; beef)</td>
<td>1.40%</td>
</tr>
<tr>
<td>Hogs</td>
<td>2.32%</td>
</tr>
<tr>
<td>Chicken (average of hens &amp; broilers)</td>
<td>2.40%</td>
</tr>
<tr>
<td>Turkey</td>
<td>2.10%</td>
</tr>
</tbody>
</table>

Nitrogen cycling between crop and animal agriculture

Animal manure

The manure N production rates applied in this balance are shown in Table 12. The amount of manure produced for each animal category was calculated using:

\[
\text{Livestock manure production (lb yr}^{-1}) = \# \text{ of slaughter livestock } \times \text{ manure N rate production (lb day}^{-1}) \times 365 \text{ days year}^{-1}
\]

Approximately 59% of chicken and turkey manure is assumed to be burned each year based on MPCA and Fibrominn records (personal communication with J. Jones, 2010), and only 41% will be available for land application.

Manure N volatilization rates during storage for each animal species were reported in Table 6. Volatilized N losses during the application of livestock manure were presented in Table 8. The manure N considered available for volatilization losses during application is the amount remaining after all storage and incineration losses were accounted for.

The available N in manure after land application is affected by soil processes, such as immobilization by soil microorganisms. For this balance, it was assumed that 50% (for beef, chicken and turkey), 55% (dairy), and 70% (hogs) of N will be available in the first year, and 25% in the second year after the initial manure application.
Table 12. Livestock manure N production rates and animal counts for each animal category used to estimate the manure N produced by the livestock in this balance

<table>
<thead>
<tr>
<th>Animal Type</th>
<th>Animal counts</th>
<th>N rates(^\pm)</th>
<th>State total ‡</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>#</td>
<td>lb N day(^{-1})</td>
<td></td>
</tr>
<tr>
<td>Beef</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bull</td>
<td>1213657</td>
<td>0.350</td>
<td></td>
</tr>
<tr>
<td>Cow</td>
<td>787172</td>
<td>0.350</td>
<td></td>
</tr>
<tr>
<td>Calf finish</td>
<td>225953</td>
<td>0.270</td>
<td></td>
</tr>
<tr>
<td>Calf</td>
<td>414801</td>
<td>0.270</td>
<td></td>
</tr>
<tr>
<td>Dairy</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cow-lactating</td>
<td>1084383</td>
<td>0.720</td>
<td></td>
</tr>
<tr>
<td>Cow-dry</td>
<td></td>
<td>0.300</td>
<td></td>
</tr>
<tr>
<td>Calf</td>
<td>343690</td>
<td>0.060</td>
<td></td>
</tr>
<tr>
<td>Heifer/steer</td>
<td>468707</td>
<td>0.230</td>
<td></td>
</tr>
<tr>
<td>Hog</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hogboar</td>
<td>156883</td>
<td>0.04</td>
<td></td>
</tr>
<tr>
<td>Sow-farrow finish</td>
<td>1246290</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td>Farrow feed</td>
<td>417560</td>
<td>0.02</td>
<td></td>
</tr>
<tr>
<td>Chicken</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Broiler big</td>
<td>6035232</td>
<td>0.002</td>
<td></td>
</tr>
<tr>
<td>Broiler little</td>
<td>17036814</td>
<td>0.0011(^†)</td>
<td></td>
</tr>
<tr>
<td>Layer big</td>
<td>343039</td>
<td>0.003</td>
<td></td>
</tr>
<tr>
<td>Layer little</td>
<td>25839825</td>
<td>0.0013(^†)</td>
<td></td>
</tr>
<tr>
<td>Turkey</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Big</td>
<td>23073859</td>
<td>0.009</td>
<td></td>
</tr>
<tr>
<td>Little</td>
<td>13754302</td>
<td>0.0047</td>
<td></td>
</tr>
</tbody>
</table>

\(^†\)Data not available. These values are half of the big broiler and big layer N production rates.
\(^‡\)MPCA Feedlot registration data with corrections for over-reporting of feedlot data.
\(^±\) MWPS (1993).

Harvested crop used for animal feed

Corn and soybean grain are used for animal feed in beef, cattle, swine, and poultry production. Also, corn silage, alfalfa and other hay are fed mainly to beef and dairy cows. Coefficients for harvested corn and soybean use in Minnesota were obtained from the Department of Agriculture (Ye, 2010; Ye, 2009a; Ye, 2009b; MDA, 2010) and are reported in Table 13.
Table 13. Percentage of corn and soybean uses in Minnesota.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Use</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Export</td>
<td></td>
<td>42</td>
</tr>
<tr>
<td>Ethanol use</td>
<td></td>
<td>34</td>
</tr>
<tr>
<td>Feed use</td>
<td></td>
<td>17</td>
</tr>
<tr>
<td>Residual use</td>
<td></td>
<td>7</td>
</tr>
<tr>
<td>Soybean</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Export</td>
<td></td>
<td>40</td>
</tr>
<tr>
<td>Crush for feed</td>
<td></td>
<td>56</td>
</tr>
<tr>
<td>Seed and Residual</td>
<td></td>
<td>4</td>
</tr>
</tbody>
</table>

In summary, 17% and 56% of the harvested corn and soybean are being used for feeding animals in Minnesota, respectively. Approximately 25% of the soybean meal from crush is used for feed (75% is exported). The percentages used for each animal are presented in Table 14.

Ethanol production comprised 34% of the harvested corn (Table 13). During ethanol production, starch is extracted from corn grain, and the remaining nutrients are converted to by-products that can be used for animal feed, including Dried Distiller Grains (DDGs). For the N balance, it was assumed that 14.5 pounds of DDGs were produced for each bushel of corn used in the ethanol process with a crude protein (CP) content of 30% and 16% N in CP. Also, 50% of DDGs were exported out of state.

Table 14. Percentage of feed use from corn and soybean for different animal categories in Minnesota.

<table>
<thead>
<tr>
<th>Feed use</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn</td>
<td></td>
</tr>
<tr>
<td>Beef</td>
<td>15</td>
</tr>
<tr>
<td>Hogs</td>
<td>46</td>
</tr>
<tr>
<td>Dairy</td>
<td>21</td>
</tr>
<tr>
<td>Poultry</td>
<td>17</td>
</tr>
<tr>
<td>Others</td>
<td>1</td>
</tr>
<tr>
<td>Soybean</td>
<td></td>
</tr>
<tr>
<td>Beef cattle</td>
<td>9</td>
</tr>
<tr>
<td>Hogs and pigs</td>
<td>41</td>
</tr>
<tr>
<td>Dairy (milk cows)</td>
<td>15</td>
</tr>
<tr>
<td>Poultry</td>
<td>35</td>
</tr>
<tr>
<td>Others</td>
<td>0.4</td>
</tr>
</tbody>
</table>
Livestock feed

Feed N intake for each category of livestock was determined based on recommended nutrient requirements for each livestock species. All the assumptions used to estimate N consumption rate for each animal species are presented in detail in Stuewe (2006). Feed N intake was summed over all species and categories of animals in order to determine whether or not enough harvested crop used for animal feed was available to meet livestock nutritional requirements. The result of this analysis was that harvested crop used for animal feed was sufficient, and consequently, no additional nutritional supplements were added to the overall N balance.

The animal population numbers used for these estimates were obtained from the MPCA.

Swine feed

The population estimates used for swine in these calculations were reported in two categories, “hogs” and “nursery hogs. The consumption rates and crude protein requirements used for both the “hogs” and “nursery hogs” are presented in Table 15 (NAS, 1998, reported by Stuewe, 2006). The N consumption rate was calculated as follows:

\[ \text{N consumption (lbs yr}^{-1} \text{)} = (\text{N}^{\circ} \text{ Hogs} + \text{N}^{\circ} \text{ nursery hogs}) \times \text{N consumption rate} \times 365 \]

Table 15. Feed consumption rates and crude protein requirements for "hogs" and "nursery hogs" used to estimate the feed N consumed annually by these animals (NAS, 1998, cited by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Livestock</th>
<th>Feed Consumption Rate (kg feed day(^{-1}))</th>
<th>Crude Protein (CP)</th>
<th>Nitrogen in CP</th>
<th>N Consumption Rate (kg N day(^{-1}))</th>
<th>N Consumption Rate (lb N day(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>&quot;Hogs&quot;</td>
<td>2.502</td>
<td>15.6%</td>
<td>16%</td>
<td>0.063</td>
<td>0.139</td>
</tr>
<tr>
<td>&quot;Nursery Hogs&quot;</td>
<td>0.750</td>
<td>22.3%</td>
<td>16%</td>
<td>0.027</td>
<td>0.060</td>
</tr>
</tbody>
</table>

Beef cattle

The population estimates acquired for beef cattle within the study area were reported in four categories, including “beef heifers”, “feedlot beef”, “calf finish”, and “beef calves”. The consumption rates and crude protein requirements used for each category are presented in Table 16 (NAS, 1998, NAS, 2000; reported by Stuewe, 2006).

Table 16. Feed consumption rates and metabolizable protein requirements for "beef heifers", “feedlot beef”, “calf finish”, and “beef calves” used to estimate the feed N consumed annually by these animals (NAS, 1998, cited by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Livestock</th>
<th>MP Consumption Rate (kg MP day(^{-1}))</th>
<th>Conversion Factor to CP</th>
<th>Nitrogen in CP</th>
<th>N Consumption Rate (kg N day(^{-1}))</th>
<th>N Consumption Rate (lb N day(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>&quot;Beef Heifers&quot;</td>
<td>0.624</td>
<td>divided by 0.67</td>
<td>16%</td>
<td>0.149</td>
<td>0.328</td>
</tr>
<tr>
<td>&quot;Feedlot Beef&quot;</td>
<td>0.665</td>
<td>divided by 0.67</td>
<td>16%</td>
<td>0.159</td>
<td>0.351</td>
</tr>
<tr>
<td>&quot;Calf finish&quot;</td>
<td></td>
<td></td>
<td>0.159</td>
<td>0.351</td>
<td></td>
</tr>
<tr>
<td>&quot;Beef Calves&quot;</td>
<td></td>
<td></td>
<td>0.027</td>
<td>0.060</td>
<td></td>
</tr>
</tbody>
</table>
**Dairy cattle**

The population estimates obtained for dairy cattle within the study area were reported in four categories including “lactating dairy”, “dry dairy”, “young dairy steers”, and “dairy calves”. The consumption rates and crude protein requirements used for each category are presented in Table 17 (Linn, 2004; MWPS, 2003; NAS, 2001; reported by Stuewe, 2006).

Table 17. Feed consumption rates and crude protein requirements for "lactating dairy", “dry dairy”, “young dairy steers”, and “dairy calves” used to estimate the feed N consumed annually by these animals (Linn, 2004; MWPS, 2003; NAS, 2001, cited by Stuewe, 2006).

<table>
<thead>
<tr>
<th>Livestock</th>
<th>Feed Consumption Rate</th>
<th>CP in Feed</th>
<th>N in Feed</th>
<th>N Consumption Rate</th>
<th>N Consumption Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>kg day(^{-1})</td>
<td>%</td>
<td>%</td>
<td>kg N day(^{-1})</td>
<td>lbs N day(^{-1})</td>
</tr>
<tr>
<td>“Lactating Dairy”</td>
<td>20.4</td>
<td>16%</td>
<td>16%</td>
<td>0.523</td>
<td>1.153</td>
</tr>
<tr>
<td>“Dry Dairy”</td>
<td>13.6</td>
<td>13%</td>
<td>16%</td>
<td>0.283</td>
<td>0.624</td>
</tr>
<tr>
<td>&quot;Young Dairy Steers&quot;</td>
<td>8.8</td>
<td>14.2%</td>
<td>16%</td>
<td>0.200</td>
<td>0.441</td>
</tr>
<tr>
<td>“Dairy Calves”</td>
<td>4.2</td>
<td>16.9%</td>
<td>16%</td>
<td>0.114</td>
<td>0.251</td>
</tr>
</tbody>
</table>

**Poultry**

The population estimates for turkeys within the study area are reported in only one category, “turkeys”. The population estimates reported for chickens within the study area are reported in two categories: “broilers” and “layers”. Nitrogen consumption rate for turkeys and chickens are presented in Table 18 (NAS, 1994; reported by Stuewe, 2006).

Table 18. Feed consumption rates and crude protein requirements for "turkeys" and "chickens" used to estimate the feed N consumed annually by this poultry (NAS, 1994, cited by Stuewe).

<table>
<thead>
<tr>
<th>Livestock</th>
<th>Feed Consumption Rate</th>
<th>Crude Protein (CP)</th>
<th>N in CP</th>
<th>N Consumption Rate</th>
<th>N Consumption Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>kg feed day(^{-1})</td>
<td>%</td>
<td>%</td>
<td>kg N day(^{-1})</td>
<td>lbs N day(^{-1})</td>
</tr>
<tr>
<td>“Turkeys”</td>
<td>0.300</td>
<td>22.3%</td>
<td>16%</td>
<td>0.011</td>
<td>0.024</td>
</tr>
<tr>
<td>“Chickens Broiler”</td>
<td>0.117</td>
<td>20.3%</td>
<td>16%</td>
<td>0.004</td>
<td>0.008</td>
</tr>
<tr>
<td>“Layer Chickens”</td>
<td></td>
<td></td>
<td></td>
<td>0.002</td>
<td>0.004</td>
</tr>
</tbody>
</table>
2: Methods: nonpoint source nitrogen loadings to surface waters

Cropland losses of nitrogen via groundwater discharge to surface waters

**N leaching losses**

"N leaching" here refers only to that N which leaches to shallow groundwater, where it will over time either be denitrified in the groundwater, or discharge into surface waters. Discharge into well waters was not considered. N leaching into tile drainage waters is a separate study component, and is not considered in the category of “N leaching losses.” However, N leaching that moves vertically on tile-drained land and does not move into tile lines is considered in the “N leaching losses” component.

Total cropland N leaching was determined based on the amount of leaching on undrained soils in: 1) fertilized crops (corn, corn silage, wheat, barley, oats, sugarbeet, potato), 2) non-fertilized crops (soybean and alfalfa), and 3) leaching losses from all crops on drained soils.

Cropland area in each agroecoregion was classified as either drained or un-drained according to soil hydrologic class for soils with slope steepness between 0-3% (SSURGO classification, USDA- NRCS, 2006b).

A literature review was conducted to determine the N leaching rate for each agroecoregion. Most of the research related to N leaching in Minnesota has been conducted in the Sand Plains area (Venterea et al., 2011, Wilson et al., 2010, Wilson et al., 2008, Errebhi et al., 1998, Sexton et al., 1996, Delin et al., 1995, Rosen et al., 2010, Rosen, pers. comm.).

Using existing data for Minnesota, statistical algorithms were developed for N leaching losses based on the applied N rate in dry and wet years. For average climatic years, the N leaching algorithm was based on a mean of the algorithms in dry and wet years. Dry years occurred when precipitation was lower than the average using a 30-year climatic record for each Minnesota location in a particular research study. Wet years occurred when precipitation was greater than the average using a 30-year climatic record for each Minnesota location studied.

Algorithms for N leaching losses to groundwater with fertilized crops in undrained areas of Region 4 (Figure 4) were, thus, a function of the N application rate (N fertilizer +N manure):

\[
\text{N losses} = \text{N rate} * 0.0602 + 22.245, \quad R^2 = 0.0871 \text{ for dry conditions}
\]

\[
\text{N losses} = \text{N rate} * 0.2945 + 37.6, \quad R^2 = 0.459 \text{ for wet conditions}
\]

Leaching is greatly reduced during dry conditions, regardless of the fertilizer rate, and thus the relationship between rate of application and nitrogen leaching during dry years was rather weak and the slope was low compared to the wet years. The poor statistical relationship during the dry years is expected. This relationship does not have much influence on the leaching loss estimates, given the narrow range of average fertilizer rates which are applied in different agroecoregions and the low leaching rates during dry years. Even if the dry years slope in figure 4 was zero, the N leaching load estimates would remain largely unaffected.
Figure 4. Cumulative NO₃-N leaching as a function of N rate for the Sand Plain area (Region 4) in Minnesota.
Each agroecoregion was assigned to one of four groundwater leaching regions according to groundwater contamination susceptibility in Minnesota. Assignment into each region was based on the measured occurrence of nitrate-N in drinking water wells from a database of 40,000 wells monitored by MDH, MPCA, USGS, and MDA. Regions were also based on results from the DRASTIC model (Depth, Recharge, Aquifer, Soil, Topography, Impact, and Conductivity).

Table 19. Groundwater contamination vulnerability regions, and associated agroecoregions and coefficients to adjust the N leaching rate in undrained fertilized cropland for each region.

<table>
<thead>
<tr>
<th>Region</th>
<th>Agroecoregion</th>
<th>Coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Drift &amp; Bedrock Complex, Forested Lake Sediments, Mahnomen Lake Sediments, Northern Till, Northshore Moraine, Peatlands, Poorly Drained Lake Sediments, Red Lake Loams, Somewhat Poorly Drained Lake, Steep Poorly Drained Moraine, Swelling Clay Lake Sediments, Very Poorly Drained Lake Sediments, Wetter BE Till, Wetter Clays &amp; Silts</td>
<td>0.007</td>
</tr>
<tr>
<td>2</td>
<td>Central Till, Coteau, Drumlins, Dryer BE Till, Dryer Clays &amp; Silts, Dryer Till, Forested Moraine, Inner Coteau, Mesabi Range, Poorly Drained BE = Till, Rolling Moraine, Steep Dryer Moraine, Steep Stream Banks, Steeper Till, Stream Banks</td>
<td>0.25</td>
</tr>
<tr>
<td>3</td>
<td>Bufflands, Inter-Beach Sand Bars, Level Plains, Steep Valley Walls, Steep Wetter Moraine, Steeper Alluvium, Undulating Plains</td>
<td>0.50</td>
</tr>
<tr>
<td>4</td>
<td>Alluvium &amp; Outwash, Anoka Sand Plains, Rochester Plateau</td>
<td>1</td>
</tr>
</tbody>
</table>

Leaching coefficients to adjust N leaching rate in Regions 1 and 2 were determined based on SWAT model information, and coefficients for Region 3 were assumed to be halfway between the coefficients for Regions 2 and 4, since no data were available (Table 19). No adjustment was needed in Region 4, because this is where experimental data on N leaching losses were abundant. Even though the geology differs in the Anoka Sand Plains region and Rochester Plateau (karst region) of southeastern Minnesota, it was justifiable to combine them into the same groundwater contamination vulnerability region. Each has roughly the same probability of groundwater contamination.
For non-fertilized soybean on undrained land, N leaching rates were assumed equal to N leaching rates for soybean under tile drainage. These N leaching rates were adjusted using the coefficients in Table 19 for Regions 1 and 2. For alfalfa, N leaching rates were assumed to be 1.56 pounds ac⁻¹ (Chung et al. 2001) for dry, average, and wet years.

Leaching losses on drained cropland were calculated assuming N rate loss of 3 pounds ac⁻¹ for dry, average, and wet years.

**Denitrification of groundwater**

The main form of N in groundwater baseflow is nitrate, which moves with water and ultimately can reach surface waters. However, nitrate can be lost before discharging to surface water through a biological process called denitrification.

Denitrification can occur within the unsaturated soil zone, within saturated soils, in the aquifer, and/or in the riparian zone. Levels of oxygen in groundwater < 0.5 mg L⁻¹ can promote denitrification, since bacteria will use nitrate to oxidize organic carbon sources, and as a result, nitrate contributions from low-oxygen baseflow will be negligible or minimal. However if these conditions are not present, then all the nitrate that moves through the soil into groundwater will eventually emerge in streams via groundwater baseflow.

The amount of groundwater N discharging to surface waters was calculated by multiplying the N leaching losses by a denitrification factor. The denitrification factor was determined based on a literature review which summarizes possible nitrate losses in the groundwater for different types of soils (Böhlke et al., 2002; Dubrovsky et al., 2010; Duff et al., 2007; Duff et al., 2008; Gentry et al., 2009; Goolsby et al., 1999; Hill, 1996; Korom, 2010; Korom et al., 2005; Masarik et al., 2007. McCallum et al., 2008; MPCA, 1998; Patch and Padmanabhan, 1994; Puckett, 2004. Puckett and Cowdery, 2002; Puckett et al., 1999; Puckett et al., 2008; Rodvang and Simpkins, 2001; Sauer et al., 2001; Schilling, 2002; Schilling and Helmers, 2008; Schilling and Libra, 2000; SCWRS, 2003; Sogbedji et al., 2000; Spahr et al., 2010; Tesoriero et al., 2009; Triska et al., 2007; and Trojan et al., 2002). The actual losses within groundwater prior to discharge into surface waters is highly variable, and will depend on the subsurface and groundwater chemistry, residence time in aquifers, and the types of sediments it moves through in the riparian zone. Based on the available information, denitrification losses within the groundwater itself were assumed to be 25% for Karst agroecoregions, 40% for Sand Plain and Alluvial agroecoregions, 60% for finer textured soil agroecoregions , and 50% for all other agroecoregions (Table 20).

Considerable lag time can occur between the time of leaching into groundwater and the point of discharge into surface waters. Land management changes that affect leaching losses can take from weeks to centuries before the changes are reflected in surface waters. This lag time was not directly accounted for in this study. Estimates of discharge into surface waters are independent of travel time, except that denitrification coefficients were adjusted based on the hydrologic conditions within the agroecoregion. The estimates of N reaching surface waters through leaching losses will not necessarily be reflected in the stream monitoring for a single year, or even a single decade.
Table 20. Groundwater denitrification factor assigned to different agroecoregions.

<table>
<thead>
<tr>
<th>Agroecoregion</th>
<th>Denitrification factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blufflands, Rochester Plateau</td>
<td>0.25</td>
</tr>
<tr>
<td>Anoka Sand Plains, Alluvium and Outwash, Inter-Beach Sand Bars, Steep Valley Walls, Steeper Alluvium.</td>
<td>0.40</td>
</tr>
<tr>
<td>Forested Lake Sediments, Mahnomen Lake Sediments, Poorly Drained BE Till, Poorly Drained Lake Sediments, Red Lake Loams, Somewhat Poorly Drained Lake, Swelling Clay Lake Sediments, Very Poorly Drained Lake Sediments</td>
<td>0.60</td>
</tr>
<tr>
<td>Other agroecoregions</td>
<td>0.50</td>
</tr>
<tr>
<td>Drained soils</td>
<td>0.60</td>
</tr>
</tbody>
</table>

Cropland nitrogen losses to surface waters in tile drainage discharge

Annual tile drainage N losses are difficult to estimate, since several factors influenced N export through tile drainage. In Minnesota, extensive research has been developed on N losses in tile drainage (Chung et al., 2001, Randall et al., 1997, Huggins et al., 2001, Nangia et al., 2008, Randall and Iragavarapu, 1995, Randall and Vetsch, 2005, Randall et al., 2003, Sands et al., 2008, Randall et al. 2000, Gast et al., 1978).

Total N losses in tile drainage were determined based on the amount of N losses in croplands under: 1) fertilized crops (corn, corn silage, wheat, barley, oats, sugar beet, potato), 2) and non-fertilized crops (soybean and alfalfa).

Based on the available information, two algorithms were developed for corn and soybean crops (Figure 6). Algorithms were a function of growing season precipitation and N rate (N fertilizer + N manure) for fertilized crops, and only growing season precipitation for non-fertilized crop (soybean) in each agroecoregion:
Figure 6. Nitrate losses through tile drainage as a function of precipitation for corn and soybean. The regression equation for corn is based on both precipitation and N rate.

Algorithms for N losses to surface waters through tile drainage took the form:

\[ \text{NO}_3^\text{-N losses (fertilized crops)} = -70.334 + \text{Precipitation} \times 0.11603 + \text{N rate} \times 0.13985 \]

\[ \text{R}^2 = 0.3742 \]

\[ \text{NO}_3^\text{-N losses (soybean)} = -29.166 + \text{Precipitation} \times 0.0726 \]

\[ \text{R}^2 = 0.2194 \]

For alfalfa forage, an N leaching rate of 1.56 lb ac\(^{-1}\) was estimated based on N leaching research in Minnesota (Chung et al., 2001).

Total N losses were calculated for dry, average, and wet climatic conditions based on growing season precipitation data (MPCA HUC8 precipitation data 1980-2010, MN DNR, 2010).
Tile drainage N losses estimated using the algorithms above were inflated an additional 12% to account for contributions of TKN (organic N forms) based on studies conducted in Minnesota (Dave Wall, personal communication with S. Matteson, Nov. 2011).

**Cropland nitrogen losses in runoff**

Nitrogen in surface runoff was calculated as a function of runoff volume and N concentration for each agroecoregion.

Thirty years of precipitation data were analyzed at the basin scale, and wet, average, and dry years were determined based on the statistical 90th, 50th, and 10th percentiles, respectfully. Discharge volume from USGS monitoring was determined using average, low and high flow discharge data for these same years in each agroecoregion.

Runoff as a percent of discharge was determined based on available data from SWAT modeling for the agroecoregions cited in Table 21. For the remaining agroecoregions runoff percentages were calculated based on a water budget approach for each agroecoregion.

**Table 21. Runoff (percent of discharge) from SWAT modeling.**

<table>
<thead>
<tr>
<th>Area</th>
<th>Agroecoregion</th>
<th>Runoff (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>7 Mile Creek</td>
<td>Wetter Clays and Silts</td>
<td>22</td>
</tr>
<tr>
<td>Root River</td>
<td>Undulating Plains</td>
<td>16</td>
</tr>
<tr>
<td>Karst</td>
<td>Blufflands, Rochester Plateau</td>
<td>24</td>
</tr>
<tr>
<td>Red River</td>
<td>Swelling Clay lake sediments, Very poorly drained lake sediments</td>
<td>71</td>
</tr>
<tr>
<td>Sunrise</td>
<td>Central till, Anoka Sand Plains, Alluvium and Outwash</td>
<td>6</td>
</tr>
</tbody>
</table>

Similar to the approach used to estimate cropland N losses through leaching and groundwater discharge, each agroecoregion was assigned a runoff category according to their susceptibility to surface runoff in Minnesota (Table 22). For Blufflands and Rochester Plateau, runoff was assigned an N concentration of 10 mg L⁻¹ based on data reported by Peterson and Vondracek (2006) for the Karst region.
Table 22. Nitrogen concentration in cropland runoff for each Agroecoregion.

<table>
<thead>
<tr>
<th>Region</th>
<th>Agroecoregion</th>
<th>N concentration (mg L$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Drift &amp; Bedrock Complex, Forested Lake Sediments, Mahnomen Lake Sediments,</td>
<td>3.5$^1$</td>
</tr>
<tr>
<td></td>
<td>Northern Till, Northshore Moraine, Peatlands, Poorly Drained Lake Sediments,</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Red Lake Loams, Somewhat Poorly Drained Lake, Steep Poorly Drained Moraine,</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Swelling Clay Lake Sediments, Very Poorly Drained Lake Sediments, Wetter BE</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Till, Wetter Clays &amp; Silts</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Central Till, Coteau, Drumlins, Dryer BE Till, Dryer Clays &amp; Silts, Dryer</td>
<td>1.8$^2$</td>
</tr>
<tr>
<td></td>
<td>Till, Forested Moraine, Inner Coteau, Mesabi Range, Poorly Drained BE =Till,</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Rolling Moraine, Steep Dryer Moraine, Steep Stream Banks, Steeper Till,</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Stream Banks</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Blufflands, Inter-Beach Sand Bars, Level Plains, Steep Valley Walls, Steep</td>
<td>0.7$^3$</td>
</tr>
<tr>
<td></td>
<td>Wetter Moraine, Steeper Alluvium, Undulating Plains</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Alluvium &amp; Outwash, Anoka Sand Plains, Rochester Plateau</td>
<td>0.24$^4$</td>
</tr>
</tbody>
</table>

$^1$ Kumar et al. 2009 (East Grand Forks, MN), Ginting et al. 2000 (southern Minnesota River Basin)
$^2$ Thoma et al. 2005 (Lamberton, MN)
$^3$ No research data were available for zone 3 (assumed intermediate values)
$^4$ Delin and Landon (2002) (Sand Plain-Princeton)

Forest export of nitrogen to surface waters


Table 23. Nitrogen export coefficients for forested lands in Minnesota.

<table>
<thead>
<tr>
<th>Conditions</th>
<th>N export (lbs N ac$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dry</td>
<td>1</td>
</tr>
<tr>
<td>Average</td>
<td>2</td>
</tr>
<tr>
<td>Wet</td>
<td>3</td>
</tr>
</tbody>
</table>
Nonpoint source nitrogen export in urban/suburban regions

Based on information from National Land Cover Database (NLCD, 2006), the total acres of developed land use (open space, light, medium, and heavy developed) was approximately 1 million acres statewide.


Table 24. Nonpoint source N export coefficients for urban/suburban lands in Minnesota.

<table>
<thead>
<tr>
<th>Conditions</th>
<th>N export (lbs N ac⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dry</td>
<td>2</td>
</tr>
<tr>
<td>Average</td>
<td>4</td>
</tr>
<tr>
<td>Wet</td>
<td>6</td>
</tr>
</tbody>
</table>

Nitrogen export from septic systems

Nitrogen losses from septic systems were based on county data from MPCA (2011). Losses were estimated for septic systems that are Imminent Public Health Threats (IPHT) and for those that are not IPHT as follows:

\[
\text{Septic N to Groundwater} = \left( \# \text{ Septics per county} \times \left( \text{Persons per household by county} \times \left( 9.1 \text{ pounds N per person} \times 0.85 \text{ for denitrification losses} \right) \right) \right) \times \left( 1 - \% \text{ NOT Imminent Public Health Threat (IPHT)} \right)
\]

\[
\text{Septic N to Surface Water} = \left( \# \text{ Septics per county} \times \left( \text{Persons per household by county} \times 9.1 \text{ pounds N per person} \right) \right) \times \left( \% \text{ Imminent Public Health Threat (IPHT)} \right)
\]

Information to determine the number of people per household by county was obtained from U.S. Census (2010). The per capita N coming out of septic systems was assumed to be 9.1 pounds N per person (Information provided by Mark Wespetal, MPCA). Denitrification was assumed to remove 15% of the septic system N within the soil prior to reaching groundwater. Once in the groundwater, the same groundwater denitrification loss coefficients for cropland (Table 20) were assigned to septic system N. All non-metropolitan population data were classified using 2008 ZIP code populations to improve spatial accuracy of county data.

Feedlot nitrogen losses in runoff

The number of out of compliance feedlots for open runoff was determined from an MPCA survey of counties in 2010 (pers. comm. Don Hauge, MPCA). Some counties had missing information for the number of feedlots out of compliance, and numbers had to be estimated using results from similar counties.
Feedlot N runoff was estimated using the Minnesota Feedlot Annualized Runoff Model (MinnFarm model). MinnFarm model (version 2.3) was run for a 75 AU beef/dairy operation to represent feedlots in the 50-100 AU category, a 150 AU beef/dairy to represent feedlots in the 100-300 AU category, and a 300 AU beef/dairy to represent feedlots in the >300AU category, recognizing that not all animals at the farm typically have access to the noncompliant lots.

The MinnFARM model assumed 200 square feet per animal on the lot and over 100% animal unit density - all soil covered with some manure in the lot. Also the model considered a small buffer downslope of the lot, which reduced the N losses by half. This is equivalent to about a 25 foot length meadow or 75 feet of fair pasture.

These estimates do not account for runoff from non-registered feedlots, feedlots in counties with minimal animal agriculture and small amounts of N from compliant feedlots using vegetation to treat runoff.

Methods for assessing sensitivity and uncertainty in cropland nitrogen balance

Due to uncertainty in the estimation of the variables that affect the agricultural N balance, a sensitivity analysis was conducted to determine how changing these variables would affect the overall agricultural N balance. In the sensitivity analysis, each source or coefficient was varied in increments of plus or minus 5, 10, 15, 25, or 50% of its baseline value. For each of these changes, we computed the resulting percentage change in the overall agricultural N balance relative to its baseline value.

Also, a sensitivity analysis was conducted for the main N pathways to surface water (runoff, leaching, tile drainage).

Conversion of agroecoregion based nitrogen loadings to watershed based nitrogen loadings

The majority of N inputs and outputs were calculated based on agroecoregion boundaries because of how inherent similarities in soil type and parent material largely influence the amount of N stored or delivered. To convert these N loadings into data representing major watershed boundaries, they were area-weighted. Agroecoregion data totals were converted to pounds per acre N yield raster data sets. Zonal statistics were then used to calculate an area-weighted average N yield for HUC8 watershed polygons. This average N yield value was converted back to a total delivery in pounds based on watershed areas.

Some rounding errors may introduce small discrepancies between the agroecoregion based and watershed based data, but this is the best representation of the original data. When possible (i.e. urban N deliveries, forest N deliveries), data were calculated for each watershed based on 30 m landuse rasters. In other words, area-weighting was avoided when data resolution could be better represented with direct landuse calculations.

Finally, the area-weighting process introduced a high amount of tile drainage N delivery from the Mississippi Twin Cities major watershed, a watershed having little to no tile drained cropland. For this specific case, the Mississippi Twin Cities watershed was determined to have zero tile drainage, and the amount removed from this watershed was “de-weighted” or assigned back to the watersheds associated with the influencing agroecoregion.
Results

Minnesota cropland nitrogen balance

Physical description of study area
Minnesota has 54,000,000 acres of land in total and nearly 36,000,000 acres of cropland, forest and urban landuses (Figure 7). Cropland accounts for 44% of the total area in Minnesota, while forest accounts for 20%, and urban landuse accounts for 1%. There are another 18 million acres (33% by area) of lakes, rivers, shrub and grasslands, and wetlands not considered in this study. Cropland accounts for 67% of the area (36,000,000 acres) represented in this study of nonpoint source N pollution, forest accounts for 31% and urban-suburban land accounts for 2% (Figure 8). Cropland includes land in corn, soybean, small grains, sugar beet, potato, alfalfa, and hay. The three largest Basins in Minnesota (Minnesota River, Red River of the North, and Upper Mississippi River) account for nearly 60% of the area in the state. The largest concentration of cropland is in the Minnesota River, Red River of the North, and Upper and Lower Mississippi River Basins (Figure 9). Cropland accounts for 74% of the area in the Minnesota River Basin, 51% of the area in the Red River of the North, and only 21% of the area in the Upper Mississippi River Basin (Figure 10). The Lower Mississippi River Basin has 47% of its area in cropland. The Rainy River and Lake Superior Basins, by contrast, have only 1.2% and 0.2% of their areas in cropland.

Figure 7. Minnesota landuse categories for this study

Figure 8. Landuse percentages for this study
Figure 9. Cultivated cropland (ac) in Minnesota.

Figure 10. Landuse distributions (ac) by river basin.
Agricultural nitrogen inputs

Agricultural N inputs to cropland include soil mineralization, N fertilizer; N fixation by legumes, atmospheric deposition, planted seeds, and purchased animals. Land applied animal manure can also be compared with agricultural N inputs, though technically it is an internally recycled nutrient, and should not be explicitly considered. When land applied animal manure is included, agricultural N inputs total about 4.8 billion pounds of N. Mineralization accounts for 36% (1.73 billion pounds) of the N inputs to cropland (Figure 11a, b), while N fertilizer accounts for 29% (1.36 billion pounds). Nitrogen fixation by legumes (0.61 billion pounds) accounts for 13% of the N inputs. Land applied manure (0.45 billion pounds), atmospheric deposition (0.22 billion pounds), and purchased animals (0.36 billion pounds) each account for roughly 5-9% of the N inputs to cropland. Purchased seeds account for less than 1%. Not surprisingly, because of relatively large areas of cropland, the largest agricultural N inputs to cropland (Figure 12) occur in the Minnesota River Basin (1.7 billion pounds), followed by the Red River of the North Basin (1.0 billion pounds) and the Upper Mississippi River Basin (0.71 billion pounds). The Lower Mississippi River Basin receives roughly 0.52 billion pounds of N annually. A majority of the N inputs for these four basins arises from soil mineralization and N fertilizer.

Figure. 11a, b. Agricultural inputs (% or lb) by source to the N balance.
When normalized by total watershed area, the largest N inputs to cropland from a single source occur with soil mineralization (Figure 13) in the Minnesota River, Missouri River, and Des Moines River Basins (54-75 pounds/acre). Mineralization of cropland soils is relatively insignificant (4-5 pounds/acre) in the Rainy River and Lake Superior Basins. The second largest source of N inputs is fertilizer (Figure 14). Fertilizer applications account for 48 to 57 pounds/acre annually in the Minnesota River, Missouri River, and Des Moines River Basins when averaged over the total watershed area (including non-cropland acres). When averaging only for cultivated acres (including unfertilized and fertilized crops, fertilizer application rates in these same basins are approximately 70 pounds/acre. Fertilizer rates range between 2.2 and 2.9 pounds/acre annually in the Rainy River and Lake Superior Basins.
Most of the fertilizer is applied to land used for growing corn, with some also applied to land used for growing other crops, including wheat, potatoes, edible beans, etc. Rates of N fertilizer applied to cropland used for growing corn generally range from 136-152 pounds/acre across a wide area covering the Minnesota River and Lower Mississippi River Basins, as well as the southern portions of the Upper Mississippi River Basin (Figure 15).

Figure 15. Fertilizer N rates (lb/ac) applied to corn.

Nitrogen fixation ranges between 12 and 22 pounds/acre annually in the Minnesota River, Missouri River, Des Moines River, and Lower Mississippi River Basins (Figure 11). Land applied manure ranges between 17 and 36 pounds/acre (normalized to total watershed area) in the Lower Mississippi River, Minnesota River, Des Moines River and Missouri River Basins (Figure 17). Not surprisingly, the heaviest concentration of farm animals (Figure 18) occurs in a broad swath covering the Minnesota River and Lower Mississippi River Basins.

Figure 16. N fixation (lb/ac) by basin.

Total N inputs (excluding land applied manure) normalized to watershed area are greatest for the Minnesota River, Missouri River, and Des Moines River Basins, ranging from 155-174 pounds/acre annually (Figure 19). Total N inputs range between 11 and 15 pounds/acre in the Rainy River and Lake Superior Basins.

Total N inputs (excluding land applied manure) are greatest for the Minnesota River Basin (1.5 billion pounds annually) and Red River of the North Basin (1.0 billion pounds) (Figure 20). The Upper Mississippi River Basin (0.63 billion pounds) and Lower Mississippi River Basin (0.44 billion pounds) have moderate amounts of total N inputs. Nitrogen inputs are less than 0.07 billion pounds in the Lake Superior Basin.
Figure 17. Land applied manure (lb/ac) by basin.

Figure 18. Animal units by major watershed.

Figure 19. Total inputs of agricultural N (lb/ac) by basin.

Figure 20. Total inputs of ag. N (lb) by basin.
Agricultural nitrogen outputs

Agricultural N outputs to cropland include crop removal (harvest), senescence, denitrification, animals sold, volatilization of fertilizer and manure, incinerated animal manure, milk and eggs sold, and N losses to the surface and groundwater by drainage, leaching and runoff. Animal feed (harvested crop fed to animals) can be compared with agricultural outputs, though technically it is an internally cycled nutrient, and should not be explicitly considered. Agricultural N outputs (including animal feed) total roughly 5.0 billion pounds annually. Crop removal (harvest) accounts for 45% (2.2 billion pounds) of the total N outputs, by far the largest pathway (Figs. 21-22). Harvested crop used for animal feed accounts for another 15% (0.75 billion pounds). Senescence and denitrification account for 14% (0.72 billion pounds) and 10% (0.48 billion pounds), respectively, of the N outputs. Volatilization of fertilizer and manure together account for 6% (0.27 billion pounds). Sales of meat, milk and eggs also account for about 3% (0.16 billion pounds). Losses to the environment by agricultural drainage, leaching and runoff together account for about 6% (0.29 billion pounds) of the total N outputs.

Figure 21. Agricultural N outputs by source (%). Crop removal is crops harvested for sale, export or ethanol production. Animal feed is crops harvested for livestock feeding in Minnesota, plus distiller dry grains from ethanol production that are fed to Minnesota livestock.

Figure 22. Agricultural N outputs by source (lb). Note that denitrification only refers to soil denitrification, and not subsequent denitrification in the underlying groundwater.
The four basins with the largest agricultural N outputs are the Minnesota River (1.5 billion pounds), Red River of the North (0.85 billion pounds), Upper Mississippi River (0.61 billion pounds), and Lower Mississippi River (0.56 billion pounds) Basins (Figure 23). A majority of the N outputs from each of these Basins arises from crop removal (harvest) and senescence plus denitrification. When normalized by watershed area, crop removal (Figure 24) is largest in the Missouri River and Des Moines River Basins (82-104 pounds/acre). Crop removal averages roughly 75-82 pounds/acre in the Minnesota River and Cedar River Basins. Crop removal accounts for 41-75 pounds/acre in the Lower Mississippi River. Crop removal averages only 3-5 pounds/acre in the Rainy River and Lake Superior Basins. Senescence is largest in the Missouri River and Des Moines River Basins (27-33 pounds/acre). Rates of senescence (Figure 25) average 14-27 pounds/acre in the Lower Mississippi River, Cedar River and Minnesota River Basins. Senescence averages 1-1.5 pounds/acre in the Rainy River and Lake Superior Basins.

Figure 23. Agricultural N outputs by source (lbs) and river basin.
Denitrification is largest in the Minnesota River and Des Moines River (14-20 pounds/acre) Basins (Figure 26). It is moderately large in the Red River of the North and Cedar River (7-14 pounds/acre) Basins. Denitrification is elevated in all four of these Basins relative to the other Basins as a result of a large proportion of land that is poorly drained. Much of this cropland, particularly in the Minnesota River Basin, has been improved by installation of artificial drainage to make growing annual crops more economically profitable or feasible (Figure 27).
Total N outputs (excluding crop harvested for animal feed) normalized to watershed area (Figure 28) are greatest for the Missouri River and Des Moines River Basins, ranging from 179-181 pounds/acre annually. Nitrogen outputs are also significant in the Minnesota River and Cedar River Basins (158 pounds/acre) and the Lower Mississippi River Basin (138 pounds/acre). Total N outputs range between 7 and 9 pounds/acre in the Rainy River and Lake Superior Basins.

Total N outputs (excluding crop harvested for animal feed) are greatest (Figure 29) for the Minnesota River Basin (1.5 billion pounds annually). Total N outputs for the Red River of the North Basin are next highest at 0.85 billion pounds. The Upper Mississippi River Basin (0.61 billion pounds) and Lower Mississippi River Basin (0.56 billion pounds) have moderate amounts of total N outputs. N outputs are less than 0.06 billion pounds in both the Rainy River and Lake Superior Basins.
The overall N balance is obtained by subtracting total N outputs (4.3 billion pounds) from total N inputs (4.2 billion pounds). The inputs and outputs do not include internally recycled N from the harvested crops which are fed to livestock and then later returned to the soil as manure. Results of this give 0.09 billion pounds of N (outputs exceed inputs), about 2.1% of the inputs or outputs. This result shows that the overall N balance is excellent. Individual N balances (Figs. 30a, b) are excellent for the Cedar, Des Moines, Lake Superior, Minnesota, Missouri, Rainy, St. Croix, and Upper Mississippi River Basins (errors less than 1% of total inputs). The errors in the N balance arise primarily from the Lower Mississippi River and Red River of the North Basins. The N balance in the Red River of the North Basin is overestimated by about 3.4% or 0.15 billion pounds (inputs exceed outputs by 13 pounds/acre), while the N balance is underestimated by about 2.8% or 0.12 billion pounds (outputs exceed inputs by 29 pounds/acre) in the Lower Mississippi River Basin (Figs. 31-32).
Figure 30a, b. Agricultural N inputs minus outputs (lb or %) by basin.

Figure 31. Ag. N inputs minus outputs (lb/ac) by basin

Figure 32. Ag. N inputs minus outputs (lb) by basin.
Agricultural nitrogen balance sensitivity analysis and uncertainty

The agricultural N balance is dependent on many sources of N, each of which itself depends on various coefficients. Development of the agricultural N balance was based on the principle that each source and coefficient should independently be estimated based on the best available data or scientific research relevant to site-specific conditions in Minnesota. As such, there was little to no calibration of sources or coefficients.

There is a certain level of uncertainty inherent with each source and coefficient used in the agricultural N balance. A sensitivity analysis was conducted to determine how varying certain key sources or coefficients would affect the overall agricultural N balance. In the sensitivity analysis, each source or coefficient was varied in increments of plus or minus 5, 10, 15, 25, or 50% of its baseline value. For each of these changes, we computed the resulting percentage change in the overall agricultural N balance relative to its baseline value.

The agricultural N balance was most sensitive to changes in three factors (Figure 33), namely; crop removal (excluding crop harvested for animal feed), net mineralization and amount of applied N fertilizer. Changing any of these three factors by plus or minus 50% would cause the overall balance to change by plus or minus 17-28%.

![Figure 33. Sensitivity analysis of N input factors on agricultural N balance.](image)

There is a difference, however, between sensitivity and uncertainty. While a change in applied N fertilizer of 50% would cause the N balance to increase by about 17%, the uncertainty in the amount of basin wide N fertilizer application rates is believed to be relatively small. The amount of N
fertilizer applied in Minnesota is accurately known compared to other inputs due to good data collection and survey methods to track fertilizer sales and application rates. Similarly, there is a low uncertainty in the amount of N removal by crops, because the amount harvested is accurately known.

In contrast, the amount of net mineralization is moderately uncertain. Net mineralization fluctuates from year to year and place to place based on variations in soil moisture and temperature. Our estimates of net mineralization are based on state-wide average soil moisture and temperature conditions. It is likely that these estimates of net mineralization have an uncertainty of plus or minus 10-25%. With these levels of uncertainty in net mineralization, the N balance would change by up to plus or minus 10% (equivalent to up to 0.4 billion pounds of N). Future research should address the impacts of variations in soil moisture and temperature on net mineralization.

As stated in the previous section, the N balance in the Red River of the North Basin is overestimated by about 0.15 billion pounds, while the N balance in the Lower Mississippi River Basin is underestimated by about 0.12 billion pounds. These differences could be a result of poor estimates of net mineralization in each Basin. The Red River of the North Basin tends to have soils which are cooler and drier than soils in many of the other Basins. This would cause net mineralization to be reduced relative to rates in other Basins. A decrease in net mineralization of 10% in the Red River Basin would be able to correct for the overestimation of N inputs in that Basin. In contrast, soils in the Lower Mississippi River Basin tend to be warmer and wetter than soils in many of the other Basins. This would cause net mineralization to be increased relative to rates in other basins. An increase in net mineralization of 10% in the Lower Mississippi River Basin would be able to correct for the underestimation of N inputs in that basin.

The agricultural N balance was moderately sensitive to three factors, namely; senescence, denitrification, and N fixation. All three of these factors are known to be rather uncertain as a result of variations in climate and soil type. Yet, changing any one of these three factors by as much as plus or minus 50% would only cause the N balance to change by plus or minus 6-9%. Therefore, while we are uncertain about the values of senescence, denitrification or N fixation, errors in estimating them would not cause large changes in the N balance.

Uncertainty in any of the remaining factors (atmospheric deposition, planted seeds, purchased animals, N losses to groundwater, N losses in runoff or tile drainage, volatilization of fertilizer or manure or sales of milk, eggs or meat) would have only minor impacts on the agricultural N balance. Changing any one of these factors would not change the agricultural N balance by at most a few percent.

Results: Minnesota nonpoint source N loadings to surface waters

Total nonpoint source N loadings to Minnesota surface waters are estimated at 254 million pounds during an average climatic year. Sources of N loadings to surface waters included cropland drainage (114 million pounds in an average year), cropland runoff (16 million pounds) and cropland leaching (93 million pounds); forest export of N (22 million pounds); urban/suburban export of nonpoint source N (3 million pounds); feedlot runoff (0.2 million pounds), and individual septic treatment system losses (5 million pounds).

The spatial distribution of modeled (estimated through this study) total nonpoint source N loadings to Minnesota surface waters during an average climatic year is shown in Figure 34. These modeled results compare well with water quality monitoring data as shown below (Figure 35). Predicted N loadings are highest for the Zumbro and Root Rivers of southeastern Minnesota, where N loadings from groundwater, drainage and runoff are all high. Predicted N loadings are next highest in a cluster of major watersheds centered in the Minnesota River Basin, where N losses in drainage are high, but
groundwater and runoff losses are smaller than in southeastern Minnesota. These 15 major watersheds in southeastern, southern, and west central Minnesota contribute 140.6 million pounds of nonpoint source N loadings to surface waters. This is 55% of estimated N loadings in the entire state of Minnesota. On a per acre basis, the highest loadings occur in 8 watersheds located in southern Minnesota (Figure 34). Loadings per acre are generally highest in the Minnesota River and Lower Mississippi River Basins.

![Figure 34. Modeled average N loading to major watersheds, in lb (left) or lb/ac (right).](image)

**Comparison between modeled and monitored nitrogen loadings**

A comparison between the combined-source modeled nonpoint source N loadings to Minnesota surface waters (in an average climatic year) and monitored N loadings (average of two typical years) was conducted for 33 MPCA monitored major watersheds across Minnesota. Monitored N loadings were not used to calibrate the modeled nonpoint source N loadings, as the modeled N loadings were estimated independently, without calibration. Linear regression between modeled and MPCA monitored N loads (Figure 35) was very good ($y = 1.33x - 631,920; R^2 = 0.69$). Modeled N loadings across all monitored watersheds were 10% higher than monitored N loads. It should be noted that modeled nonpoint source loadings are estimated before in-stream and channel losses would take place (these are reflected in observed monitoring results). Thus, it is not surprising that modeled N loads are larger than monitored N loads. Other differences could arise because monitoring results include effects of point sources, whereas modeled results do not.

Further analyses comparing river monitoring results and estimated total N delivered to waters from all sources (point sources, nonpoint sources and atmospheric deposition directly into waters) are described in Chapter E1.
Figure 35. Modeled versus monitored major watershed N loads. Each point represents one HUC8 watershed average TN load obtained between 2007 and 2009.

The Lake Superior Basin consists of 5 major watersheds, 3 of which have one year of water quality monitoring data. Modeled nonpoint N loads in the Cloquet River are comparable in magnitude to average monitored total N loads (Figure 36). Modeled nonpoint N loads are lower than monitored loads in the St. Louis River watershed.

The Rainy River Basin consists of 9 major watersheds, 3 of which each have three years of water quality monitoring data. Modeled nonpoint N loads in the Vermillion, Little Fork, and Big Fork River watersheds are comparable in magnitude to average monitored total N loads (Figure 37). Modeled nonpoint loads are slightly lower than monitored loads in the Little Fork watershed.

The Red River of the North Basin consists of 17 major watersheds. Eleven of these have one to three years of water quality monitoring data. Modeled nonpoint source N loads in the Otter Tail, Buffalo, Marsh, Wild Rice, Sandhill, Thief, Clearwater, Snake, and Tamarac River watersheds are higher than average water quality monitoring results (Figure 38). This is reasonable, given the fact that modeled N loads do not account for in-water denitrification beyond the edge of field. Modeled nonpoint source N loads in the Two Rivers watershed are lower than monitored N loads.

Figure 36. Modeled versus monitored N loads Lake Superior Basin.
Figure 37. Modeled versus monitored N loads Rainy River Basin.

Figure 38. Modeled versus monitored N loads Red River of the North Basin.
There are 15 major watersheds in the Upper Mississippi River Basin. Nine watersheds have from one to three years of water quality monitoring data. Modeled nonpoint source N loads are higher than average water quality monitoring results in the Leech Lake, Pine, Crow Wing, Red Eye, Long Prairie, and Rum River watersheds (Figure 39). This is to be expected, since modeled N loads do not account for denitrification or biological uptake that occurs beyond the edge of field. Modeled N loads are quite a bit higher than measured loads in the North Fork of the Crow River watershed. The South Fork of the Crow River watershed has about 0.5 million pounds of N from point sources that are not included in modeled results. Modeled N loads in the Upper Mississippi River Twin Cities watershed also do not include about 11 million pounds of point sources.

The St. Croix River Basin includes 4 major watersheds, of which 2 have each been monitored for three years of water quality data. Modeled nonpoint source N loads are very comparable to (although somewhat higher than) average monitored water quality data in the Kettle and Snake River watersheds (Figure 40).

The Lower Mississippi River Basin includes 7 major watersheds. The Cannon and Root River watersheds have been monitored for water quality during the last 17 to 18 years. Modeled nonpoint source N loads are slightly larger than average water quality monitoring data in both the Cannon and Root River watersheds (Figure 41). Thus, modeled and monitored N loads agree quite well in the Lower Mississippi River Basin.

The Minnesota River Basin includes 12 major watersheds. Nine of these watersheds have been monitored for one to three years by the MPCA. Modeled nonpoint source N loads are somewhat larger than, or comparable in magnitude to average water quality monitoring data in the Pomme de Terre, Chippewa, Redwood, and Watonwan River watersheds (Figure 42). Modeled N loads are significantly higher than measured loads in the Cottonwood River watershed. Modeled N loads are significantly lower than measured N loads in the Le Sueur and Blue Earth River watersheds.

Figure 39. Modeled versus monitored N loads in the Upper Mississippi River Basin
Figure 40. Modeled versus monitored N loads in the St. Croix River Basin.

Figure 41. Modeled versus monitored N loads in the Lower Mississippi River Basin.
Several possible reasons could be invoked to explain the difference between modeled and monitored N loads in the Blue Earth and Le Sueur River watersheds. These watersheds have large areas of lacustrine soils and are intensively tile-drained. The modeled tile drainage losses may be underestimated in these watersheds, due to underestimates of tile-drained lands and/or underestimating losses from tile-drained fields. Second, the Blue Earth and Le Sueur River watersheds have some very deeply incised river channels and there is significant seepage along the bluff faces. This seepage of groundwater could be a source of additional N that is not accounted for in the modeled results.

Uncertainties in nitrogen loadings

The three primary pathways for N loadings in agricultural regions were by drainage, leaching, and runoff. There are uncertainties in the factors and coefficients used to estimate N loadings via each pathway. Losses of N in agricultural drainage are primarily dependent on three factors, namely; the areal extent of tile drainage, growing season precipitation, and the amount of N applied to cropland from fertilizer and manure. A sensitivity analysis was conducted to determine how changes in each of these factors affected the losses of N in agricultural drainage (Figure 43). Nitrogen losses in agricultural drainage were very sensitive to growing season precipitation. Increasing or decreasing growing season precipitation by 50% caused N losses in agricultural drainage to increase or decrease by 150%. This has important implications for comparisons between modeled nonpoint source N losses and monitored N losses in tile-drained regions. If the period when water quality monitoring data were collected is wetter or dryer than average, modeled N losses will be smaller than or larger than monitored N losses, respectively. Nitrogen losses in drainage were much less sensitive to changes in tile-drained area or applied N rates. Changes in either factor of up to plus or minus 50% would change the modeled N losses in drainage by less than plus or minus 50%. As mentioned previously, there is little relative uncertainty in applied N rates. The areal extent of tile-drained lands may be larger than the area estimated for this study if landscapes steeper than 3% slope or soils with hydrologic group B have subsurface tile drainage.
Underestimation of tile drained acreages was limited to less than 10% in the small Beauford Watershed located in the Le Sueur major watershed. If the underestimation of tile drainage is limited to 10% or less, then the resulting uncertainty in drainage N losses would be less than 10%. The extent of tile drainage is likely underestimated in the Minnesota River Basin. Adjusting for this would increase N loadings in tile drained regions of the Minnesota River Basin.

Figure 43. Sensitivity analysis for N losses in agricultural drainage.
Figure 44. Sensitivity analysis for agricultural leaching contributions to surface water N loads.

Losses of N in agricultural leaching, with subsequent discharge of groundwater to surface waters, are estimated using algorithms that depend on applied N rate and precipitation. Modeled losses of N for a given precipitation or irrigation regime are primarily affected by four coefficients along with the rate of applied N (Figure 44). As discussed previously, there is little uncertainty in the rate of applied N. The four coefficients determine by how much the leaching algorithm is adjusted for each region of the state. Changing coefficient one would have an insignificant impact on N losses by leaching. Changing any one of coefficients two-four by plus or minus 50% would change the modeled N losses by leaching by plus or minus 9-18%. Because of limited leaching quantification studies, uncertainty exists in the four leaching coefficients, especially coefficients for regions 2 and 3. More important, and more uncertain, are values of groundwater denitrification prior to surface water discharge. Changing the first three denitrification coefficients by plus or minus 50% would increase or decrease groundwater discharge of N to surface water by 17-22%.

Losses of N in agricultural runoff are estimated based on amounts of river discharge contributed by runoff and by concentrations of N in runoff. There are five values used for concentration of N in runoff (ranging from 0.24 to 10 mg/L), which vary region by region across the state. In general, results of the sensitivity analysis showed that as N concentration in runoff increased, the sensitivity of the modeled N losses in runoff also increased (Figure 45). In regions where N concentration in runoff is between 1.8 and 10 mg/L, changing N concentrations in runoff by 50% would change modeled N losses in runoff by up to plus or minus 25%. In regions where N concentration in runoff is between 0.24 and 0.7 mg/L, changing N concentration in runoff by 50% would have little impact on modeled N losses in runoff. Of greater importance is the sensitivity of the model to river discharge, which is sensitive to precipitation. Changing discharge by plus or minus 50% would change modeled N losses in runoff by plus or minus 50%.
Fortunately, river discharges are well known for dry, average and wet climatic conditions for various regions across the state. Hence, there is little uncertainty in river discharge for these three climatic regimes.

**Variation in nitrogen loads for dry, average and wet climatic years**

Climate has a significant effect on nonpoint source N loadings to surface waters in Minnesota. Total loadings of N to surface waters modeled for dry, average and wet years are roughly 106, 254, and 409 million pounds, respectively (Figure 46). Nitrogen losses by cropland leaching to groundwater and tile drainage are particularly sensitive to an increasingly wetter climate.

During a dry year (10th percentile precipitation years), the majority (46%) of nonpoint source N losses to surface waters arises from groundwater discharge (Figure 47). Losses from tile drainage (30%) and runoff (7%) on cropland are much smaller in comparison during a dry year. Losses from forested regions account for 10% of the total nonpoint source losses to surface waters. Septic system losses account for 5%. Losses of nonpoint source N from urban areas and feedlots are very small.
Losses of nonpoint source N during a dry year are largest for the Minnesota River watershed (30 million pounds), followed by the Lower Mississippi River watershed (27 million pounds) and the Upper Mississippi River watershed, with 21 million pounds of losses (Figure 48). Losses in the Red River of the North are about 10 million pounds. The other basins all have very small losses of nonpoint source N during a dry year.

During an average year (Figure 49), the nonpoint source losses from agricultural drainage (45%) increase relative to the losses from agricultural groundwater discharge (37%) in comparison with the losses during a dry year. Forest export of N accounts for 9% of the nonpoint source N losses during an average year, while agricultural runoff accounts for 6%. Septic system and urban losses account for only 2% and 1% of the total nonpoint sources, respectively. Losses from feedlots are insignificant.

During an average year, the Minnesota River Basin (34% or 86 million pounds of total nonpoint source N loadings) contributes more nonpoint source N losses than any other basin (Figs. 50-51). The Lower Mississippi River Basin (21% or 54 million pounds) contributes less than the Minnesota River Basin during an average year, in contrast to their relative contributions in a dry year. Modeled losses of nonpoint source N in the Upper Mississippi River are 18% or 46 million pounds. Losses from the Red River of the North are about 9% or 22 million pounds. The other basins contribute small nonpoint source N losses in comparison to the Minnesota, and Lower and Upper Mississippi River Basins.
During a wet year (90th percentile precipitation year), the majority of nonpoint source N losses statewide (Fig. 52) arise from agricultural drainage (49%). Discharge of groundwater from agricultural regions contributes another 34%. Forested regions generate 8%, and agricultural runoff generates 7% of the statewide nonpoint source N losses during a wet year. Other sources are relatively small in comparison.

The largest nonpoint source N losses in a wet year (Figure 53) occur in the Minnesota River Basin (146 million pounds). The Lower and Upper Mississippi River Basins generate 82 and 70 million pounds, respectively, of nonpoint source N during a wet year. The Red River of the North generates 36 million pounds. Other basins generate less than 15 million pounds each of nonpoint source N during a wet year.
Figure 51. Nonpoint source N loads (lb) for Minnesota in an average year.
Comparison between modeled and monitored N loads (including effects of point sources)

Data for point source N loads were provided by MPCA in the Minnesota, Red River of the North, St. Croix, and Upper Mississippi River Basins. These N loads were added to the modeled basin wide nonpoint source N loads described in previous sections. The basin total modeled plus point source N loads were compared with water quality monitoring data in each of the four river basins for dry, average, and wet climatic conditions. The monitoring data only represent the Minnesota contributions to the rivers.

In dry years (Figure 54), there was excellent agreement between monitoring data and modeled plus point source N loads in the Minnesota River Basin (32 million pounds), Red River of the North (10.3 million pounds), and St. Croix River (3.9 million pounds). In the Upper Mississippi River Basin monitored N loads were less than modeled plus point source N loads by about 7.3 million pounds, but this is not unexpected. Watersheds upstream of Sartell in the Upper Mississippi River Basin have from 10-40% of their area covered by wetlands. Nitrogen loads leaving fields and forest and entering these wetlands would be subject to further losses that are not reflected in modeled N loads. This could...
partially explain why modeled N loads are larger than monitored N loads in the Upper Mississippi River Basin. Additionally, because of the importance of the groundwater pathway in this region and the slow movement of groundwater, some of the nitrate from past decades has not yet reached the river and therefore would not be included in the monitoring data.

![Graph of dry monitored vs modeled N loads](image1.png)

**Figure 54.** Comparison of modeled + point source N loads with monitored N loads in a dry year.

During average climatic years (Figure 55), there was excellent agreement between monitoring data and modeled plus point source N loads in the Red River of the North (22.5 million pounds), St. Croix River (7.7 million pounds), and Upper Mississippi River (49.5 million pounds) Basins. Modeled loads underestimated monitored loads by about 28 million pounds in the Minnesota River Basin. It appears that there may be other sources of N (such as additional groundwater discharge and/or more tiled land than assumed) in the Minnesota River Basin that are not adequately accounted for in the modeled results.

![Graph of average monitored vs modeled N loads](image2.png)

**Figure 55.** Comparison of modeled + point source N loads with monitored N loads in an average year.

During wet climatic years (Figure 56), there was reasonably good agreement between monitoring data and modeled plus point source N loads in the Minnesota River (148.5 million pounds), Red River of the North (36.7 million pounds), St. Croix River (11.3 million pounds), and Upper Mississippi River (74.2 million pounds) Basins. As with average years, N loadings to the Upper Mississippi River were
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overestimated (by 14.6 million pounds), probably because as a result of denitrification losses occurring in wetlands and slow movement of groundwater. Monitored N loads in the Minnesota River Basin were underpredicted by about 34.5 million pounds. This is an underprediction of monitored loads by about 18%. Again, this indicates that there may be underestimated or other sources of N in the Minnesota River Basin not accounted for by modeled results.

![Figure 56. Comparison of modeled + point source N loads with monitored N loads in a wet year.](image)

Conclusions

An N budget was estimated for Minnesota's agricultural land as a separate and distinct analysis from the efforts to determine sources to surface waters. Inputs included mineralization (1.7 billion pounds), N fertilizer (1.36 billion pounds), N fixation by legumes (0.61 billion pounds), and several other smaller sources. The largest inputs occurred in the Minnesota River Basin. Outputs included crop removal (2.2 billion pounds), senescence (0.72 billion pounds), and denitrification (0.48 billion pounds), and several other smaller sources. The overall statewide N balance (inputs-outputs) was very good, with a difference of only 0.09 billion pounds (or 2.1% of the total inputs). This difference suggests that today's high biomass crops may be mining N from the soil, however, small increases in rates of soil mineralization assumed for the study could easily bring the system into balance.

Total nonpoint source N loadings to Minnesota surface waters were estimated at 254 million pounds during an average climatic year. This is about 6% of the total inputs of N on all Minnesota cropland (including soil mineralization). Sources of N loadings included cropland drainage (114 million pounds), cropland leaching (93 million pounds), forest N export (22 million pounds), cropland runoff (16 million pounds), individual septic treatment systems (5 million pounds), urban/suburban nonpoint source N (3 million pounds), and feedlot runoff (0.2 million pounds). During an average year, the Minnesota River Basin contributes more nonpoint source N loading to surface water (86 million pounds) than any other basin. The Lower Mississippi River Basin contributes (54 million pounds), the Upper Mississippi River Basin contributes 46 million pounds, and the Red River of the North contributes 22 million pounds. At the major watershed scale, modeled nonpoint source N loadings were highest for the Zumbro and Root Rivers of southeastern Minnesota, which are large watersheds where N loadings from groundwater
discharge, drainage and runoff are all significant. N loadings were next highest in a cluster of major watersheds in the Minnesota River Basin, where N losses in drainage are high. These major watersheds include the Lower Minnesota, Blue Earth, and Le Sueur River watersheds.

A comparison between the modeled nonpoint source N loadings to Minnesota surface waters (in an average climatic year) and monitored N loadings (average of two typical years) was conducted for 33 MPCA monitored major watersheds across Minnesota. Monitored N loadings were not used to calibrate the modeled nonpoint source N loadings, as the modeled N loadings were estimated independently, without calibration. Linear regression between modeled and MPCA monitored N loads was very good, with an $R^2$ value of 0.69. However, the modeled N loads were lower than monitored loads for several watersheds in south-central Minnesota. Modeled N loadings across all monitored watersheds were 10% higher than monitored N loads, which is not surprising given that additional losses in predicted N loadings may occur as nitrate travels downstream to the mouth of the watershed.

Climate has a significant effect on nonpoint source N loadings to Minnesota surface waters. Total statewide nonpoint source N loadings to surface waters for dry, average and wet years were predicted to be 106, 254, and 409 million pounds, respectively. During a dry year, the majority (46%) of nonpoint source N losses to surface waters arises from groundwater discharge. Losses from tile drainage (30%) and runoff (7%) on cropland are much smaller in comparison during a dry year. Losses from forested regions account for 10% of the total nonpoint source losses to surface waters. During an average year, the nonpoint source losses from agricultural drainage (45%) increase relative to the losses from agricultural groundwater discharge (37%) in comparison with the losses during a dry year. Forest export of N accounts for 9% of the nonpoint source N losses during an average year, while agricultural runoff accounts for 6%. During a wet year, the majority of nonpoint source N losses statewide arise from agricultural drainage (49%). Discharge of groundwater from agricultural regions contributes another 34%. Forested regions generate 8%, and agricultural runoff generates 7% of the statewide nonpoint source N losses during a wet year.
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