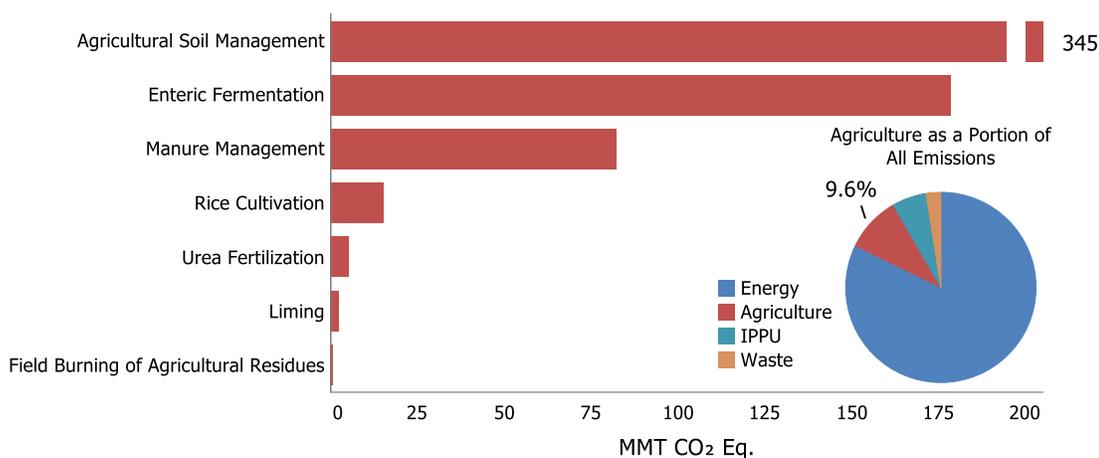


5. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. This chapter provides an assessment of methane (CH₄) and nitrous oxide (N₂O) emissions from enteric fermentation in domestic livestock, livestock manure management, rice cultivation, agricultural soil management, and field burning of agricultural residues; as well as carbon dioxide (CO₂) emissions from liming and urea fertilization (see Figure 5-1). Additional CO₂, CH₄ and N₂O fluxes from agriculture-related land-use and land-use conversion activities, such as cultivation of cropland, grassland fires, and conversion of forest land to cropland, are presented in the Land Use, Land-Use Change, and Forestry (LULUCF) chapter. Carbon dioxide emissions from stationary and mobile on-farm energy use and CH₄ and N₂O emissions from stationary on-farm energy use are reported in the Energy chapter under the Industrial sector emissions. Methane and N₂O emissions from mobile on-farm energy use are reported in the Energy chapter under mobile fossil fuel combustion emissions.

Figure 5-1: 2019 Agriculture Chapter Greenhouse Gas Emission Sources



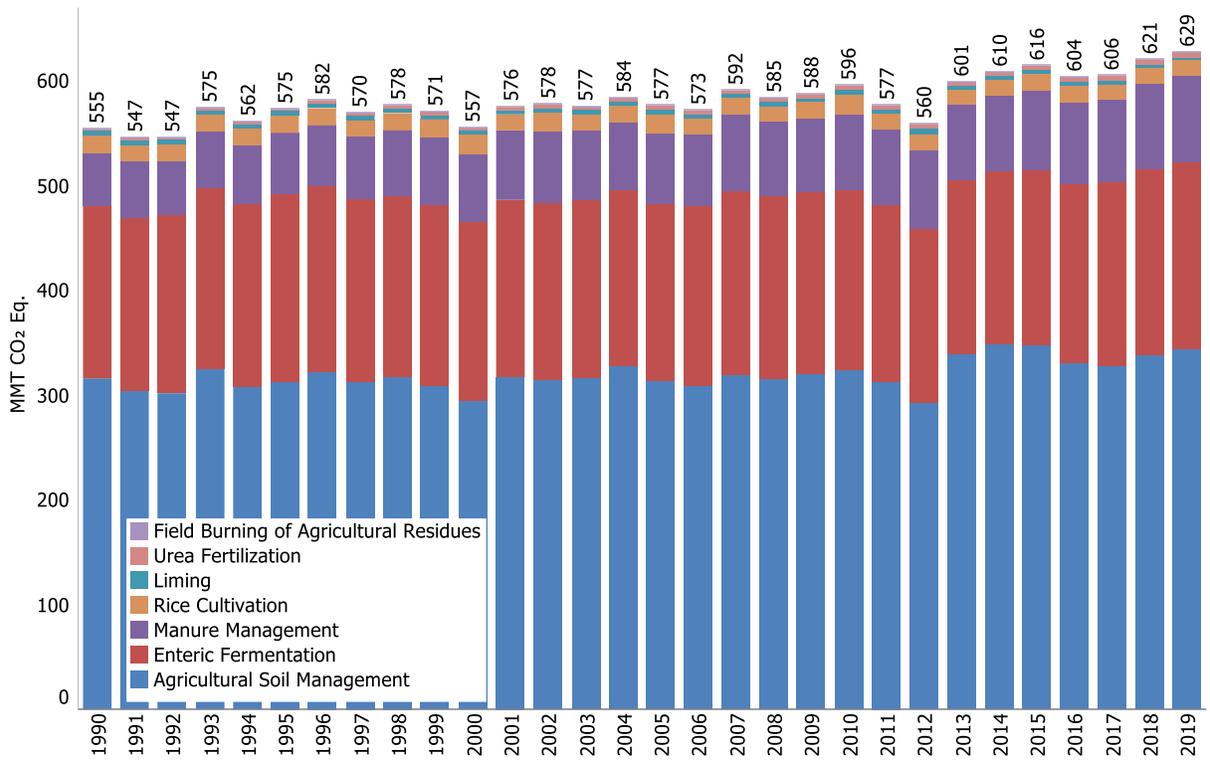
In 2019, the Agriculture sector was responsible for emissions of 628.6 MMT CO₂ Eq.,¹ or 9.6 percent of total U.S. greenhouse gas emissions. Methane emissions from enteric fermentation and manure management represent 27.1 percent and 9.4 percent of total CH₄ emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were the largest emitters of CH₄. Rice cultivation and field burning of agricultural residues were minor sources of CH₄. Emissions of N₂O by agricultural soil management through activities such as fertilizer application and other agricultural practices that increased nitrogen availability in the soil was the largest source of U.S. N₂O emissions, accounting for 75.3 percent. Manure management and field burning

¹ Following the current reporting requirements under the United Nations Framework Convention on Climate Change (UNFCCC), this Inventory report presents CO₂ equivalent values based on the *IPCC Fourth Assessment Report (AR4)* GWP values. See the Introduction chapter for more information.

1 of agricultural residues were also small sources of N₂O emissions. Urea fertilization and liming accounted for 0.10
 2 percent and 0.05 percent of total CO₂ emissions from anthropogenic activities, respectively.

3 Table 5-1 and Table 5-2 present emission estimates for the Agriculture sector. Between 1990 and 2019, CO₂ and
 4 CH₄ emissions from agricultural activities increased by 9.9 percent and 17.5 percent, respectively, while N₂O
 5 emissions from agricultural activities fluctuated from year to year, but increased by 10.4 percent overall. Trends in
 6 sources of agricultural emissions over the 1990 to 2019 time series are shown in Figure 5-2.

7 **Figure 5-2: Trends in Agriculture Chapter Greenhouse Gas Emission Sources**



8
 9 Each year, some emission estimates in the Agriculture sector of the Inventory are recalculated and revised with
 10 improved methods and/or data. In general, recalculations are made to the U.S. greenhouse gas emission estimates
 11 either to incorporate new methodologies or, most commonly, to update recent historical data. These
 12 improvements are implemented consistently across the previous Inventory's time series (i.e., 1990 through 2018)
 13 to ensure that the trend is accurate. This year's notable updates include (1) Enteric Fermentation: default national
 14 emission factors were updated for sheep and goats; (2) Field Burning of Agricultural Residues: updated parameters
 15 within the methodology for combustion efficiency; and (3) Urea Fertilization: updated methodology based on the
 16 analytical solution from the Monte Carlo analysis. In total, the improvements made to the Agriculture sector in this
 17 Inventory increased greenhouse gas emissions by 2.5 MMT CO₂ Eq. (0.4 percent) in 2018. For more information on
 18 specific methodological updates, please see the Recalculations discussions within the respective source category
 19 sections of this chapter.

20 Emissions reported in the Agriculture chapter include those from all states, however, for Hawaii and Alaska some
 21 agricultural practices that can increase nitrogen availability in the soil, and thus cause N₂O emissions, are not
 22 included (see chapter sections on "Uncertainty and Time-Series Consistency" and "Planned Improvements" for
 23 more details). In addition, U.S. Territories are not included with the exception of Urea Fertilization in Puerto Rico.
 24 See Annex 5 for more information on EPA's assessment of the sources not included in this inventory.

25

1 **Table 5-1: Emissions from Agriculture (MMT CO₂ Eq.)**

Gas/Source	1990	2005	2015	2016	2017	2018	2019
CO₂	7.1	7.9	8.5	8.0	8.1	7.4	7.8
Urea Fertilization	2.4	3.5	4.7	4.9	5.1	5.2	5.3
Liming	4.7	4.3	3.7	3.1	3.1	2.2	2.4
CH₄	218.2	239.3	241.4	248.1	251.0	255.7	256.4
Enteric Fermentation	164.7	169.3	166.9	172.2	175.8	178.0	178.6
Manure Management	37.1	51.6	57.9	59.6	59.9	61.7	62.4
Rice Cultivation	16.0	18.0	16.2	15.8	14.9	15.6	15.1
Field Burning of Agricultural Residues	0.4	0.4	0.4	0.4	0.4	0.4	0.4
N₂O	330.1	329.9	366.2	348.4	346.4	357.9	364.4
Agricultural Soil Management	315.9	313.4	348.5	330.1	327.6	338.2	344.6
Manure Management	14.0	16.4	17.5	18.1	18.7	19.4	19.6
Field Burning of Agricultural Residues	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Total	555.3	577.1	616.1	604.4	605.5	621.0	628.6

Note: Totals may not sum due to independent rounding.

2 **Table 5-2: Emissions from Agriculture (kt)**

Gas/Source	1990	2005	2015	2016	2017	2018	2019
CO₂	7,084	7,854	8,464	7,959	8,131	7,440	7,782
Urea Fertilization	2,417	3,504	4,728	4,877	5,051	5,192	5,341
Liming	4,667	4,349	3,737	3,081	3,080	2,248	2,442
CH₄	8,728	9,572	9,656	9,923	10,040	10,226	10,256
Enteric Fermentation	6,588	6,772	6,675	6,890	7,032	7,119	7,142
Manure Management	1,485	2,062	2,316	2,385	2,395	2,467	2,495
Rice Cultivation	640	720	648	631	596	623	602
Field Burning of Agricultural Residues	15	17	18	17	17	17	17
N₂O	1,108	1,107	1,229	1,169	1,162	1,201	1,223
Agricultural Soil Management	1,060	1,052	1,169	1,108	1,099	1,135	1,156
Manure Management	47	55	59	61	63	65	66
Field Burning of Agricultural Residues	1	1	1	1	1	1	1

Note: Totals may not sum due to independent rounding.

3 **Box 5-1: Methodological Approach for Estimating and Reporting U.S. Emissions and Removals**

In following the United Nations Framework Convention on Climate Change (UNFCCC) requirement under Article 4.1 to develop and submit national greenhouse gas emission inventories, the emissions and removals presented in this report and this chapter, are organized by source and sink categories and calculated using internationally-accepted methods provided by the Intergovernmental Panel on Climate Change (IPCC) in the *2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006 IPCC Guidelines)*. Additionally, the calculated emissions and removals in a given year for the United States are presented in a common manner in line with the UNFCCC reporting guidelines for the reporting of inventories under this international agreement. The use of consistent methods to calculate emissions and removals by all nations providing their inventories to the UNFCCC ensures that these reports are comparable. The presentation of emissions provided in the Agriculture chapter do not preclude alternative examinations, but rather, this chapter presents emissions in a common format consistent with how countries are to report inventories under the UNFCCC. The report itself, and this chapter, follows this standardized format, and provides an explanation of the application of methods used to calculate emissions from agricultural activities.

5.1 Enteric Fermentation (CRF Source Category 3A)

Methane is produced as part of normal digestive processes in animals. During digestion, microbes resident in an animal's digestive system ferment food consumed by the animal. This microbial fermentation process, referred to as enteric fermentation, produces CH₄ as a byproduct, which can be exhaled or eructated by the animal. The amount of CH₄ produced and emitted by an individual animal depends primarily upon the animal's digestive system, and the amount and type of feed it consumes.²

Ruminant animals (e.g., cattle, buffalo, sheep, goats, and camels) are the major emitters of CH₄ because of their unique digestive system. Ruminants possess a rumen, or large "fore-stomach," in which microbial fermentation breaks down the feed they consume into products that can be absorbed and metabolized. The microbial fermentation that occurs in the rumen enables them to digest coarse plant material that non-ruminant animals cannot. Ruminant animals, consequently, have the highest CH₄ emissions per unit of body mass among all animal types.

Non-ruminant animals (e.g., swine, horses, and mules and asses) also produce CH₄ emissions through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants emit significantly less CH₄ on a per-animal-mass basis than ruminants because the capacity of the large intestine to produce CH₄ is lower.

In addition to the type of digestive system, an animal's feed quality and feed intake also affect CH₄ emissions. In general, lower feed quality and/or higher feed intake leads to higher CH₄ emissions. Feed intake is positively correlated to animal size, growth rate, level of activity and production (e.g., milk production, wool growth, pregnancy, or work). Therefore, feed intake varies among animal types as well as among different management practices for individual animal types (e.g., animals in feedlots or grazing on pasture).

Methane emission estimates from enteric fermentation are provided in Table 5-3 and Table 5-4. Total livestock CH₄ emissions in 2019 were 178.6 MMT CO₂ Eq. (7,142 kt). Beef cattle remain the largest contributor of CH₄ emissions from enteric fermentation, accounting for 72 percent in 2019. Emissions from dairy cattle in 2019 accounted for 24 percent, and the remaining emissions were from horses, sheep, swine, goats, American bison, mules and asses.³

Table 5-3: CH₄ Emissions from Enteric Fermentation (MMT CO₂ Eq.)

Livestock Type	1990	2005	2015	2016	2017	2018	2019
Beef Cattle	119.1	125.2	118.0	123.0	126.3	128.1	129.1
Dairy Cattle	39.4	37.6	42.6	43.0	43.3	43.6	43.2
Swine	2.0	2.3	2.6	2.6	2.7	2.8	2.9
Horses	1.0	1.7	1.4	1.4	1.3	1.2	1.1
Sheep	2.6	1.4	1.2	1.2	1.2	1.2	1.2
American Bison	0.6	0.7	0.6	0.6	0.6	0.6	0.6
Goats	0.1	0.4	0.4	0.4	0.4	0.4	0.4
Mules and Asses	+	0.1	0.1	0.1	0.1	0.1	0.1
Total	164.7	169.3	166.9	172.2	175.8	178.0	178.6

Note: Totals may not sum due to independent rounding.

+ Does not exceed 0.05 MMT CO₂ Eq.

² CO₂ emissions from livestock are not estimated because annual net CO₂ emissions are assumed to be zero – the CO₂ photosynthesized by plants is returned to the atmosphere as respired CO₂ (IPCC 2006).

³ Enteric fermentation emissions from camels and poultry are not estimated for this Inventory. See Annex 5 for more information on sources and sinks of greenhouse gas emissions not included in this Inventory.

1 **Table 5-4: CH₄ Emissions from Enteric Fermentation (kt)**

Livestock Type	1990	2005	2015	2016	2017	2018	2019
Beef Cattle	4,763	5,007	4,722	4,919	5,052	5,125	5,162
Dairy Cattle	1,574	1,503	1,706	1,722	1,730	1,744	1,729
Swine	81	92	102	105	108	111	115
Horses	40	70	57	54	51	48	46
Sheep	102	55	47	48	47	47	47
American Bison	23	26	24	24	24	24	24
Goats	4	17	14	15	15	15	16
Mules and Asses	1	2	3	3	3	3	3
Total	6,588	6,772	6,675	6,890	7,032	7,119	7,142

Note: Totals may not sum due to independent rounding.

2 From 1990 to 2019, emissions from enteric fermentation have increased by 8.4 percent. From 2018 to 2019,
 3 emissions increased by 0.3 percent, largely driven by an increase in beef cattle populations. While emissions
 4 generally follow trends in cattle populations, over the long term there are exceptions. For example, while dairy
 5 cattle emissions increased 9.8 percent over the entire time series, the population has declined by 3.1 percent, and
 6 milk production increased 58 percent (USDA 2019). These trends indicate that while emissions per head are
 7 increasing, emissions per unit of product (i.e., meat, milk) are decreasing.

8 Generally, from 1990 to 1995 emissions from beef cattle increased and then decreased from 1996 to 2004. These
 9 trends were mainly due to fluctuations in beef cattle populations and increased digestibility of feed for feedlot
 10 cattle. Beef cattle emissions generally increased from 2004 to 2007, as beef cattle populations increased, and an
 11 extensive literature review indicated a trend toward a decrease in feed digestibility for those years. Beef cattle
 12 emissions decreased again from 2007 to 2014, as populations again decreased, but increased from 2015 to 2019,
 13 consistent with another increase in population over those same years. Emissions from dairy cattle generally
 14 trended downward from 1990 to 2004, along with an overall dairy cattle population decline during the same
 15 period. Similar to beef cattle, dairy cattle emissions rose from 2004 to 2007 due to population increases and a
 16 decrease in feed digestibility (based on an analysis of more than 350 dairy cow diets used by producers across the
 17 United States). Dairy cattle emissions have continued to trend upward since 2007, in line with dairy cattle
 18 population increases. Regarding trends in other animals, populations of sheep have steadily declined, with an
 19 overall decrease of 54 percent since 1990. Horse populations are 15 percent greater than they were in 1990, but
 20 their numbers have been declining by an average of 4 percent annually since 2007. Goat populations increased by
 21 about 20 percent through 2007, steadily decreased through 2012, then increased again, by about 1 percent
 22 annually, through 2019. Swine populations have trended upward through most of the time series, increasing 43
 23 percent from 1990 to 2019. The population of American bison more than tripled over the 1990 to 2019 time
 24 period, while the population of mules and asses increased by a factor of four.

25 Methodology

26 Livestock enteric fermentation emission estimate methodologies fall into two categories: cattle and other
 27 domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics,
 28 account for the majority of enteric fermentation CH₄ emissions from livestock in the United States. A more detailed
 29 methodology (i.e., IPCC Tier 2) was therefore applied to estimate emissions for all cattle. Emission estimates for
 30 other domesticated animals (horses, sheep, swine, goats, American bison, and mules and asses) were estimated
 31 using the IPCC Tier 1 approach, as suggested by the 2006 IPCC Guidelines.

32 While the large diversity of animal management practices cannot be precisely characterized and evaluated,
 33 significant scientific literature exists that provides the necessary data to estimate cattle emissions using the IPCC
 34 Tier 2 approach. The Cattle Enteric Fermentation Model (CEFM), developed by EPA and used to estimate cattle CH₄
 35 emissions from enteric fermentation, incorporates this information and other analyses of livestock population,
 36 feeding practices, and production characteristics. For the current Inventory, CEFM results for 1990 through 2017

1 were carried over from the 1990 to 2017 Inventory (i.e., 2019 Inventory submission) to focus resources on CEFM
2 improvements, and a simplified approach was used to estimate 2018 and 2019 enteric emissions from cattle.
3 See Annex 3.10 for more detailed information on the methodology and data used to calculate CH₄ emissions from
4 enteric fermentation. In addition, variables and the resulting emissions are also available at the state level in Annex
5 3.10.

6 *1990 to 2017 Inventory Methodology for Cattle*

7 National cattle population statistics were disaggregated into the following cattle sub-populations:

- 8 • Dairy Cattle
 - 9 ○ Calves
 - 10 ○ Heifer Replacements
 - 11 ○ Cows
- 12 • Beef Cattle
 - 13 ○ Calves
 - 14 ○ Heifer Replacements
 - 15 ○ Heifer and Steer Stockers
 - 16 ○ Animals in Feedlots (Heifers and Steer)
 - 17 ○ Cows
 - 18 ○ Bulls

19 Calf birth rates, end-of-year population statistics, detailed feedlot placement information, and slaughter weight
20 data were used to create a transition matrix that models cohorts of individual animal types and their specific
21 emission profiles. The key variables tracked for each of the cattle population categories are described in Annex
22 3.10. These variables include performance factors such as pregnancy and lactation as well as average weights and
23 weight gain. Annual cattle population data were obtained from the U.S. Department of Agriculture's (USDA)
24 National Agricultural Statistics Service (NASS) *QuickStats* database (USDA 2016).

25 Diet characteristics were estimated by region for dairy, grazing beef, and feedlot beef cattle. These diet
26 characteristics were used to calculate digestible energy (DE) values (expressed as the percent of gross energy
27 intake digested by the animal) and CH₄ conversion rates (Y_m) (expressed as the fraction of gross energy converted
28 to CH₄) for each regional population category. The IPCC recommends Y_m ranges of 3.0 ± 1.0 percent for feedlot
29 cattle and 6.5 ± 1.0 percent for other well-fed cattle consuming temperate-climate feed types (IPCC 2006). Given
30 the availability of detailed diet information for different regions and animal types in the United States, DE and Y_m
31 values unique to the United States were developed. The diet characterizations and estimation of DE and Y_m values
32 were based on information from state agricultural extension specialists, a review of published forage quality
33 studies and scientific literature, expert opinion, and modeling of animal physiology.

34 The diet characteristics for dairy cattle were based on Donovan (1999) and an extensive review of nearly 20 years
35 of literature from 1990 through 2009. Estimates of DE were national averages based on the feed components of
36 the diets observed in the literature for the following year groupings: 1990 through 1993, 1994 through 1998, 1999
37 through 2003, 2004 through 2006, 2007, and 2008 onward.⁴ Base year Y_m values by region were estimated using
38 Donovan (1999). As described in ERG (2016), a ruminant digestion model (COWPOLL, as selected in Kebreab et al.
39 2008) was used to evaluate Y_m for each diet evaluated from the literature, and a function was developed to adjust
40 regional values over time based on the national trend. Dairy replacement heifer diet assumptions were based on
41 the observed relationship in the literature between dairy cow and dairy heifer diet characteristics.

⁴ Due to inconsistencies in the 2003 literature values, the 2002 values were used for 2003 as well.

1 For feedlot animals, the DE and Y_m values used for 1990 were recommended by Johnson (1999). Values for DE and
 2 Y_m for 1991 through 1999 were linearly extrapolated based on the 1990 and 2000 data. DE and Y_m values for 2000
 3 onwards were based on survey data in Galyean and Gleghorn (2001) and Vasconcelos and Galyean (2007).

4 For grazing beef cattle, Y_m values were based on Johnson (2002), DE values for 1990 through 2006 were based on
 5 specific diet components estimated from Donovan (1999), and DE values from 2007 onwards were developed from
 6 an analysis by Archibeque (2011), based on diet information in Preston (2010) and USDA-APHIS:VS (2010). Weight
 7 and weight gains for cattle were estimated from Holstein (2010), Doren et al. (1989), Enns (2008), Lippke et al.
 8 (2000), Pinchack et al. (2004), Platter et al. (2003), Skogerboe et al. (2000), and expert opinion. See Annex 3.10 for
 9 more details on the method used to characterize cattle diets and weights in the United States.

10 Calves younger than 4 months are not included in emission estimates because calves consume mainly milk and the
 11 IPCC recommends the use of a Y_m of zero for all juveniles consuming only milk. Diets for calves aged 4 to 6 months
 12 are assumed to go through a gradual weaning from milk decreasing to 75 percent at 4 months, 50 percent at age 5
 13 months, and 25 percent at age 6 months. The portion of the diet made up with milk still results in zero emissions.
 14 For the remainder of the diet, beef calf DE and Y_m are set equivalent to those of beef replacement heifers, while
 15 dairy calf DE is set equal to that of dairy replacement heifers and dairy calf Y_m is provided at 4 and 7 months of age
 16 by Soliva (2006). Estimates of Y_m for 5 and 6 month old dairy calves are linearly interpolated from the values
 17 provided for 4 and 7 months.

18 To estimate CH₄ emissions, the population was divided into state, age, sub-type (i.e., dairy cows and replacements,
 19 beef cows and replacements, heifer and steer stockers, heifers and steers in feedlots, bulls, beef calves 4 to 6
 20 months, and dairy calves 4 to 6 months), and production (i.e., pregnant, lactating) groupings to more fully capture
 21 differences in CH₄ emissions from these animal types. The transition matrix was used to simulate the age and
 22 weight structure of each sub-type on a monthly basis in order to more accurately reflect the fluctuations that
 23 occur throughout the year. Cattle diet characteristics were then used in conjunction with Tier 2 equations from
 24 IPCC (2006) to produce CH₄ emission factors for the following cattle types: dairy cows, beef cows, dairy
 25 replacements, beef replacements, steer stockers, heifer stockers, steer feedlot animals, heifer feedlot animals,
 26 bulls, and calves. To estimate emissions from cattle, monthly population data from the transition matrix were
 27 multiplied by the calculated emission factor for each cattle type. More details are provided in Annex 3.10.

28 **2018 and 2019 Inventory Methodology for Cattle**

29 As noted above, a simplified approach for cattle enteric emissions was used in lieu of the CEFM for 2018 and 2019.
 30 First, 2018 and 2019 populations for each of the CEFM cattle sub-populations were estimated, then these
 31 populations were multiplied by the corresponding implied emission factors developed from the CEFM for the 1990
 32 to 2017 Inventory year. Dairy cow, beef cow, and bull populations for 2019 were based on data directly from the
 33 USDA-NASS *QuickStats* database (USDA 2020, USDA 2019). Because the remaining CEFM cattle sub-population
 34 categories do not correspond exactly to the remaining *QuickStats* cattle categories, 2018 and 2019 populations for
 35 these categories were estimated by extrapolating the 2017 populations based on percent changes from 2017 to
 36 2018 and 2018 to 2019 in similar *QuickStats* categories, consistent with Volume 1, Chapter 5 of the *2006 IPCC*
 37 *Guidelines* on time-series consistency. Table 5-5 lists the *QuickStats* categories used to estimate the percent
 38 change in population for each of the CEFM categories.

39 **Table 5-5: Cattle Sub-Population Categories for 2018 Population Estimates**

CEFM Cattle Category	USDA-NASS <i>QuickStats</i> Cattle Category
Dairy Calves	Cattle, Calves
Dairy Cows	Cattle, Cows, Milk
Dairy Replacements 7-11 months	Cattle, Heifers, GE 500 lbs, Milk Replacement
Dairy Replacements 12-23 months	Cattle, Heifers, GE 500 lbs, Milk Replacement
Bulls	Cattle, Bulls, GE 500 lbs
Beef Calves	Cattle, Calves
Beef Cows	Cattle, Cows, Beef
Beef Replacements 7-11 months	Cattle, Heifers, GE 500 lbs, Beef Replacement

Beef Replacements 12-23 months	Cattle, Heifers, GE 500 lbs, Beef Replacement
Steer Stockers	Cattle, Steers, GE 500 lbs
Heifer Stockers	Cattle, Heifers, GE 500 lbs, (Excl. Replacement)
Steer Feedlot	Cattle, On Feed
Heifer Feedlot	Cattle, On Feed

1 *Non-Cattle Livestock*

2 Emission estimates for other animal types were based on average emission factors (Tier 1 default IPCC emission
3 factors) representative of entire populations of each animal type. Methane emissions from these animals
4 accounted for a minor portion of total CH₄ emissions from livestock in the United States from 1990 through 2019.
5 Additionally, the variability in emission factors for each of these other animal types (e.g., variability by age,
6 production system, and feeding practice within each animal type) is less than that for cattle.

7 Annual livestock population data for 1990 to 2019 for sheep; swine; goats; horses; mules and asses; and American
8 bison were obtained for available years from USDA-NASS (USDA 2016). Horse, goat and mule and ass population
9 data were available for 1987, 1992, 1997, 2002, 2007, and 2012 (USDA 1992, 1997, 2016); the remaining years
10 between 1990 and 2019 were interpolated and extrapolated from the available estimates (with the exception of
11 goat populations being held constant between 1990 and 1992). American bison population estimates were
12 available from USDA for 2002, 2007, and 2012 (USDA 2016) and from the National Bison Association (1999) for
13 1990 through 1999. Additional years were based on observed trends from the National Bison Association (1999),
14 interpolation between known data points, and extrapolation beyond 2012, as described in more detail in Annex
15 3.10.

16 Methane emissions from sheep, goats, swine, horses, American bison, and mules and asses were estimated by
17 using emission factors utilized in Crutzen et al. (1986, cited in IPCC 2006). These emission factors are
18 representative of typical animal sizes, feed intakes, and feed characteristics in developed countries. For American
19 bison the emission factor for buffalo was used and adjusted based on the ratio of live weights to the 0.75 power.
20 The methodology is the same as that recommended by IPCC (2006).

21 **Uncertainty and Time-Series Consistency**

22 A quantitative uncertainty analysis for this source category was performed using the IPCC-recommended Approach
23 2 uncertainty estimation methodology based on a Monte Carlo Stochastic Simulation technique as described in ICF
24 (2003). These uncertainty estimates were developed for the 1990 through 2001 Inventory (i.e., 2003 submission to
25 the UNFCCC). While there are plans to update the uncertainty to reflect recent methodological updates and
26 forthcoming changes (see Planned Improvements, below), at this time the uncertainty estimates were directly
27 applied to the 2019 emission estimates in this Inventory.

28 A total of 185 primary input variables (177 for cattle and 8 for non-cattle) were identified as key input variables for
29 the uncertainty analysis. A normal distribution was assumed for almost all activity- and emission factor-related
30 input variables. Triangular distributions were assigned to three input variables (specifically, cow-birth ratios for the
31 three most recent years included in the 2001 model run) to ensure only positive values would be simulated. For
32 some key input variables, the uncertainty ranges around their estimates (used for inventory estimation) were
33 collected from published documents and other public sources; others were based on expert opinion and best
34 estimates. In addition, both endogenous and exogenous correlations between selected primary input variables
35 were modeled. The exogenous correlation coefficients between the probability distributions of selected activity-
36 related variables were developed through expert judgment.

37 The uncertainty ranges associated with the activity data-related input variables were plus or minus 10 percent or
38 lower. However, for many emission factor-related input variables, the lower- and/or the upper-bound uncertainty
39 estimates were over 20 percent. The results of the quantitative uncertainty analysis are summarized in Table 5-6.
40 Based on this analysis, enteric fermentation CH₄ emissions in 2019 were estimated to be between 158.9 and 210.7
41 MMT CO₂ Eq. at a 95 percent confidence level, which indicates a range of 11 percent below to 18 percent above

1 the 2019 emission estimate of 178.6 MMT CO₂ Eq. Among the individual cattle sub-source categories, beef cattle
 2 account for the largest amount of CH₄ emissions, as well as the largest degree of uncertainty in the emission
 3 estimates—due mainly to the difficulty in estimating the diet characteristics for grazing members of this animal
 4 group. Among non-cattle, horses represent the largest percent of uncertainty in the previous uncertainty analysis
 5 because the Food and Agricultural Organization of the United Nations (FAO) population estimates used for horses
 6 at that time had a higher degree of uncertainty than for the USDA population estimates used for swine, goats, and
 7 sheep. The horse populations are now from the same USDA source as the other animal types, and therefore the
 8 uncertainty range around horses is likely overestimated. Cattle calves, American bison, mules and asses were
 9 excluded from the initial uncertainty estimate because they were not included in emission estimates at that time.

10 **Table 5-6: Approach 2 Quantitative Uncertainty Estimates for CH₄ Emissions from Enteric**
 11 **Fermentation (MMT CO₂ Eq. and Percent)**

Source	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^{a, b, c}			
			(MMT CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Enteric Fermentation	CH ₄	178.6	158.9	210.7	-11%	+18%

^a Range of emissions estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

^b Note that the relative uncertainty range was estimated with respect to the 2001 emission estimates from the 2003 submission and applied to the 2019 estimates.

^c The overall uncertainty calculated in 2003, and applied to the 2019 emission estimate, did not include uncertainty estimates for calves, American bison, and mules and asses. Additionally, for bulls the emissions estimate was based on the Tier 1 methodology. Since bull emissions are now estimated using the Tier 2 method, the uncertainty surrounding their estimates is likely lower than indicated by the previous uncertainty analysis.

12 Methodological approaches, changes to historic data, and other parameters were applied to the entire time series
 13 to ensure consistency in emissions estimates from 1990 through 2019. Details on the emission trends and
 14 methodologies through time are described in more detail in the Introduction and Methodology sections.

15 QA/QC and Verification

16 In order to ensure the quality of the emission estimates from enteric fermentation, the General (IPCC Tier 1) and
 17 category-specific (Tier 2) Quality Assurance/Quality Control (QA/QC) procedures were implemented consistent
 18 with the U.S. Inventory QA/QC plan outlined in Annex 8. Category-specific or Tier 2 QA procedures included
 19 independent review of emission estimate methodologies from previous inventories.

20 Over the past few years, particular importance has been placed on harmonizing the data exchange between the
 21 enteric fermentation and manure management source categories. The current Inventory now utilizes the transition
 22 matrix from the CEFM for estimating cattle populations and weights for both source categories, and the CEFM is
 23 used to output volatile solids and nitrogen excretion estimates using the diet assumptions in the model in
 24 conjunction with the energy balance equations from the IPCC (2006). This approach facilitates the QA/QC process
 25 for both of these source categories. As noted in the Methodology discussion above, a simplified approach for cattle
 26 enteric emissions was used in lieu of the CEFM for 2018 and 2019.

27 Recalculations Discussion

28 For sheep and goats, default national emission factors were updated to reflect revisions made in the *2019 IPCC*
 29 *Refinement to the 2006 IPCC Guidelines*. These revised emission factors were applied to the entire time series and
 30 resulted in between 5 to 12 kt and 9 to 12 kt higher emissions for sheep and goat livestock categories, respectively.

1 Planned Improvements

2 Regular annual data reviews and updates are necessary to maintain an emissions inventory that reflects the
3 current base of knowledge. EPA conducts the following list of regular annual assessments of data availability when
4 updating the estimates to extend time series each year:

- 5 • Further research to improve the estimation of dry matter intake (as gross energy intake) using data from
6 appropriate production systems;
- 7 • Updating input variables that are from older data sources, such as beef births by month, beef and dairy
8 annual calving rates, and beef cow lactation rates;
- 9 • Investigating the availability of data for dairy births by month, to replace the current assumption that
10 births are evenly distributed throughout the year;
- 11 • Updating the diet data to incorporate monthly or annual milk fat data in place of the fixed IPCC default
12 value of 4 percent milk fat. EPA has investigated the availability of data across the time series and plans to
13 incorporate annual U.S. milk fat values into the CEFM calculations in the next (i.e., 1990 to 2020)
14 Inventory, as opposed to using a default 4 percent milk fat across the entire time series;
- 15 • Investigating the availability of annual data for the DE, Y_m , and crude protein values of specific diet and
16 feed components for grazing and feedlot animals;
- 17 • Further investigation on additional sources or methodologies for estimating DE for dairy cattle, given the
18 many challenges in characterizing dairy cattle diets;
- 19 • Further evaluation of the assumptions about weights and weight gains for beef cows, such that trends
20 beyond 2007 are updated, rather than held constant;
- 21 • Further evaluation of the estimated weight for dairy cows (i.e., 1,500 lbs) that is based solely on Holstein
22 cows as mature dairy cow weight is likely slightly overestimated, based on knowledge of the breeds of
23 dairy cows in the United States.

24 Depending upon the outcome of ongoing investigations, future improvement efforts for enteric fermentation
25 could include some of the following options which are additional to the regular updates, and may or may have
26 implications for regular updates once addressed:

- 27 • Potentially updating to a Tier 2 methodology for other animal types (i.e., sheep, swine, goats, horses);
- 28 • Investigation of methodologies and emission factors for including enteric fermentation emission
29 estimates from poultry;
- 30 • Comparison of the current CEFM processing of animal population data to estimates developed using
31 annual average populations to determine if the model could be simplified to use annual population data;
- 32 • Comparison of the current CEFM with other models that estimate enteric fermentation emissions for
33 quality assurance and verification;
- 34 • Investigation of recent research implications suggesting that certain parameters in enteric models may be
35 simplified without significantly diminishing model accuracy;
- 36 • Recent changes that have been implemented to the CEFM warrant an assessment of the current
37 uncertainty analysis; therefore, a revision of the quantitative uncertainty surrounding emission estimates
38 from this source category will be initiated; and
- 39 • Analysis and integration of a more representative spatial distribution of animal populations by state,
40 particularly for poultry animal populations.

41 EPA received comments during the Expert Review period of the current (i.e., 1990 through 2019) Inventory to
42 provide additional explanation as to specific animal emissions trends. EPA is currently assessing how best to
43 incorporate an updated trends discussion.

1 EPA received comments during the previous two Public Review periods of the Inventory regarding the CEFM model
2 and data and assumptions used to calculate enteric fermentation cattle emissions. Many of the comments
3 received are consistent with potential planned improvement options listed above. EPA is continuously
4 investigating these recommendations and potential improvements and working with USDA and other experts to
5 utilize the best available data and methods for estimating emissions. Many of these improvements are major
6 updates and may take multiple years to implement in full. In addition, EPA received comments during the Public
7 Review period of the previous (1990 through 2018) Inventory regarding the use of alternate metrics for weighting
8 non-CO₂ emissions such as methane that differ from those required in reporting under the UNFCCC to facilitate
9 comparability as described in Box 5-1.

10 5.2 Manure Management (CRF Source 11 Category 3B)

12 The treatment, storage, and transportation of livestock manure can produce anthropogenic CH₄ and N₂O
13 emissions.⁵ Methane is produced by the anaerobic decomposition of manure and nitrous oxide is produced from
14 direct and indirect pathways through the processes of nitrification and denitrification; in addition, there are many
15 underlying factors that can affect these resulting emissions from manure management, as described below.

16 When livestock manure is stored or treated in systems that promote anaerobic conditions (e.g., as a liquid/slurry in
17 lagoons, ponds, tanks, or pits), the decomposition of the volatile solids component in the manure tends to produce
18 CH₄. When manure is handled as a solid (e.g., in stacks or drylots) or deposited on pasture, range, or paddock
19 lands, it tends to decompose aerobically and produce CO₂ and little or no CH₄. Ambient temperature, moisture,
20 and manure storage or residency time affect the amount of CH₄ produced because they influence the growth of
21 the bacteria responsible for CH₄ formation. For non-liquid-based manure systems, moist conditions (which are a
22 function of rainfall and humidity) can promote CH₄ production. Manure composition, which varies by animal diet,
23 growth rate, and animal type (particularly the different animal digestive systems), also affects the amount of CH₄
24 produced. In general, the greater the energy content of the feed, the greater the potential for CH₄ emissions.
25 However, some higher-energy feeds also are more digestible than lower quality forages, which can result in less
26 overall waste excreted from the animal.

27 As previously stated, N₂O emissions are produced through both direct and indirect pathways. Direct N₂O emissions
28 are produced as part of the nitrogen (N) cycle through the nitrification and denitrification of the N in livestock dung
29 and urine.⁶ There are two pathways for indirect N₂O emissions. The first is the result of the volatilization of N in
30 manure (as NH₃ and NO_x) and the subsequent deposition of these gases and their products (NH₄⁺ and NO₃⁻) onto
31 soils and the surface of lakes and other waters. The second pathway is the runoff and leaching of N from manure
32 into the groundwater below, into riparian zones receiving drain or runoff water, or into the ditches, streams,
33 rivers, and estuaries into which the land drainage water eventually flows.

34 The production of direct N₂O emissions from livestock manure depends on the composition of the manure
35 (manure includes both feces and urine), the type of bacteria involved in the process, and the amount of oxygen
36 and liquid in the manure system. For direct N₂O emissions to occur, the manure must first be handled aerobically
37 where organic N is mineralized or decomposed to NH₄ which is then nitrified to NO₃ (producing some N₂O as a
38 byproduct) (nitrification). Next, the manure must be handled anaerobically where the nitrate is then denitrified to

⁵ CO₂ emissions from livestock are not estimated because annual net CO₂ emissions are assumed to be zero – the CO₂ photosynthesized by plants is returned to the atmosphere as respired CO₂ (IPCC 2006).

⁶ Direct and indirect N₂O emissions from dung and urine spread onto fields either directly as daily spread or after it is removed from manure management systems (i.e., lagoon, pit, etc.) and from livestock dung and urine deposited on pasture, range, or paddock lands are accounted for and discussed in the Agricultural Soil Management source category within the Agriculture sector.

1 N₂O and N₂ (denitrification). NO_x can also be produced during denitrification (Groffman et al. 2000; Robertson and
2 Groffman 2015). These emissions are most likely to occur in dry manure handling systems that have aerobic
3 conditions, but that also contain pockets of anaerobic conditions due to saturation. A very small portion of the
4 total N excreted is expected to convert to N₂O in the waste management system (WMS).

5 Indirect N₂O emissions are produced when nitrogen is lost from the system through volatilization (as NH₃ or NO_x)
6 or through runoff and leaching. The vast majority of volatilization losses from these operations are NH₃. Although
7 there are also some small losses of NO_x, there are no quantified estimates available for use, so losses due to
8 volatilization are only based on NH₃ loss factors. Runoff losses would be expected from operations that house
9 animals or store manure in a manner that is exposed to weather. Runoff losses are also specific to the type of
10 animal housed on the operation due to differences in manure characteristics. Little information is known about
11 leaching from manure management systems as most research focuses on leaching from land application systems.
12 Since leaching losses are expected to be minimal, leaching losses are coupled with runoff losses and the
13 runoff/leaching estimate provided in this chapter does not account for any leaching losses.

14 Estimates of CH₄ emissions from manure management in 2019 were 62.4 MMT CO₂ Eq. (2,495 kt); in 1990,
15 emissions were 37.1 MMT CO₂ Eq. (1,485 kt). This represents a 68 percent increase in emissions from 1990.
16 Emissions increased on average by 0.8 MMT CO₂ Eq. (2.3 percent) annually over this period. The majority of this
17 increase is due to swine and dairy cow manure, where emissions increased 49 and 119 percent, respectively. From
18 2018 to 2019, there was a 1.1 percent increase in total CH₄ emissions from manure management, due to an
19 increase in animal populations.

20 Although a large quantity of managed manure in the United States is handled as a solid, producing little CH₄, the
21 general trend in manure management, particularly for dairy cattle and swine (which are both shifting towards
22 larger facilities), is one of increasing use of liquid systems. Also, new regulations controlling the application of
23 manure nutrients to land have shifted manure management practices at smaller dairies from daily spread systems
24 to storage and management of the manure on site. In many cases, manure management systems with the most
25 substantial methane emissions are those associated with confined animal management operations where manure
26 is handled in liquid-based systems. Nitrous oxide emissions from manure management vary significantly between
27 the types of management system used and can also result in indirect emissions due to other forms of nitrogen loss
28 from the system (IPCC 2006).

29 While national dairy animal populations have decreased since 1990, some states have seen increases in their dairy
30 cattle populations as the industry becomes more concentrated in certain areas of the country and the number of
31 animals contained on each facility increases. These areas of concentration, such as California, New Mexico, and
32 Idaho, tend to utilize more liquid-based systems to manage (flush or scrape) and store manure. Thus, the shift
33 toward larger dairy cattle and swine facilities since 1990 has translated into an increasing use of liquid manure
34 management systems, which have higher potential CH₄ emissions than dry systems. This significant shift in both
35 the dairy cattle and swine industries was accounted for by incorporating state and WMS-specific CH₄ conversion
36 factor (MCF) values in combination with the 1992, 1997, 2002, 2007, 2012, and 2017 farm-size distribution data
37 reported in the U.S. Department of Agriculture (USDA) *Census of Agriculture* (USDA 2019d).

38 In 2019, total N₂O emissions from manure management were estimated to be 19.6 MMT CO₂ Eq. (66 kt); in 1990,
39 emissions were 14.0 MMT CO₂ Eq. (47 kt). These values include both direct and indirect N₂O emissions from
40 manure management. Nitrous oxide emissions have increased since 1990. Small changes in N₂O emissions from
41 individual animal groups exhibit the same trends as the animal group populations, with the overall net effect that
42 N₂O emissions showed a 40 percent increase from 1990 to 2019 and a 0.9 percent increase from 2018 through
43 2019. Overall shifts toward liquid systems have driven down the emissions per unit of nitrogen excreted as dry
44 manure handling systems have greater aerobic conditions that promote N₂O emissions.

1 Table 5-7 and Table 5-8 provide estimates of CH₄ and N₂O emissions from manure management by animal
 2 category.⁷

3 **Table 5-7: CH₄ and N₂O Emissions from Manure Management (MMT CO₂ Eq.)**

Gas/Animal Type	1990	2005	2015	2016	2017	2018	2019
CH₄^a	37.1	51.6	57.9	59.6	59.9	61.7	62.4
Dairy Cattle	14.7	24.3	30.8	31.5	31.8	32.3	32.0
Swine	15.5	20.3	20.2	21.1	21.0	22.2	23.1
Poultry	3.3	3.2	3.4	3.4	3.4	3.5	3.6
Beef Cattle	3.1	3.3	3.1	3.3	3.4	3.4	3.4
Horses	0.2	0.3	0.2	0.2	0.2	0.2	0.2
Sheep	0.2	0.1	0.1	0.1	0.1	0.1	0.1
Goats	+	+	+	+	+	+	+
American Bison	+	+	+	+	+	+	+
Mules and Asses	+	+	+	+	+	+	+
N₂O^b	14.0	16.4	17.5	18.1	18.7	19.4	19.6
Beef Cattle	5.9	7.2	7.7	8.1	8.6	9.2	9.4
Dairy Cattle	5.3	5.5	6.0	6.1	6.1	6.1	6.1
Swine	1.2	1.6	1.8	1.9	2.0	2.0	2.1
Poultry	1.4	1.6	1.6	1.6	1.6	1.7	1.7
Sheep	0.1	0.3	0.3	0.3	0.3	0.3	0.3
Horses	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Goats	+	+	+	+	+	+	+
Mules and Asses	+	+	+	+	+	+	+
American Bison ^c	NA						
Total	51.1	67.9	75.4	77.7	78.5	81.1	82.0

Notes: N₂O emissions from manure deposited on pasture, range and paddock are included in the Agricultural Soils Management sector. Totals may not sum due to independent rounding.

+ Does not exceed 0.05 MMT CO₂ Eq.

NA (Not Available)

^a Accounts for CH₄ reductions due to capture and destruction of CH₄ at facilities using anaerobic digesters.

^b Includes both direct and indirect N₂O emissions.

^c There are no American bison N₂O emissions from managed systems; American bison are maintained entirely on pasture, range, and paddock.

4 **Table 5-8: CH₄ and N₂O Emissions from Manure Management (kt)**

Gas/Animal Type	1990	2005	2015	2016	2017	2018	2019
CH₄^a	1,485	2,062	2,316	2,385	2,395	2,467	2,495
Dairy Cattle	589	970	1,233	1,259	1,270	1,292	1,281
Swine	622	812	808	846	840	888	924
Poultry	131	129	136	136	137	141	142
Beef Cattle	126	133	126	132	136	135	136
Horses	9	12	8	8	7	7	7
Sheep	7	3	3	3	3	3	3
Goats	1	1	1	1	1	1	1
American Bison	+	+	+	+	+	+	+
Mules and Asses	+	+	+	+	+	+	+
N₂O^b	47	55	59	61	63	65	66
Beef Cattle	20	24	26	27	29	31	31
Dairy Cattle	18	18	20	20	20	21	20

⁷ Manure management emissions from camels are not estimated for this Inventory. See Annex 5 for more information on sources and sinks of greenhouse gas emissions not included in this Inventory.

Swine	4	5	6	6	7	7	7
Poultry	5	5	5	5	5	6	6
Sheep	+	1	1	1	1	1	1
Horses	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+
Mules and Asses	+	+	+	+	+	+	+
American Bison ^c	NA						

Notes: N₂O emissions from manure deposited on pasture, range and paddock are included in the Agricultural Soils Management sector. Totals may not sum due to independent rounding.

+ Does not exceed 0.5 kt.

NA (Not Available)

^aAccounts for CH₄ reductions due to capture and destruction of CH₄ at facilities using anaerobic digesters.

^bIncludes both direct and indirect N₂O emissions.

^cThere are no American bison N₂O emissions from managed systems; American bison are maintained entirely on pasture, range, and paddock.

1 Methodology

2 The methodologies presented in IPCC (2006) form the basis of the CH₄ and N₂O emission estimates for each animal
3 type, including Tier 1, Tier 2, and use of the CEFM previously described for Enteric Fermentation. This section
4 presents a summary of the methodologies used to estimate CH₄ and N₂O emissions from manure management.
5 For the current Inventory, time-series results were carried over from the 1990 to 2018 Inventory (i.e., 2020
6 submission) and a simplified approach was used to estimate manure management emissions for 2019. See Annex
7 3.11 for more detailed information on the methodology, data used to calculate CH₄ and N₂O emissions, and
8 emission results (including input variables and results at the state-level) from manure management.

9 Methane Calculation Methods

10 The following inputs were used in the calculation of manure management CH₄ emissions for 1990 through 2018:

- 11 • Animal population data (by animal type and state);
- 12 • Typical animal mass (TAM) data (by animal type);
- 13 • Portion of manure managed in each WMS, by state and animal type;
- 14 • Volatile solids (VS) production rate (by animal type and state or United States);
- 15 • Methane producing potential (B₀) of the volatile solids (by animal type); and
- 16 • Methane conversion factors (MCF), the extent to which the CH₄ producing potential is realized for each
17 type of WMS (by state and manure management system, including the impacts of any biogas collection
18 efforts).

19 Methane emissions were estimated by first determining activity data, including animal population, TAM, WMS
20 usage, and waste characteristics. The activity data sources are described below:

- 21 • Annual animal population data for 1990 through 2018 for all livestock types, except goats, horses, mules
22 and asses, and American bison were obtained from the USDA-NASS. For cattle, the USDA populations
23 were utilized in conjunction with birth rates, detailed feedlot placement information, and slaughter
24 weight data to create the transition matrix in the Cattle Enteric Fermentation Model (CEFM) that models
25 cohorts of individual animal types and their specific emission profiles. The key variables tracked for each
26 of the cattle population categories are described in Section 5.1 and in more detail in Annex 3.10. Goat
27 population data for 1992, 1997, 2002, 2007, 2012, and 2017; horse and mule and ass population data for
28 1987, 1992, 1997, 2002, 2007, 2012, and 2017; and American bison population for 2002, 2007, 2012, and
29 2017 were obtained from the *Census of Agriculture* (USDA 2019d). American bison population data for
30 1990 through 1999 were obtained from the National Bison Association (1999).
- 31 • The TAM is an annual average weight that was obtained for animal types other than cattle from
32 information in USDA's *Agricultural Waste Management Field Handbook* (USDA 1996), the American

1 Society of Agricultural Engineers, Standard D384.1 (ASAE 1998) and others (Meagher 1986; EPA 1992;
2 Safley 2000; ERG 2003b; IPCC 2006; ERG 2010a). For a description of the TAM used for cattle, see Annex
3 3.10.

- 4 • WMS usage was estimated for swine and dairy cattle for different farm size categories using state and
5 regional data from USDA (USDA APHIS 1996; Bush 1998; Ott 2000; USDA 2016c) and EPA (ERG 2000a; EPA
6 2002a and 2002b; ERG 2018, ERG 2019). For beef cattle and poultry, manure management system usage
7 data were not tied to farm size but were based on other data sources (ERG 2000a; USDA APHIS 2000; UEP
8 1999). For other animal types, manure management system usage was based on previous estimates (EPA
9 1992). American bison WMS usage was assumed to be the same as not on feed (NOF) cattle, while mules
10 and asses were assumed to be the same as horses.
- 11 • VS production rates for all cattle except for calves were calculated by head for each state and animal type
12 in the CEFM. VS production rates by animal mass for all other animals were determined using data from
13 USDA's *Agricultural Waste Management Field Handbook* (USDA 1996 and 2008; ERG 2010b and 2010c)
14 and data that was not available in the most recent *Handbook* were obtained from the American Society of
15 Agricultural Engineers, Standard D384.1 (ASAE 1998) or the *2006 IPCC Guidelines* (IPCC 2006). American
16 bison VS production was assumed to be the same as NOF bulls.
- 17 • B_0 was determined for each animal type based on literature values (Morris 1976; Bryant et al. 1976;
18 Hashimoto 1981; Hashimoto 1984; EPA 1992; Hill 1982; Hill 1984).
- 19 • MCFs for dry systems were set equal to default IPCC factors based on state climate for each year (IPCC
20 2006). MCFs for liquid/slurry, anaerobic lagoon, and deep pit systems were calculated based on the
21 forecast performance of biological systems relative to temperature changes as predicted in the van't Hoff-
22 Arrhenius equation which is consistent with IPCC (2006) Tier 2 methodology.
- 23 • Data from anaerobic digestion systems with CH_4 capture and combustion were obtained from the EPA
24 AgSTAR Program, including information available in the AgSTAR project database (EPA 2019). Anaerobic
25 digester emissions were calculated based on estimated methane production and collection and
26 destruction efficiency assumptions (ERG 2008).
- 27 • For all cattle except for calves, the estimated amount of VS (kg per animal-year) managed in each WMS
28 for each animal type, state, and year were taken from the CEFM, assuming American bison VS production
29 to be the same as NOF bulls. For animals other than cattle, the annual amount of VS (kg per year) from
30 manure excreted in each WMS was calculated for each animal type, state, and year. This calculation
31 multiplied the animal population (head) by the VS excretion rate (kg VS per 1,000 kg animal mass per
32 day), the TAM (kg animal mass per head) divided by 1,000, the WMS distribution (percent), and the
33 number of days per year (365.25).

34 The estimated amount of VS managed in each WMS was used to estimate the CH_4 emissions (kg CH_4 per year) from
35 each WMS. The amount of VS (kg per year) were multiplied by the B_0 (m^3 CH_4 per kg VS), the MCF for that WMS
36 (percent), and the density of CH_4 (kg CH_4 per m^3 CH_4). The CH_4 emissions for each WMS, state, and animal type
37 were summed to determine the total U.S. CH_4 emissions.

38 The following approach was used in the calculation of manure management CH_4 emissions for 2019:

- 39 • EPA obtained 2019 national-level animal population data. Sheep, poultry, and swine data were
40 downloaded from USDA-NASS Quickstats (USDA 2020). Cattle populations were obtained from the CEFM
41 (see NIR Section 5.1 and Annex 3.10). Data for goats, horses, bison, mules, and asses were extrapolated
42 based on the 2009 through 2018 population values to reflect recent trends in animal populations.

- 1 • EPA multiplied the national populations by the animal-specific 2018 implied emission factors⁸ for CH₄ to
2 calculate national-level 2019 CH₄ emissions estimates by animal type. These methods were utilized in
3 order to maintain time-series consistency as referenced in Volume 1, Chapter 5 of the *2006 IPCC*
4 *Guidelines*.

5 Nitrous Oxide Calculation Methods

6 The following inputs were used in the calculation of direct and indirect manure management N₂O emissions for
7 1990 through 2018:

- 8 • Animal population data (by animal type and state);
9 • TAM data (by animal type);
10 • Portion of manure managed in each WMS (by state and animal type);
11 • Total Kjeldahl N excretion rate (N_{ex});
12 • Direct N₂O emission factor (EF_{WMS});
13 • Indirect N₂O emission factor for volatilization (EF_{volatilization});
14 • Indirect N₂O emission factor for runoff and leaching (EF_{runoff/leach});
15 • Fraction of N loss from volatilization of NH₃ and NO_x (Frac_{gas}); and
16 • Fraction of N loss from runoff and leaching (Frac_{runoff/leach}).

17 Nitrous oxide emissions were estimated by first determining activity data, including animal population, TAM, WMS
18 usage, and waste characteristics. The activity data sources (except for population, TAM, and WMS, which were
19 described above) are described below:

- 20 • N_{ex} for all cattle except for calves were calculated by head for each state and animal type in the CEFM.
21 N_{ex} rates by animal mass for all other animals were determined using data from USDA's *Agricultural*
22 *Waste Management Field Handbook* (USDA 1996 and 2008; ERG 2010b and 2010c) and data from the
23 American Society of Agricultural Engineers, Standard D384.1 (ASAE 1998) and IPCC (2006). American bison
24 N_{ex} were assumed to be the same as NOF bulls.⁹
25 • All N₂O emission factors (direct and indirect) were taken from IPCC (2006).
26 • Country-specific estimates for the fraction of N loss from volatilization (Frac_{gas}) and runoff and leaching
27 (Frac_{runoff/leach}) were developed. Frac_{gas} values were based on WMS-specific volatilization values as
28 estimated from EPA's *National Emission Inventory - Ammonia Emissions from Animal Agriculture*
29 *Operations* (EPA 2005). Frac_{runoff/leaching} values were based on regional cattle runoff data from EPA's Office
30 of Water (EPA 2002b; see Annex 3.11).

31 To estimate N₂O emissions for cattle (except for calves), the estimated amount of N excreted (kg per animal-year)
32 that is managed in each WMS for each animal type, state, and year were taken from the CEFM. For calves and
33 other animals, the amount of N excreted (kg per year) in manure in each WMS for each animal type, state, and
34 year was calculated. The population (head) for each state and animal was multiplied by TAM (kg animal mass per

⁸ An implied emission factor is defined as emissions divided by the relevant measure of activity; the implied emission factor is equal to emissions per activity data unit. For source/sink categories that are composed of several subcategories, the emissions and activity data are summed up across all subcategories. Hence, the implied emission factors are generally not equivalent to the emission factors used to calculate emission estimates, but are average values that could be used, with caution, in data comparisons (UNFCCC 2017).

⁹ N_{ex} of American bison on grazing lands are accounted for and discussed in the Agricultural Soil Management source category and included under pasture, range and paddock (PRP) emissions. Because American bison are maintained entirely on unmanaged WMS and N₂O emissions from unmanaged WMS are not included in the Manure Management source category, there are no N₂O emissions from American bison included in the Manure Management source category.

1 head) divided by 1,000, the nitrogen excretion rate (N_{ex} , in kg N per 1,000 kg animal mass per day), WMS
2 distribution (percent), and the number of days per year.

3 Direct N_2O emissions were calculated by multiplying the amount of N excreted (kg per year) in each WMS by the
4 N_2O direct emission factor for that WMS (EF_{WMS} , in kg N_2O -N per kg N) and the conversion factor of N_2O -N to N_2O .
5 These emissions were summed over state, animal, and WMS to determine the total direct N_2O emissions (kg of
6 N_2O per year).

7 Indirect N_2O emissions from volatilization (kg N_2O per year) were then calculated by multiplying the amount of N
8 excreted (kg per year) in each WMS by the fraction of N lost through volatilization ($Frac_{gas}$) divided by 100, the
9 emission factor for volatilization ($EF_{volatilization}$, in kg N_2O per kg N), and the conversion factor of N_2O -N to N_2O .

10 Indirect N_2O emissions from runoff and leaching (kg N_2O per year) were then calculated by multiplying the amount
11 of N excreted (kg per year) in each WMS by the fraction of N lost through runoff and leaching ($Frac_{runoff/leach}$)
12 divided by 100, and the emission factor for runoff and leaching ($EF_{runoff/leach}$, in kg N_2O per kg N), and the conversion
13 factor of N_2O -N to N_2O . The indirect N_2O emissions from volatilization and runoff and leaching were summed to
14 determine the total indirect N_2O emissions.

15 Following these steps, direct and indirect N_2O emissions were summed to determine total N_2O emissions (kg N_2O
16 per year) for the years 1990 to 2018.

17 The following approach was used in the calculation of manure management N_2O emissions for 2019:

- 18 • EPA obtained 2019 national-level animal population data. Sheep, poultry, and swine data were
19 downloaded from USDA-NASS Quickstats (USDA 2020). Cattle populations were obtained from the CEFM,
20 see Section 5.1 and Annex 3.10. Data for goats, horses, bison, mules, and asses were extrapolated based
21 on the 2009 through 2018 population values to reflect recent trends in animal populations.
- 22 • The national populations were multiplied by the animal-specific 2018 implied emission factors for N_2O
23 (which combines both direct and indirect N_2O) to calculate national-level 2019 N_2O emissions estimates
24 by animal type. These methods were utilized in order to maintain time-series consistency as referenced in
25 Volume 1, Chapter 5 of the *2006 IPCC Guidelines*.

26 Uncertainty and Time-Series Consistency

27 An analysis (ERG 2003a) was conducted for the manure management emission estimates presented in the 1990
28 through 2001 Inventory (i.e., 2003 submission to the UNFCCC) to determine the uncertainty associated with
29 estimating CH_4 and N_2O emissions from livestock manure management. The quantitative uncertainty analysis for
30 this source category was performed in 2002 through the IPCC-recommended Approach 2 uncertainty estimation
31 methodology, the Monte Carlo Stochastic Simulation technique. The uncertainty analysis was developed based on
32 the methods used to estimate CH_4 and N_2O emissions from manure management systems. A normal probability
33 distribution was assumed for each source data category. The series of equations used were condensed into a single
34 equation for each animal type and state. The equations for each animal group contained four to five variables
35 around which the uncertainty analysis was performed for each state. While there are plans to update the
36 uncertainty to reflect recent manure management updates and forthcoming changes (see Planned Improvements,
37 below), at this time the uncertainty estimates were directly applied to the 2019 emission estimates.

38 The results of the Approach 2 quantitative uncertainty analysis are summarized in Table 5-9. Manure management
39 CH_4 emissions in 2019 were estimated to be between 51.1 and 74.8 MMT CO_2 Eq. at a 95 percent confidence level,
40 which indicates a range of 18 percent below to 20 percent above the actual 2019 emission estimate of 62.4 MMT
41 CO_2 Eq. At the 95 percent confidence level, N_2O emissions were estimated to be between 16.5 and 24.3 MMT CO_2
42 Eq. (or approximately 16 percent below and 24 percent above the actual 2019 emission estimate of 19.6 MMT CO_2
43 Eq.).

44

Table 5-9: Approach 2 Quantitative Uncertainty Estimates for CH₄ and N₂O (Direct and Indirect) Emissions from Manure Management (MMT CO₂ Eq. and Percent)

Source	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
			(MMT CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Manure Management	CH ₄	62.4	51.1	74.8	-18%	+20%
Manure Management	N ₂ O	19.6	16.5	24.3	-16%	+24%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

Methodological approaches, changes to historic data, and other parameters were applied to the entire time series to ensure consistency in emissions estimates from 1990 through 2019. Details on the emission trends and methodologies through time are described in more detail in the Introduction and Methodology sections.

QA/QC and Verification

General (Tier 1) and category-specific (Tier 2) QA/QC activities were conducted consistent with the U.S. Inventory QA/QC plan outlined in Annex 8. Tier 2 activities focused on comparing estimates for the previous and current Inventories for N₂O emissions from managed systems and CH₄ emissions from livestock manure. All errors identified were corrected. Order of magnitude checks were also conducted, and corrections made where needed. In addition, manure N data were checked by comparing state-level data with bottom-up estimates derived at the county level and summed to the state level. Similarly, a comparison was made by animal and WMS type for the full time series, between national level estimates for N excreted and the sum of county estimates for the full time series.

Time-series data, including population, are validated by experts to ensure they are representative of the best available U.S.-specific data. The U.S.-specific values for TAM, Nex, VS, B₀, and MCF were also compared to the IPCC default values and validated by experts. Although significant differences exist in some instances, these differences are due to the use of U.S.-specific data and the differences in U.S. agriculture as compared to other countries. The U.S. manure management emission estimates use the most reliable country-specific data, which are more representative of U.S. animals and systems than the IPCC (2006) default values.

For additional verification of the 1990 to 2018 estimates, the implied CH₄ emission factors for manure management (kg of CH₄ per head per year) were compared against the default IPCC (2006) values.¹⁰ Table 5-10 presents the implied emission factors of kg of CH₄ per head per year used for the manure management emission estimates as well as the IPCC (2006) default emission factors. The U.S. implied emission factors fall within the range of the IPCC (2006) default values, except in the case of sheep, goats, and some years for horses and dairy cattle. The U.S. implied emission factors are greater than the IPCC (2006) default value for those animals due to the use of U.S.-specific data for typical animal mass and VS excretion. There is an increase in implied emission factors for dairy cattle and swine across the time series. This increase reflects the dairy cattle and swine industry trend towards larger farm sizes; large farms are more likely to manage manure as a liquid and therefore produce more CH₄ emissions.

¹⁰ CH₄ implied emission factors were not calculated for 2019 due to the simplified emissions estimation approach used to estimate emissions for that year.

1 **Table 5-10: IPCC (2006) Implied Emission Factor Default Values Compared with Calculated**
 2 **Values for CH₄ from Manure Management (kg/head/year)**

Animal Type	IPCC Default CH ₄ Emission Factors (kg/head/year) ^a	Implied CH ₄ Emission Factors (kg/head/year)						
		1990	2005	2015	2016	2017	2018	2019
Dairy Cattle	48-112	30.2	54.5	65.6	66.8	67.2	67.9	67.9
Beef Cattle	1-2	1.5	1.6	1.7	1.7	1.7	1.6	1.6
Swine	10-45	11.5	13.3	11.8	12.1	11.7	12.0	12.0
Sheep	0.19-0.37	0.6	0.6	0.5	0.5	0.5	0.5	0.5
Goats	0.13-0.26	0.4	0.3	0.3	0.3	0.3	0.3	0.3
Poultry	0.02-1.4	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Horses	1.56-3.13	4.3	3.1	2.6	2.6	2.6	2.6	2.6
American Bison	NA	1.8	2.0	2.1	2.1	2.1	2.1	2.1
Mules and Asses	0.76-1.14	0.9	1.0	1.0	1.0	1.0	1.0	1.0

Note: CH₄ implied emission factors were not calculated for 2019 due to the simplified emissions estimation approach used to estimate emissions for that year. 2018 values were used for 2019.

NA (Not Applicable)

^a Ranges reflect 2006 IPCC Guidelines (Volume 4, Table 10.14) default emission factors for North America across different climate zones.

3 In addition, default IPCC (2006) emission factors for N₂O were compared to the U.S. Inventory implied N₂O
 4 emission factors. Default N₂O emission factors from the 2006 IPCC Guidelines were used to estimate N₂O emission
 5 from each WMS in conjunction with U.S.-specific Nex values. The implied emission factors differed from the U.S.
 6 Inventory values due to the use of U.S.-specific Nex values and differences in populations present in each WMS
 7 throughout the time series.

8 Recalculations Discussion

9 No recalculations were performed for the 1990 to 2018 estimates. The 2019 estimates were developed using a
 10 simplified approach, as discussed in the Methodology section.

11 Planned Improvements

12 Regular annual data reviews and updates are necessary to maintain an emissions inventory that reflects the
 13 current base of knowledge. EPA conducts the following list of regular annual assessments of data availability when
 14 updating the estimates to extend time series each year. EPA is actively pursuing the following updates but notes
 15 that implementation may be based on available resources and data availability:

- 16 • Continuing to investigate new sources of WMS data. EPA is working with the USDA Natural Resources
 17 Conservation Service to collect data for potential improvements to the Inventory. EPA expects the next
 18 WMS systems to be updated for the next (i.e., 1990 to 2020) Inventory submission include poultry and
 19 beef cattle.
- 20 • Updating the B₀ data used in the Inventory, as data become available.
- 21 • Revising the methodology for population distribution to states where USDA population data are withheld
 22 due to disclosure concerns. These updates will be made in collaboration with the EPA National Emissions
 23 Inventory staff to improve consistency across U.S. inventories. EPA plans to incorporate these updates
 24 into the next (i.e., 1990 to 2020) Inventory submission.

25 IPCC's 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories provides updated
 26 emission factors that EPA plans to review and implement for manure management (IPCC 2019). EPA maintains
 27 from previous reports that many of the improvements identified below are major updates and may take multiple
 28 years to fully implement. Potential improvements (long-term improvements) for future Inventory years include:

- 1 • Revising the anaerobic digestion estimates to estimate CH₄ emissions *reductions* due to the use of
2 anaerobic digesters (the Inventory currently estimates only emissions from anaerobic digestion systems).
- 3 • Investigating improved emissions estimate methodologies for swine pit systems with less than one month
4 of storage (the recently updated swine WMS data included this WMS category).
- 5 • Comparing CH₄ and N₂O emission estimates with estimates from other models and more recent studies
6 and compare the results to the Inventory.
- 7 • Comparing manure management emission estimates with on-farm measurement data to identify
8 opportunities for improved estimates.
- 9 • Comparing VS and Nex data to literature data to identify opportunities for improved estimates.
- 10 • Improving collaboration with the Enteric Fermentation source category estimates. For future inventories,
11 it may be beneficial to have the CEFM and Manure Management calculations in the same model, as they
12 rely on much of the same activity data and they depend on each other’s outputs to properly calculate
13 emissions.
- 14 • Revising the uncertainty analysis to address changes that have been implemented to the CH₄ and N₂O
15 estimates.

16 5.3 Rice Cultivation (CRF Source Category 3C)

17 Most of the world’s rice is grown on flooded fields (Baicich 2013) that create anaerobic conditions leading to CH₄
18 production through a process known as methanogenesis. Approximately 60 to 90 percent of the CH₄ produced by
19 methanogenic bacteria in flooded rice fields is oxidized in the soil and converted to CO₂ by methanotrophic
20 bacteria. The remainder is emitted to the atmosphere (Holzapfel-Pschorn et al. 1985; Sass et al. 1990) or
21 transported as dissolved CH₄ into groundwater and waterways (Neue et al. 1997). Methane is transported to the
22 atmosphere primarily through the rice plants, but some CH₄ also escapes via ebullition (i.e., bubbling through the
23 water) and to a much lesser extent by diffusion through the water (van Bodegom et al. 2001).

24 Water management is arguably the most important factor affecting CH₄ emissions in rice cultivation, and improved
25 water management has the largest potential to mitigate emissions (Yan et al. 2009). Upland rice fields are not
26 flooded, and therefore do not produce CH₄, but large amounts of CH₄ can be emitted in continuously irrigated
27 fields, which is the most common practice in the United States (USDA 2012). Single or multiple aeration events
28 with drainage of a field during the growing season can significantly reduce these emissions (Wassmann et al.
29 2000a), but drainage may also increase N₂O emissions. Deepwater rice fields (i.e., fields with flooding depths
30 greater than one meter, such as natural wetlands) tend to have fewer living stems reaching the soil, thus reducing
31 the amount of CH₄ transport to the atmosphere through the plant compared to shallow-flooded systems (Sass
32 2001).

33 Other management practices also influence CH₄ emissions from flooded rice fields including rice residue straw
34 management and application of organic amendments, in addition to cultivar selection due to differences in the
35 amount of root exudates¹¹ among rice varieties (Neue et al. 1997). These practices influence the amount of
36 organic matter available for methanogenesis, and some practices, such as mulching rice straw or composting
37 organic amendments, can reduce the amount of labile carbon and limit CH₄ emissions (Wassmann et al. 2000b).
38 Fertilization practices also influence CH₄ emissions, particularly the use of fertilizers with sulfate (Wassmann et al.
39 2000b; Linquist et al. 2012), which can reduce CH₄ emissions. Other environmental variables also impact the
40 methanogenesis process such as soil temperature and soil type. Soil temperature regulates the activity of

¹¹ The roots of rice plants add organic material to the soil through a process called “root exudation.” Root exudation is thought to enhance decomposition of the soil organic matter and release nutrients that the plant can absorb and use to stimulate more production. The amount of root exudate produced by a rice plant over a growing season varies among rice varieties.

1 methanogenic bacteria, which in turn affects the rate of CH₄ production. Soil texture influences decomposition of
 2 soil organic matter, but is also thought to have an impact on oxidation of CH₄ in the soil (Sass et al. 1994).

3 Rice is currently cultivated in thirteen states, including Arkansas, California, Florida, Illinois, Kentucky, Louisiana,
 4 Minnesota, Mississippi, Missouri, New York, South Carolina, Tennessee and Texas. Soil types, rice varieties, and
 5 cultivation practices vary across the United States, but most farmers apply fertilizers and do not harvest crop
 6 residues. In addition, a second, ratoon rice crop is grown in the Southeastern region of the country. Ratoon crops
 7 are produced from regrowth of the stubble remaining after the harvest of the first rice crop. Methane emissions
 8 from ratoon crops are higher than those from the primary crops due to the increased amount of labile organic
 9 matter available for anaerobic decomposition in the form of relatively fresh crop residue straw. Emissions tend to
 10 be higher in rice fields if the residues have been in the field for less than 30 days before planting the next rice crop
 11 (Lindau and Bollich 1993; IPCC 2006; Wang et al. 2013).

12 A combination of Tier 1 and 3 methods are used to estimate CH₄ emissions from rice cultivation across most of the
 13 time series, while a surrogate data method has been applied to estimate national emissions for 2016 to 2019 in
 14 this Inventory due to lack of data in the later years of the time series. National emission estimates based on
 15 surrogate data will be recalculated in a future Inventory with the Tier 1 and 3 methods as data becomes available.

16 Overall, rice cultivation is a minor source of CH₄ emissions in the United States relative to other source categories
 17 (see Table 5-11, Table 5-12, and Figure 5-3). Most emissions occur in Arkansas, California, Louisiana, Mississippi,
 18 Missouri and Texas. In 2019, CH₄ emissions from rice cultivation were 15.1 MMT CO₂ Eq. (602 kt). Annual emissions
 19 fluctuate between 1990 and 2019, which is largely due to differences in the amount of rice harvested areas over
 20 time, which has been decreasing over the past two decades. Consequently, emissions in 2019 are six percent lower
 21 than emissions in 1990.

22 **Table 5-11: CH₄ Emissions from Rice Cultivation (MMT CO₂ Eq.)**

State	1990	2005	2015	2016	2017	2018	2019
Arkansas	5.4	7.9	6.4	NE	NE	NE	NE
California	3.3	3.4	4.1	NE	NE	NE	NE
Florida	+	+	+	NE	NE	NE	NE
Illinois	+	+	+	NE	NE	NE	NE
Kentucky	+	+	+	NE	NE	NE	NE
Louisiana	2.6	2.8	2.6	NE	NE	NE	NE
Minnesota	+	0.1	+	NE	NE	NE	NE
Mississippi	1.1	1.4	1.0	NE	NE	NE	NE
Missouri	0.6	1.1	0.7	NE	NE	NE	NE
New York	+	+	+	NE	NE	NE	NE
South Carolina	+	+	+	NE	NE	NE	NE
Tennessee	+	+	+	NE	NE	NE	NE
Texas	3.0	1.3	1.4	NE	NE	NE	NE
Total	16.0	18.0	16.2	15.8	14.9	15.6	15.1

Note: Totals may not sum due to independent rounding.

+ Does not exceed 0.05 MMT CO₂ Eq.

NE (Not Estimated). State-level emissions are not estimated for 2016 through 2019 in this Inventory because data are unavailable. A surrogate data method is used to estimate emissions for these years and are produced only at the national scale.

23 **Table 5-12: CH₄ Emissions from Rice Cultivation (kt)**

State	1990	2005	2015	2016	2017	2018	2019
Arkansas	216	315	256	NE	NE	NE	NE
California	131	135	166	NE	NE	NE	NE
Florida	+	1	+	NE	NE	NE	NE
Illinois	+	+	+	NE	NE	NE	NE
Kentucky	+	+	+	NE	NE	NE	NE
Louisiana	103	113	103	NE	NE	NE	NE

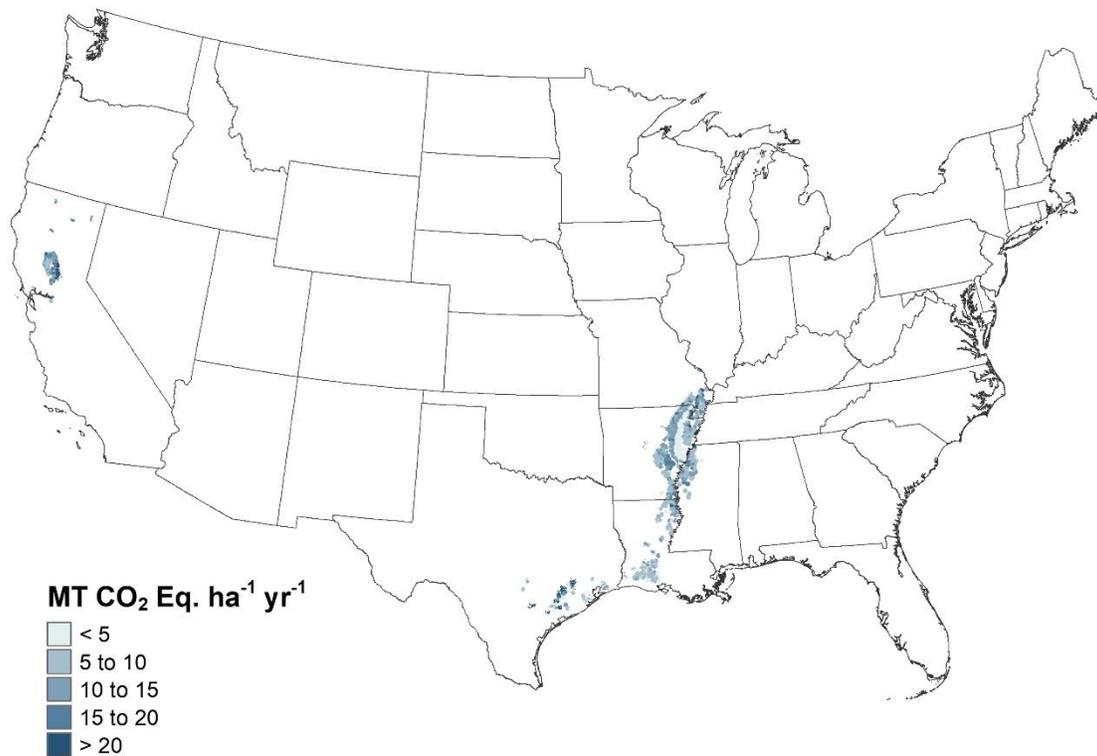
Minnesota	1	2	+	NE	NE	NE	NE
Mississippi	45	55	40	NE	NE	NE	NE
Missouri	22	45	26	NE	NE	NE	NE
New York	+	+	+	NE	NE	NE	NE
South Carolina	+	+	+	NE	NE	NE	NE
Tennessee	+	+	+	NE	NE	NE	NE
Texas	122	54	57	NE	NE	NE	NE
Total	640	720	648	631	596	623	602

Note: Totals may not sum due to independent rounding.

+ Does not exceed 0.5 kt.

NE (Not Estimated). State-level emissions are not estimated for 2016 through 2019 in this Inventory because data are unavailable. A surrogate data method is used to estimate emissions for these years and are produced only at the national scale.

1 **Figure 5-3: Annual CH₄ Emissions from Rice Cultivation, 2015**



2
3 Note: Only national-scale emissions are estimated for 2016 through 2019 in this Inventory using the surrogate data method
4 described in the Methodology section; therefore, the fine-scale emission patterns in this map are based on the estimates for
5 2015.

6 Methodology

7 The methodology used to estimate CH₄ emissions from rice cultivation is based on a combination of IPCC Tier 1 and
8 3 approaches. The Tier 3 method utilizes the DayCent process-based model to estimate CH₄ emissions from rice
9 cultivation (Cheng et al. 2013), and has been tested in the United States (see Annex 3.12) and Asia (Cheng et al.
10 2013, 2014). The model simulates hydrological conditions and thermal regimes, organic matter decomposition,
11 root exudation, rice plant growth and its influence on oxidation of CH₄, as well as CH₄ transport through the plant
12 and via ebullition (Cheng et al. 2013). The method captures the influence of organic amendments and rice straw
13 management on methanogenesis in the flooded soils, and ratooning of rice crops with a second harvest during the

1 growing season. In addition to CH₄ emissions, DayCent simulates soil C stock changes and N₂O emissions (Parton et
 2 al. 1987 and 1998; Del Grosso et al. 2010), and allows for a seamless set of simulations for crop rotations that
 3 include both rice and non-rice crops.

4 The Tier 1 method is applied to estimate CH₄ emissions from rice when grown in rotation with crops that are not
 5 simulated by DayCent, such as vegetable crops. The Tier 1 method is also used for areas converted between
 6 agriculture (i.e., cropland and grassland) and other land uses, such as forest land, wetland, and settlements. In
 7 addition, the Tier 1 method is used to estimate CH₄ emissions from organic soils (i.e., Histosols) and from areas
 8 with very gravelly, cobbly, or shaley soils (greater than 35 percent by volume). The Tier 3 method using DayCent
 9 has not been fully tested for estimating emissions associated with these crops and rotations, land uses, as well as
 10 organic soils or cobbly, gravelly, and shaley mineral soils.

11 The Tier 1 method for estimating CH₄ emissions from rice production utilizes a default base emission rate and
 12 scaling factors (IPCC 2006). The base emission rate represents emissions for continuously flooded fields with no
 13 organic amendments. Scaling factors are used to adjust the base emission rate for water management and organic
 14 amendments that differ from continuous flooding with no organic amendments. The method accounts for pre-
 15 season and growing season flooding; types and amounts of organic amendments; and the number of rice
 16 production seasons within a single year (i.e., single cropping, ratooning, etc.). The Tier 1 analysis is implemented in
 17 the Agriculture and Land Use National Greenhouse Gas Inventory (ALU) software (Ogle et al. 2016).¹²

18 Rice cultivation areas are based on cropping and land use histories recorded in the USDA National Resources
 19 Inventory (NRI) survey (USDA-NRCS 2018). The NRI is a statistically-based sample of all non-federal land, and
 20 includes 489,178 survey locations in agricultural land for the conterminous United States and Hawaii of which
 21 1,960 include one or more years of rice cultivation. The Tier 3 method is used to estimate CH₄ emissions from
 22 1,655 of the NRI survey locations, and the remaining 305 survey locations are estimated with the Tier 1 method.
 23 Each NRI survey location is associated with an “expansion factor” that allows scaling of CH₄ emission to the entire
 24 land base with rice cultivation (i.e., each expansion factor represents the amount of area with the same land-
 25 use/management history as the survey location). Land-use and some management information in the NRI (e.g.,
 26 crop type, soil attributes, and irrigation) were collected on a 5-year cycle beginning in 1982, along with cropping
 27 rotation data in 4 out of 5 years for each 5-year time period (i.e., 1979 to 1982, 1984 to 1987, 1989 to 1992, and
 28 1994 to 1997). The NRI program began collecting annual data in 1998, with data currently available through 2015
 29 (USDA-NRCS 2018). The current Inventory only uses NRI data through 2015 because newer data are not available,
 30 but will be incorporated when additional years of data are released by USDA-NRCS. The harvested rice areas in
 31 each state are presented in Table 5-13.

32 **Table 5-13: Rice Area Harvested (1,000 Hectares)**

State/Crop	1990	2005	2015	2016	2017	2018	2019
Arkansas	600	784	679	NE	NE	NE	NE
California	249	236	280	NE	NE	NE	NE
Florida	0	4	0	NE	NE	NE	NE
Illinois	0	0	0	NE	NE	NE	NE
Kentucky	0	0	0	NE	NE	NE	NE
Louisiana	381	402	368	NE	NE	NE	NE
Minnesota	4	9	1	NE	NE	NE	NE
Mississippi	123	138	98	NE	NE	NE	NE
Missouri	48	94	62	NE	NE	NE	NE
New York	1	0	0	NE	NE	NE	NE
South Carolina	0	0	0	NE	NE	NE	NE
Tennessee	0	1	0	NE	NE	NE	NE
Texas	302	118	131	NE	NE	NE	NE
Total	1,707	1,788	1,619	NE	NE	NE	NE

¹² See <<http://www.nrel.colostate.edu/projects/ALUsoftware/>>.

Note: Totals may not sum due to independent rounding.

NE (Not Estimated). State-level area data are not available for 2016 through 2019 but will be added in a future Inventory with release of new NRI survey data.

1 The Southeastern states have sufficient growing periods for a ratoon crop in some years (Table 5-14). For example,
2 the growing season length is occasionally sufficient for ratoon crops to be grown on about 1 percent of the rice
3 fields in Arkansas. No data are available about ratoon crops in Missouri or Mississippi, and the average amount of
4 ratooning in Arkansas was assigned to these states. Ratoon cropping occurs much more frequently in Louisiana
5 (LSU 2015 for years 2000 through 2013, 2015) and Texas (TAMU 2015 for years 1993 through 2015), averaging 32
6 percent and 45 percent of rice acres planted, respectively. Florida also has a large fraction of area with a ratoon
7 crop (49 percent). Ratoon rice crops are not grown in California.

8 **Table 5-14: Average Ratooned Area as Percent of Primary Growth Area (Percent)**

State	1990-2015
Arkansas ^a	1%
California	0%
Florida ^b	49%
Louisiana ^c	32%
Mississippi ^a	1%
Missouri ^a	1%
Texas ^d	45%

9 ^aArkansas: 1990–2000 (Slaton 1999 through 2001); 2001–2011 (Wilson 2002 through 2007, 2009 through 2012); 2012–2013
10 (Hardke 2013, 2014). Estimates of ratooning for Missouri and Mississippi are based on the data from Arkansas.

11 ^bFlorida - Ratoon: 1990–2000 (Schueneman 1997, 1999 through 2001); 2001 (Deren 2002); 2002–2003 (Kirstein 2003
12 through 2004, 2006); 2004 (Cantens 2004 through 2005); 2005–2013 (Gonzalez 2007 through 2014).

13 ^cLouisiana: 1990–2013 (Linscombe 1999, 2001 through 2014).

14 ^dTexas: 1990–2002 (Klosterboer 1997, 1999 through 2003); 2003–2004 (Stansel 2004 through 2005); 2005 (Texas Agricultural
15 Experiment Station 2006); 2006–2013 (Texas Agricultural Experiment Station 2007 through 2014).

16 While rice crop production in the United States includes a minor amount of land with mid-season drainage or
17 alternate wet-dry periods, the majority of rice growers use continuously flooded water management systems
18 (Hardke 2015; UCCE 2015; Hollier 1999; Way et al. 2014). Therefore, continuous flooding was assumed in the
19 DayCent simulations and the Tier 1 method. Variation in flooding can be incorporated in future Inventories if water
20 management data are collected.

21 Winter flooding is another key practice associated with water management in rice fields, and the impact of winter
22 flooding on CH₄ emissions is addressed in the Tier 3 and Tier 1 analyses. Flooding is used to prepare fields for the
23 next growing season, and to create waterfowl habitat (Young 2013; Miller et al. 2010; Fleskes et al. 2005).
24 Fitzgerald et al. (2000) suggests that as much as 50 percent of the annual emissions may occur during winter
25 flooding. Winter flooding is a common practice with an average of 34 percent of fields managed with winter
26 flooding in California (Miller et al. 2010; Fleskes et al. 2005), and approximately 21 percent of the fields managed
27 with winter flooding in Arkansas (Wilson and Branson 2005 and 2006; Wilson and Runsick 2007 and 2008; Wilson
28 et al. 2009 and 2010; Hardke and Wilson 2013 and 2014; Hardke 2015). No data are available on winter flooding
29 for Texas, Louisiana, Florida, Missouri, or Mississippi. For these states, the average amount of flooding is assumed
30 to be similar to Arkansas. In addition, the amount of flooding is assumed to be relatively constant over the
31 Inventory time series.

32 A surrogate data method is used to estimate emissions from 2016 to 2019 associated with the rice CH₄ emissions
33 for Tier 1 and 3 methods. Specifically, a linear regression model with autoregressive moving-average (ARMA)
34 errors was used to estimate the relationship between the surrogate data and emissions data from 1990 through

1 2015, which were derived using the Tier 1 and 3 methods (Brockwell and Davis 2016). Surrogate data are based on
2 rice commodity statistics from USDA-NASS.¹³ See Box 5-2 for more information about the surrogate data method.

3 **Box 5-2: Surrogate Data Method**

An approach to extend the time series is needed to estimate emissions from Rice Cultivation because there are gaps in activity data at the end of the time series. This is mainly due to the fact that the National Resources Inventory (NRI) does not release data every year, and the NRI is a key data source for estimating greenhouse gas emissions.

A surrogate data method has been selected to impute missing emissions at the end of the time series. A linear regression model with autoregressive moving-average (ARMA) errors (Brockwell and Davis 2016) is used to estimate the relationship between the surrogate data and the observed 1990 to 2015 emissions data that has been compiled using the inventory methods described in this section. The model to extend the time series is given by

$$Y=X\beta+ \epsilon,$$

where Y is the response variable (e.g., CH₄ emissions), Xβ is the surrogate data that is used to predict the missing emissions data, and ε is the remaining unexplained error. Models with a variety of surrogate data were tested, including commodity statistics, weather data, or other relevant information. Parameters are estimated from the observed data for 1990 to 2015 using standard statistical techniques, and these estimates are used to predict the missing emissions data for 2016 to 2019.

A critical issue in using splicing methods is to adequately account for the additional uncertainty introduced by predicting emissions with related information without compiling the full inventory. For example, predicting CH₄ emissions will increase the total variation in the emission estimates for these specific years, compared to those years in which the full inventory is compiled. This added uncertainty is quantified within the model framework using a Monte Carlo approach. The approach requires estimating parameters for results in each Monte Carlo simulation for the full inventory (i.e., the surrogate data model is refit with the emissions estimated in each Monte Carlo iteration from the full inventory analysis with data from 1990 to 2015).

4

5 **Uncertainty and Time-Series Consistency**

6 Sources of uncertainty in the Tier 3 method include management practices, uncertainties in model structure (i.e.,
7 algorithms and parameterization), and variance associated with the NRI sample. Sources of uncertainty in the IPCC
8 (2006) Tier 1 method include the emission factors, management practices, and variance associated with the NRI
9 sample. A Monte Carlo analysis was used to propagate uncertainties in the Tier 1 and 3 methods. For 2016 to 2019,
10 there is additional uncertainty propagated through the Monte Carlo analysis associated with the surrogate data
11 method (See Box 5-2 for information about propagating uncertainty with the surrogate data method). The
12 uncertainties from the Tier 1 and 3 approaches are combined to produce the final CH₄ emissions estimate using
13 simple error propagation (IPCC 2006). Additional details on the uncertainty methods are provided in Annex 3.12.
14 Rice cultivation CH₄ emissions in 2019 were estimated to be between 3.8 and 37.5 MMT CO₂ Eq. at a 95 percent
15 confidence level, which indicates a range of 75 percent below to 149 percent above the 2019 emission estimate of
16 15.1 MMT CO₂ Eq. (see Table 5-15).

¹³ See <<https://quickstats.nass.usda.gov/>>.

Table 5-15: Approach 2 Quantitative Uncertainty Estimates for CH₄ Emissions from Rice Cultivation (MMT CO₂ Eq. and Percent)

Source	Inventory Method	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
				(MMT CO ₂ Eq.)		(%)	
				Lower Bound	Upper Bound	Lower Bound	Upper Bound
Rice Cultivation	Tier 3	CH ₄	12.5	1.4	23.7	-89%	+89%
Rice Cultivation	Tier 1	CH ₄	2.5	1.3	3.7	-48%	+48%
Rice Cultivation	Total	CH₄	15.1	3.8	37.5	-75%	+149%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

Methodological approaches, changes to historic data, and other parameters were applied to the entire time series to ensure consistency in emissions estimates from 1990 through 2019. Details on the emission trends and methodologies through time are described in more detail in the Introduction and Methodology sections.

QA/QC and Verification

General (Tier 1) and category-specific (Tier 2) QA/QC activities were conducted consistent with the U.S. Inventory QA/QC plan outlined in Annex 8. Quality control measures include checking input data, model scripts, and results to ensure data are properly handled throughout the inventory process. Inventory reporting forms and text are reviewed and revised as needed to correct transcription errors. One error was found in the Tier-3 linear regression with ARMA surrogate data method and corrected. For each Monte Carlo iteration, total CH₄ emissions data were transformed using a constant scaler to meet the model requirement, however during the back-transformation only one constant was used for all Monte Carlo iteration. This results in a bias model prediction and lower uncertainty in the previous year's inventory. The estimates were corrected by updating the code and emissions were re-estimated for the years 2016 to 2019.

Model results are compared to field measurements to verify if results adequately represent CH₄ emissions. The comparisons included over 17 long-term experiments, representing about 238 combinations of management treatments across all the sites. A statistical relationship was developed to assess uncertainties in the model structure, adjusting the estimates for model bias and assessing precision in the resulting estimates (methods are described in Ogle et al. 2007). See Annex 3.12 for more information.

Recalculations Discussion

Emissions data from 2016 to 2018 were corrected based on an error in the data splicing method (see QA/QC and Verification section). This change resulted in an average increase in CH₄ emissions of 2.2 MMT CO₂ Eq., or 2.3 percent, from 2016 to 2018 relative to the previous Inventory.

Planned Improvements

A key planned improvement for rice cultivation is to fill several gaps in the management activity including compiling new data on water management, organic amendments and ratooning practices in rice cultivation systems. This improvement is expected to be completed for the next Inventory, but may be prioritized considering overall improvements to make best use of available resources.

5.4 Agricultural Soil Management (CRF Source Category 3D)

Nitrous oxide is naturally produced in soils through the microbial processes of nitrification and denitrification that is driven by the availability of mineral nitrogen (N) (Firestone and Davidson 1989).¹⁴ Mineral N is made available in soils through decomposition of soil organic matter and plant litter, as well as asymbiotic fixation of N from the atmosphere.¹⁵ Several agricultural activities increase mineral N availability in soils that lead to direct N₂O emissions at the site of a management activity (see Figure 5-4) (Mosier et al. 1998). These activities include synthetic N fertilization; application of managed livestock manure; application of other organic materials such as biosolids (i.e., treated sewage sludge); deposition of manure on soils by domesticated animals in pastures, range, and paddocks (PRP) (i.e., unmanaged manure); retention of crop residues (N-fixing legumes and non-legume crops and forages); and drainage of organic soils¹⁶ (i.e., Histosols) (IPCC 2006). Additionally, agricultural soil management activities, including irrigation, drainage, tillage practices, cover crops, and fallowing of land, can influence N mineralization from soil organic matter and levels of asymbiotic N fixation. Indirect emissions of N₂O occur when N is transported from a site and is subsequently converted to N₂O; there are two pathways for indirect emissions: (1) volatilization and subsequent atmospheric deposition of applied/mineralized N, and (2) surface runoff and leaching of applied/mineralized N into groundwater and surface water.¹⁷ Direct and indirect emissions from agricultural lands are included in this section (i.e., cropland and grassland as defined in Section 6.1 Representation of the U.S. Land Base). Nitrous oxide emissions from Forest Land and Settlements soils are found in Sections 6.2 and 6.10, respectively.

¹⁴ Nitrification and denitrification are driven by the activity of microorganisms in soils. Nitrification is the aerobic microbial oxidation of ammonium (NH₄⁺) to nitrate (NO₃⁻), and denitrification is the anaerobic microbial reduction of nitrate to N₂. Nitrous oxide is a gaseous intermediate product in the reaction sequence of nitrification and denitrification.

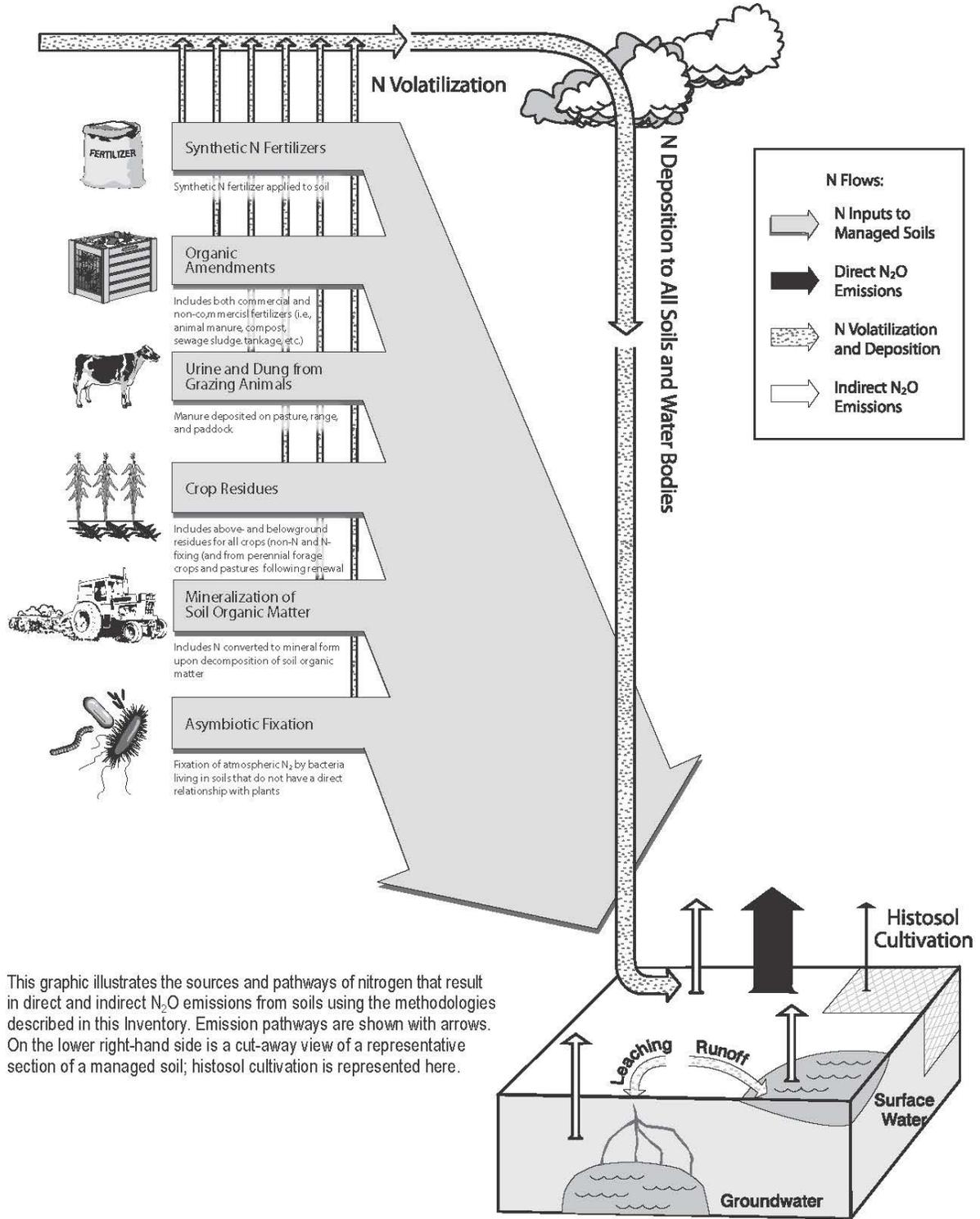
¹⁵ Asymbiotic N fixation is the fixation of atmospheric N₂ by bacteria living in soils that do not have a direct relationship with plants.

¹⁶ Drainage of organic soils in former wetlands enhances mineralization of N-rich organic matter, thereby increasing N₂O emissions from these soils.

¹⁷ These processes entail volatilization of applied or mineralized N as NH₃ and NO_x, transformation of these gases in the atmosphere (or upon deposition), and deposition of the N primarily in the form of particulate NH₄⁺, nitric acid (HNO₃), and NO_x. In addition, hydrological processes lead to leaching and runoff of NO₃⁻ that is converted to N₂O in aquatic systems, e.g., wetlands, rivers, streams and lakes. Note: N₂O emissions are not estimated for aquatic systems associated with N inputs from terrestrial systems in order to avoid double-counting.

1 **Figure 5-4: Sources and Pathways of N that Result in N₂O Emissions from Agricultural Soil Management**
 2

Sources and Pathways of N that Result in N₂O Emissions from Agricultural Soil Management



This graphic illustrates the sources and pathways of nitrogen that result in direct and indirect N₂O emissions from soils using the methodologies described in this Inventory. Emission pathways are shown with arrows. On the lower right-hand side is a cut-away view of a representative section of a managed soil; histosol cultivation is represented here.

1 Agricultural soils produce the majority of N₂O emissions in the United States. Estimated emissions in 2019 are
 2 344.6 MMT CO₂ Eq. (1,156 kt) (see Table 5-16 and Table 5-17). Annual N₂O emissions from agricultural soils are 9
 3 percent greater in the 2019 compared to 1990, but emissions fluctuated between 1990 and 2019 due to inter-
 4 annual variability largely associated with weather patterns, synthetic fertilizer use, and crop production. From
 5 1990 to 2019, cropland accounted for 68 percent of total direct emissions on average, while grassland accounted
 6 for 32 percent. On average, 79 percent of indirect emissions are from croplands and 21 percent from grasslands.
 7 Estimated direct and indirect N₂O emissions by sub-source category are shown in Table 5-18 and Table 5-19.

8 **Table 5-16: N₂O Emissions from Agricultural Soils (MMT CO₂ Eq.)**

Activity	1990	2005	2015	2016	2017	2018	2019
Direct	272.5	272.6	295.0	281.4	280.3	285.9	290.4
Cropland	185.9	183.7	199.5	190.8	190.4	195.1	196.4
Grassland	86.6	88.8	95.4	90.6	89.9	90.9	94.0
Indirect	43.4	40.8	53.5	48.7	47.3	52.3	54.2
Cropland	34.2	31.6	42.7	38.8	37.4	42.3	43.8
Grassland	9.2	9.2	10.8	9.9	9.8	10.0	10.4
Total	315.9	313.4	348.5	330.1	327.6	338.2	344.6

Notes: Estimates after 2015 are based on a data splicing method (See Methodology section). Totals may not sum due to independent rounding.

9 **Table 5-17: N₂O Emissions from Agricultural Soils (kt)**

Activity	1990	2005	2015	2016	2017	2018	2019
Direct	914.5	914.7	989.9	944.3	940.6	959.5	974.5
Cropland	623.8	616.6	669.6	640.3	639.0	654.5	659.1
Grassland	290.7	298.1	320.2	304.1	301.6	305.0	315.5
Indirect	145.6	137.0	179.6	163.4	158.6	175.5	181.9
Cropland	114.8	106.1	143.2	130.3	125.5	142.0	147.1
Grassland	30.7	30.9	36.4	33.1	33.0	33.4	34.8
Total	1,060.1	1,051.6	1,169.4	1,107.7	1,099.2	1,135.0	1,156.4

Notes: Estimates after 2015 are based on a data splicing method (See Methodology section). Totals may not sum due to independent rounding.

10 **Table 5-18: Direct N₂O Emissions from Agricultural Soils by Land Use Type and N Input Type (MMT CO₂ Eq.)**
 11

Activity	1990	2005	2015	2016	2017	2018	2019
Cropland	185.8	183.7	199.5	190.8	190.4	195.0	196.4
Mineral Soils	182.1	180.0	196.1	187.4	187.0	191.6	193.0
Synthetic Fertilizer	63.1	64.0	64.8	68.8	68.5	70.1	70.4
Organic Amendment ^a	12.6	13.0	13.4	14.5	14.3	14.3	14.2
Residue N ^b	39.3	39.6	39.0	40.1	40.1	41.2	41.6
Mineralization and Asymbiotic Fixation	67.1	63.3	78.9	64.0	64.1	66.1	66.8
Drained Organic Soils	3.8	3.7	3.4	3.4	3.4	3.4	3.4
Grassland	86.7	88.9	95.5	90.6	89.9	90.9	94.0
Mineral Soils	84.2	86.5	93.0	88.2	87.4	88.4	91.6
Synthetic Fertilizer	+	+	+	+	+	+	+
PRP Manure	14.6	13.4	12.8	12.8	12.8	12.9	13.2
Managed Manure ^c	+	+	+	+	+	+	+
Biosolids (i.e., treated Sewage Sludge)	0.2	0.5	0.6	0.6	0.6	0.6	0.7
Residue N ^d	29.7	30.8	30.4	31.5	31.2	31.6	32.8
Mineralization and Asymbiotic Fixation	39.5	41.7	49.2	43.2	42.8	43.3	44.9

Drained Organic Soils	2.5	2.4	2.5	2.5	2.5	2.5	2.5
Total	272.5	272.6	295.0	281.4	280.3	285.9	290.4

Notes: Estimates after 2015 are based on a data splicing method (See Methodology section). Totals may not sum due to independent rounding.

+ Does not exceed 0.05 MMT CO₂ Eq.

^a Organic amendment inputs include managed manure, daily spread manure, and commercial organic fertilizers (i.e., dried blood, dried manure, tankage, compost, and other).

^b Cropland residue N inputs include N in unharvested legumes as well as crop residue N.

^c Managed manure inputs include managed manure and daily spread manure amendments that are applied to grassland soils.

^d Grassland residue N inputs include N in ungrazed legumes as well as ungrazed grass residue N.

1 **Table 5-19: Indirect N₂O Emissions from Agricultural Soils (MMT CO₂ Eq.)**

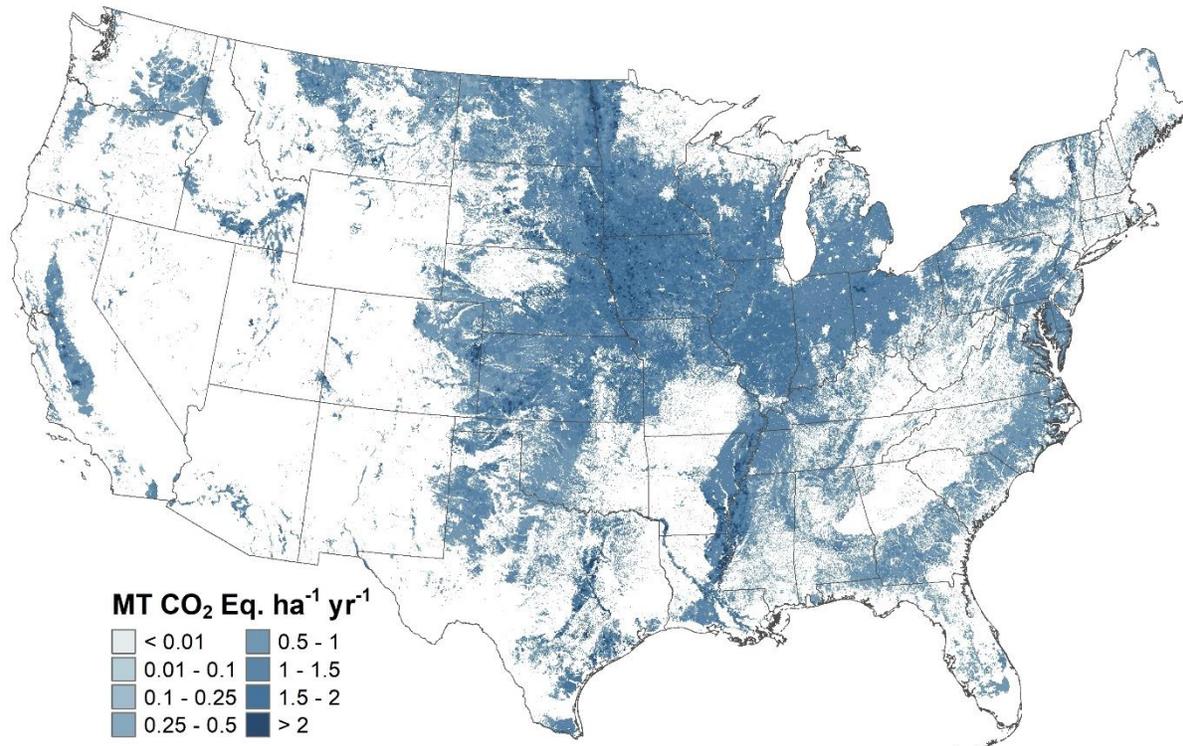
Activity	1990	2005	2015	2016	2017	2018	2019
Cropland	34.2	31.6	42.7	38.8	37.4	42.3	43.8
Volatilization & Atm.							
Deposition	6.5	7.3	8.5	8.1	7.9	8.0	7.9
Surface Leaching & Run-Off	27.7	24.4	34.2	30.7	29.5	34.4	35.9
Grassland	9.2	9.2	10.8	9.9	9.8	10.0	10.4
Volatilization & Atm.							
Deposition	3.6	3.6	3.7	3.5	3.5	3.5	3.6
Surface Leaching & Run-Off	5.6	5.6	7.2	6.4	6.3	6.4	6.8
Total	43.4	40.8	53.5	48.7	47.3	52.3	54.2

Notes: Estimates after 2015 are based on a data splicing method (See Methodology section). Totals may not sum due to independent rounding.

2 Figure 5-5 and Figure 5-6 show regional patterns for direct N₂O emissions. Figure 5-7 and Figure 5-8 show indirect
3 N₂O emissions from volatilization, and Figure 5-9 and Figure 5-10 show the indirect N₂O emissions from leaching
4 and runoff in croplands and grasslands, respectively.

5

1 **Figure 5-5: Croplands, 2015 Annual Direct N₂O Emissions Estimated Using the Tier 3**
2 **DayCent Model**

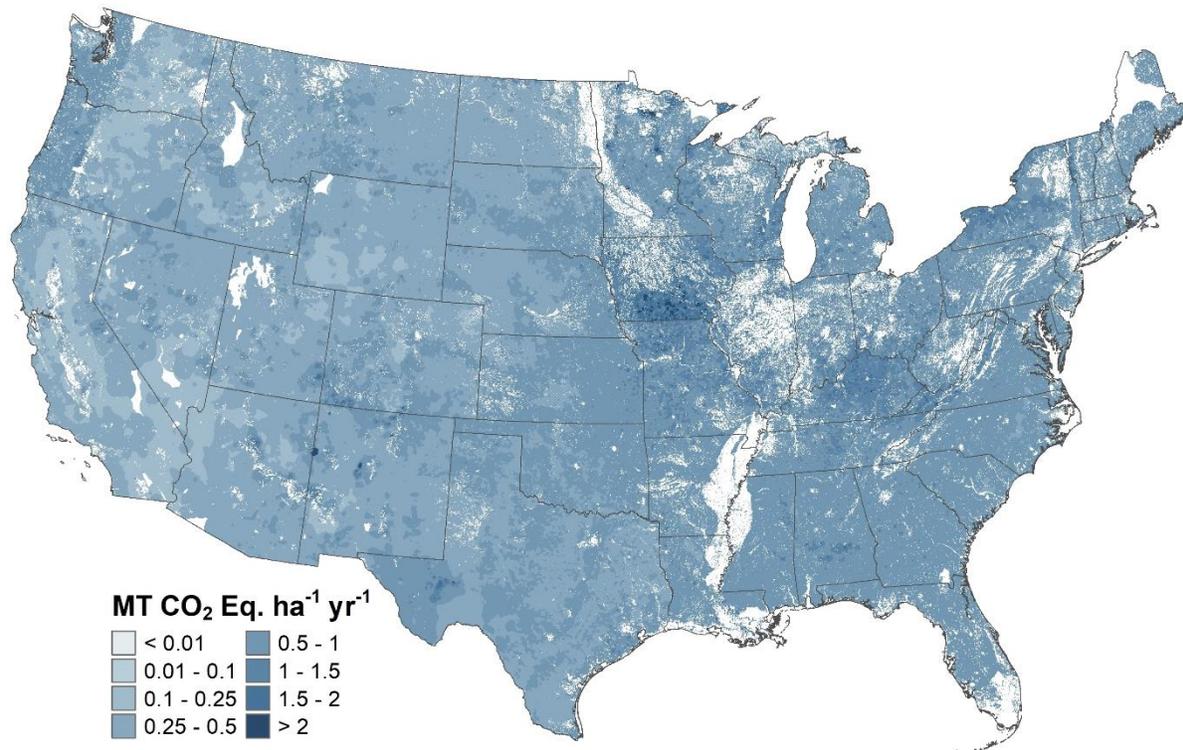


3
4 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
5 emission patterns in this map are based on Inventory data from 2015.

6 Direct N₂O emissions from croplands occur throughout all of the cropland regions but tend to be high in the
7 Midwestern Corn Belt Region (Illinois, Iowa, Indiana, Ohio, southern Minnesota and Wisconsin, and eastern
8 Nebraska), where a large portion of the land is used for growing highly fertilized corn and N-fixing soybean crops
9 (see Figure 5-5). Kansas, South Dakota and North Dakota have relatively high emissions from large areas of crop
10 production that are found in the Great Plains region. Emissions are also high in the Lower Mississippi River Basin
11 from Missouri to Louisiana, and highly productive irrigated areas, such as Platte River, which flows from Colorado
12 through Nebraska, Snake River Valley in Idaho and the Central Valley in California. Direct emissions are low in
13 many parts of the eastern United States because only a small portion of land is cultivated, and in many western
14 states where rainfall and access to irrigation water are limited.

15 Direct emissions from grasslands are more evenly distributed throughout the United States (see Figure 5-6), but
16 total emissions tend to be highest in the Great Plains and western United States where a large proportion of the land
17 is dominated by grasslands with cattle and sheep grazing. However, there are relatively large emissions from local
18 areas in the Eastern United States, particularly Kentucky and Tennessee, in addition to areas in Missouri and Iowa,
19 where there can be higher rates of Pasture/Range/Paddock (PRP) manure N additions on a relatively small amount
20 of pasture. These areas have greater stocking rates of livestock per unit of area, compared to other regions of the
21 United States.

1 **Figure 5-6: Grasslands, 2015 Annual Direct N₂O Emissions Estimated Using the Tier 3**
 2 **DayCent Model**

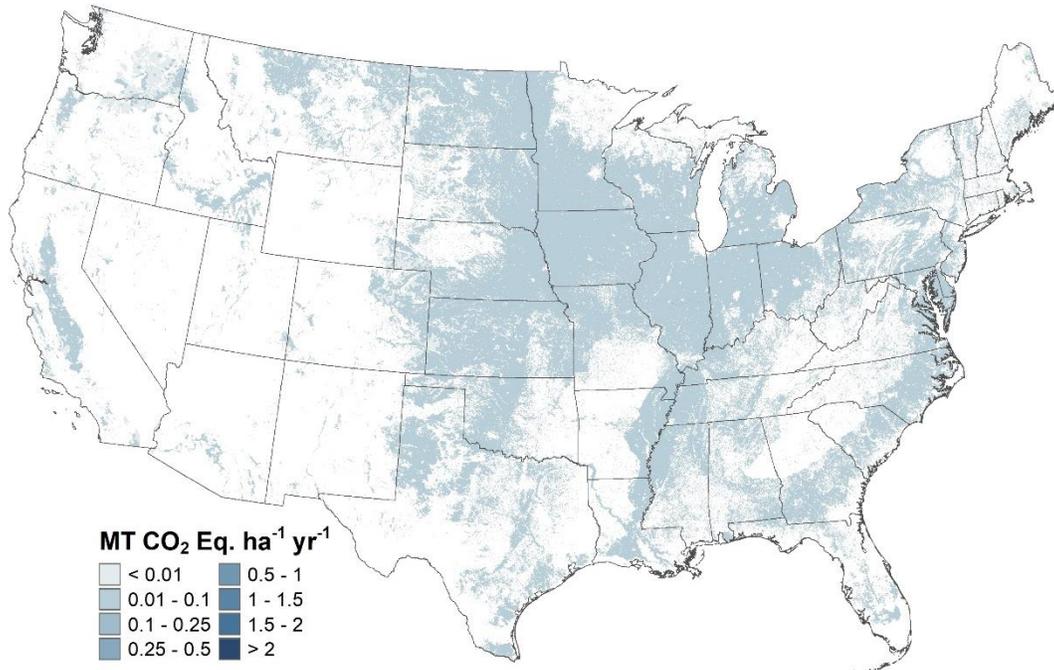


3
 4 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
 5 emission patterns in this map are based on Inventory data from 2015.

6 Indirect N₂O emissions from volatilization in croplands have a similar pattern as the direct N₂O emissions with
 7 higher emissions in the Midwestern Corn Belt, Lower Mississippi River Basin and Great Plains. Indirect N₂O
 8 emissions from volatilization in grasslands are higher in the Southeastern United States, along with portions of the
 9 Mid-Atlantic and southern Iowa. The higher emissions in this region are mainly due to large additions of PRP
 10 manure N on relatively small but productive pastures that support intensive grazing, which in turn, stimulates NH₃
 11 volatilization.

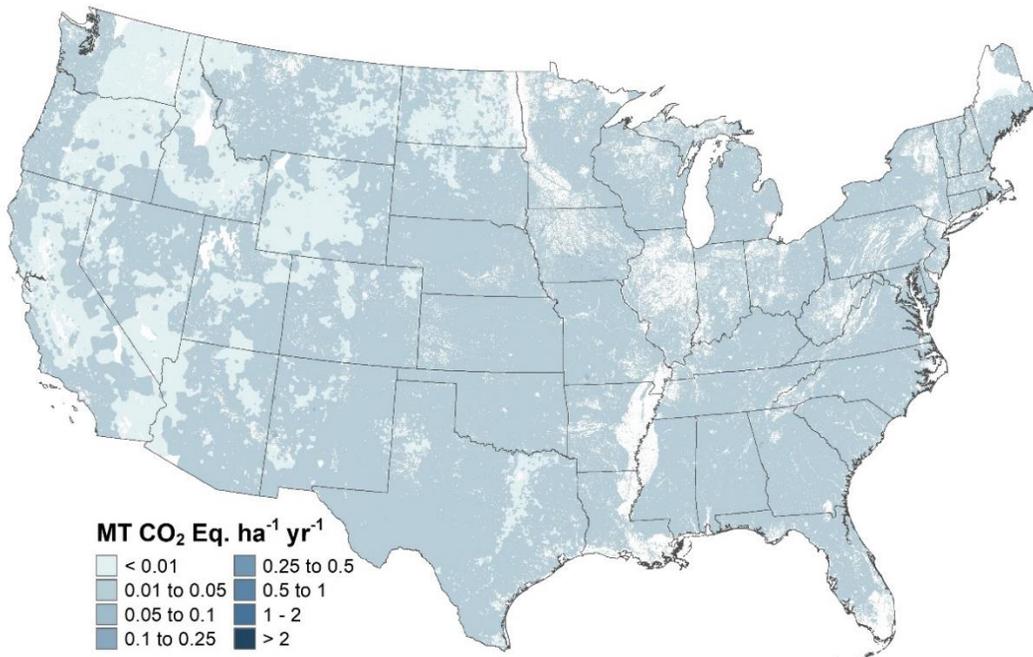
12 Indirect N₂O emissions from surface runoff and leaching of applied/mineralized N in croplands is highest in the
 13 Midwestern Corn Belt. There are also relatively high emissions associated with N management in the Lower
 14 Mississippi River Basin, Piedmont region of the Southeastern United States and the Mid-Atlantic states. In addition,
 15 areas of high emissions occur in portions of the Great Plains that have relatively large areas of irrigated croplands
 16 with high leaching rates of applied/mineralized N. Indirect N₂O emissions from surface runoff and leaching of
 17 applied/mineralized N in grasslands are higher in the eastern United States and coastal Northwest region. These
 18 regions have greater precipitation and higher levels of leaching and runoff compared to arid to semi-arid regions in
 19 the Western United States.

1 **Figure 5-7: Croplands, 2015 Annual Indirect N₂O Emissions from Volatilization Using the**
 2 **Tier 3 DayCent Model**



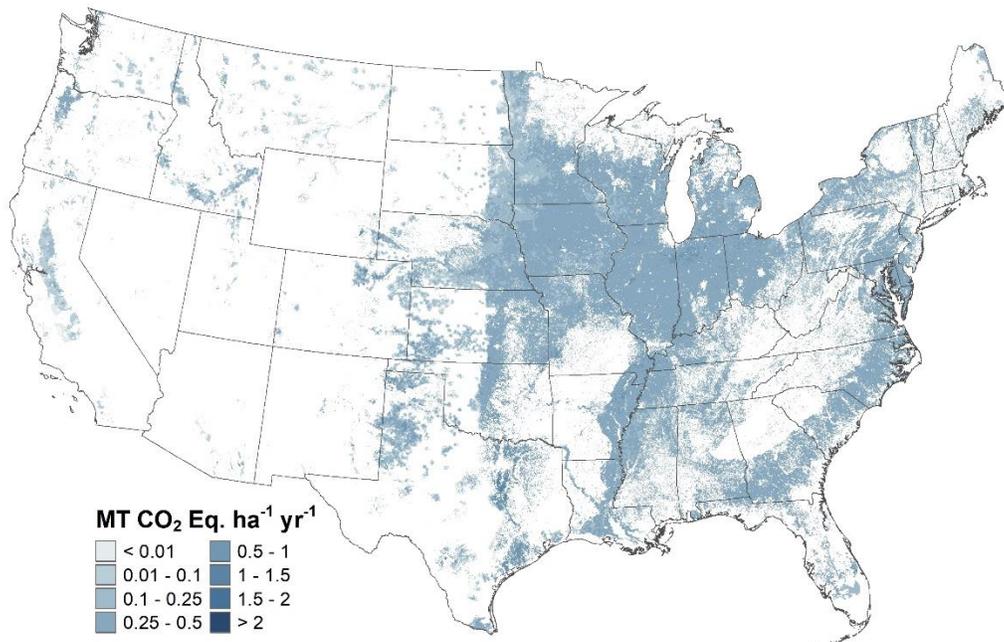
3
 4 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
 5 emission patterns in this map are based on Inventory data from 2015.

6 **Figure 5-8: Grasslands, 2015 Annual Indirect N₂O Emissions from Volatilization Using the**
 7 **Tier 3 DayCent Model**



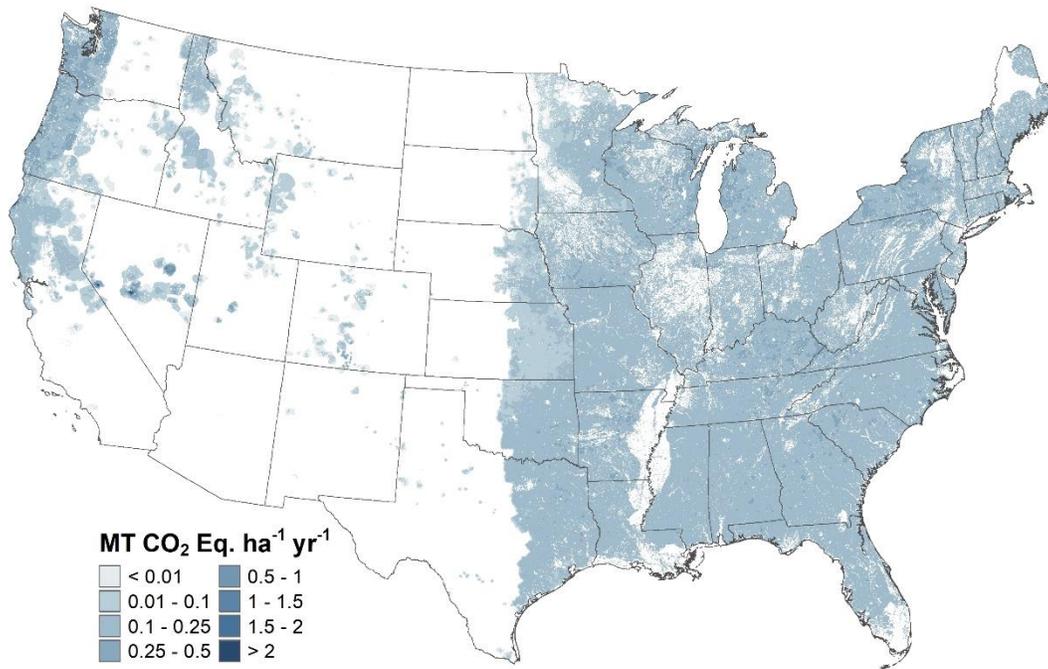
8
 9 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
 10 emission patterns in this map are based on Inventory data from 2015.

1 **Figure 5-9: Croplands, 2015 Annual Indirect N₂O Emissions from Leaching and Runoff Using**
 2 **the Tier 3 DayCent Model**



3
 4 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
 5 emission patterns in this map are based on Inventory data from 2015.

6 **Figure 5-10: Grasslands, 2015 Annual Indirect N₂O Emissions from Leaching and Runoff**
 7 **Using the Tier 3 DayCent Model**



8
 9 Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale
 10 emission patterns in this map are based on Inventory data from 2015.

1 Methodology

2 The *2006 IPCC Guidelines* (IPCC 2006) divide emissions from the agricultural soil management source category into
3 five components, including (1) direct emissions from N additions to cropland and grassland mineral soils from
4 synthetic fertilizers, biosolids (i.e., treated sewage sludge), crop residues (legume N-fixing and non-legume crops),
5 and organic amendments; (2) direct emissions from soil organic matter mineralization due to land use and
6 management change; (3) direct emissions from drainage of organic soils in croplands and grasslands; (4) direct
7 emissions from soils due to manure deposited by livestock on PRP grasslands; and (5) indirect emissions from soils
8 and water from N additions and manure deposition to soils that lead to volatilization, leaching, or runoff of N and
9 subsequent conversion to N₂O.

10 In this source category, the United States reports on all croplands, as well as all managed grasslands, whereby
11 anthropogenic greenhouse gas emissions are estimated consistent with the managed land concept (IPCC 2006),
12 including direct and indirect N₂O emissions from asymbiotic fixation¹⁸ and mineralization of N associated with
13 decomposition of soil organic matter and residues. One recommendation from IPCC (2006) that has not been
14 completely adopted is the estimation of emissions from grassland pasture renewal, which involves occasional
15 plowing to improve forage production in pastures. Currently no data are available to address pasture renewal.

16 Direct N₂O Emissions

17 The methodology used to estimate direct N₂O emissions from agricultural soil management in the United States is
18 based on a combination of IPCC Tier 1 and 3 approaches, along with application of a splicing method for latter
19 years in the Inventory time series (IPCC 2006; Del Grosso et al. 2010) where data are not yet available. A Tier 3
20 process-based model (DayCent) is used to estimate direct emissions from a variety of crops that are grown on
21 mineral (i.e., non-organic) soils, as well as the direct emissions from non-federal grasslands except for applications
22 of biosolids (i.e., treated sewage sludge) (Del Grosso et al. 2010). The Tier 3 approach has been specifically
23 designed and tested to estimate N₂O emissions in the United States, accounting for more of the environmental and
24 management influences on soil N₂O emissions than the IPCC Tier 1 method (see Box 5-3 for further elaboration).
25 Moreover, the Tier 3 approach addresses direct N₂O emissions and soil C stock changes from mineral cropland soils
26 in a single analysis. Carbon and N dynamics are linked in plant-soil systems through biogeochemical processes of
27 microbial decomposition and plant production (McGill and Cole 1981). Coupling the two source categories (i.e.,
28 agricultural soil C and N₂O) in a single inventory analysis ensures that there is consistent activity data and
29 treatment of the processes, and interactions are considered between C and N cycling in soils.

30 The Tier 3 approach is based on the crop and land use histories recorded in the USDA National Resources Inventory
31 (NRI) (USDA-NRCS 2018a). The NRI is a statistically-based sample of all non-federal land,¹⁹ and includes 349,464
32 points on agricultural land for the conterminous United States that are included in the Tier 3 method. The Tier 1
33 approach is used to estimate the emissions from 175,527 locations in the NRI survey across the time series, which
34 are designated as cropland or grassland (discussed later in this section). Each survey location is associated with an
35 “expansion factor” that allows scaling of N₂O emissions from NRI points to the entire country (i.e., each expansion
36 factor represents the amount of area with the same land-use/management history as the survey location). Each
37 NRI survey location was sampled on a 5-year cycle from 1982 until 1997. For cropland, data were collected in 4 out
38 of 5 years in the cycle (i.e., 1979 through 1982, 1984 through 1987, 1989 through 1992, and 1994 through 1997).
39 In 1998, the NRI program began collecting annual data, which are currently available through 2015 (USDA-NRCS
40 2018a).

¹⁸ N inputs from asymbiotic N fixation are not directly addressed in *2006 IPCC Guidelines*, but are a component of the N inputs and total emissions from managed lands and are included in the Tier 3 approach developed for this source.

¹⁹ The NRI survey does include sample points on federal lands, but the program does not collect data from those sample locations.

1 Box 5-3: Tier 1 vs. Tier 3 Approach for Estimating N₂O Emissions

The IPCC (2006) Tier 1 approach is based on multiplying activity data on different N inputs (i.e., synthetic fertilizer, manure, N fixation, etc.) by the appropriate default IPCC emission factors to estimate N₂O emissions on an input-by-input basis. The Tier 1 approach requires a minimal amount of activity data, readily available in most countries (e.g., total N applied to crops); calculations are simple; and the methodology is highly transparent. In contrast, the Tier 3 approach developed for this Inventory is based on application of a process-based model (i.e., DayCent) that represents the interaction of N inputs, land use and management, as well as environmental conditions at specific locations, such as freeze-thaw effects that generate hot moments of N₂O emissions (Wagner-Riddle et al. 2017). Consequently, the Tier 3 approach accounts for land-use and management impacts and their interaction with environmental factors, such as weather patterns and soil characteristics, in a more comprehensive manner, which will enhance or dampen anthropogenic influences. However, the Tier 3 approach requires more detailed activity data (e.g., crop-specific N fertilization rates), additional data inputs (e.g., daily weather, soil types), and considerable computational resources and programming expertise. The Tier 3 methodology is less transparent, and thus it is critical to evaluate the output of Tier 3 methods against measured data in order to demonstrate that the method is an improvement over lower tier methods for estimating emissions (IPCC 2006). Another important difference between the Tier 1 and Tier 3 approaches relates to assumptions regarding N cycling. Tier 1 assumes that N added to a system is subject to N₂O emissions only during that year and cannot be stored in soils and contribute to N₂O emissions in subsequent years. This is a simplifying assumption that may create bias in estimated N₂O emissions for a specific year. In contrast, the process-based model in the Tier 3 approach includes the legacy effect of N added to soils in previous years that is re-mineralized from soil organic matter and emitted as N₂O during subsequent years.

2
3 DayCent is used to estimate N₂O emissions associated with production of alfalfa hay, barley, corn, cotton, grass
4 hay, grass-clover hay, oats, peanuts, potatoes, rice, sorghum, soybeans, sugar beets, sunflowers, tobacco and
5 wheat, but is not applied to estimate N₂O emissions from other crops or rotations with other crops,²⁰ such as
6 sugarcane, some vegetables, and perennial/horticultural crops. Areas that are converted between agriculture (i.e.,
7 cropland and grassland) and other land uses, such as forest land, wetland and settlements, are not simulated with
8 DayCent. DayCent is also not used to estimate emissions from land areas with very gravelly, cobbly, or shaley soils
9 in the topsoil (greater than 35 percent by volume in the top 30 cm of the soil profile), or to estimate emissions
10 from drained organic soils (Histosols). The Tier 3 method has not been fully tested for estimating N₂O emissions
11 associated with these crops and rotations, land uses, as well as organic soils or cobbly, gravelly, and shaley mineral
12 soils. In addition, federal grassland areas are not simulated with DayCent due to limited activity data on land use
13 histories. For areas that are not included in the DayCent simulations, Tier 1 methods are used to estimate
14 emissions, including (1) direct emissions from N inputs for crops on mineral soils that are not simulated by
15 DayCent; (2) direct emissions from PRP N additions on federal grasslands; (3) direct emissions for land application
16 of biosolids (i.e., treated sewage sludge) to soils; and (4) direct emissions from drained organic soils in croplands
17 and grasslands.

18 A splicing method is used to estimate soil N₂O emissions from 2016 to 2019 at the national scale because new NRI
19 activity data are not available for those years. Specifically, linear regression models with autoregressive moving-
20 average (ARMA) errors (Brockwell and Davis 2016) are used to estimate the relationship between surrogate data
21 and the 1990 to 2015 emissions that are derived using the Tier 3 method. Surrogate data for these regression
22 models includes corn and soybean yields from USDA-NASS statistics,²¹ and weather data from the PRISM Climate
23 Group (PRISM 2018). For the Tier 1 method, a linear-time series model is used to estimate emissions from 2016 to
24 2019 without surrogate data for most of the N sources (exceptions include biosolids, drainage of organic soils, and

²⁰ A small proportion of the major commodity crop production, such as corn and wheat, is included in the Tier 1 analysis because these crops are rotated with other crops or land uses (e.g., forest lands) that are not simulated by DayCent.

²¹ See <<https://quickstats.nass.usda.gov/>>.

1 crop residue N). See Box 5-4 for more information about the splicing method. Emission estimates for 2016 to 2019
2 will be recalculated in future Inventory reports when new NRI data are available.

3 **Box 5-4: Surrogate Data Method**

An approach to extend the time series is needed for Agricultural Soil Management because there are typically activity data gaps at the end of the time series. This is mainly because the NRI survey program, which provides critical information for estimating greenhouse gas emissions and removals, does not release data every year.

Splicing methods have been used to impute missing data at the end of the emission time series for both the Tier 1 and 3 methods. Specifically, a linear regression model with autoregressive moving-average (ARMA) errors (Brockwell and Davis 2016) is used to estimate emissions based on the modeled 1990 to 2015 emissions data, which has been compiled using the inventory methods described in this section. The model to extend the time series is given by

$$Y = X\beta + \epsilon,$$

where Y is the response variable (e.g., soil nitrous oxide), Xβ for the Tier 3 method contains specific surrogate data depending on the response variable, and ε is the remaining unexplained error. Models with a variety of surrogate data were tested, including commodity statistics, weather data, or other relevant information. The term Xβ for the Tier 1 method only contains year as a predictor of emission patterns over the time series (change in emissions per year), and therefore, is a linear time series model with no surrogate data. Parameters are estimated from the emissions data for 1990 to 2015 using standard statistical techniques, and these estimates are used in the model described above to predict the missing emissions data for 2016 to 2019.

A critical issue with splicing methods is to account for the additional uncertainty introduced by predicting emissions without compiling the full inventory. Specifically, uncertainty will increase for years with imputed estimates based on the splicing methods, compared to those years in which the full inventory is compiled. This additional uncertainty is quantified within the model framework using a Monte Carlo approach. Consequently, the uncertainty from the original inventory data is combined with the uncertainty in the data splicing model. The approach requires estimating parameters in the data splicing models in each Monte Carlo simulation for the full inventory (i.e., the surrogate data model is refit with the draws of parameters values that are selected in each Monte Carlo iteration, and used to produce estimates with inventory data from 1990 to 2015). Therefore, the data splicing method generates emissions estimates from each surrogate data model in the Monte Carlo analysis, which are used to derive confidence intervals in the estimates for the missing emissions data from 2016 to 2019. Furthermore, the 95 percent confidence intervals are estimated using the 3 sigma rules assuming a unimodal density (Pukelsheim 1994).

4

5 *Tier 3 Approach for Mineral Cropland Soils*

6 The DayCent biogeochemical model (Parton et al. 1998; Del Grosso et al. 2001 and 2011) is used to estimate direct
7 N₂O emissions from mineral cropland soils that are managed for production of a wide variety of crops (see list in
8 previous section) based on the crop histories in the 2015 NRI (USDA-NRCS 2018a). Crops simulated by DayCent are
9 grown on approximately 85 percent of total cropland area in the United States. The model simulates net primary
10 productivity (NPP) using the NASA-CASA production algorithm MODIS Enhanced Vegetation Index (EVI) products,
11 MOD13Q1 and MYD13Q1²² (Potter et al. 1993, 2007). The model simulates soil temperature and water dynamics,
12 using daily weather data from a 4-kilometer gridded product developed by the PRISM Climate Group (2018), and

²² NPP is estimated with the NASA-CASA algorithm for most of the cropland that is used to produce major commodity crops in the central United States from 2000 to 2015. Other regions and years prior to 2000 are simulated with a method that incorporates water, temperature, and moisture stress on crop production (see Metherell et al. 1993), but does not incorporate the additional information about crop condition provided with remote sensing data.

1 soil attributes from the Soil Survey Geographic Database (SSURGO) (Soil Survey Staff 2019). DayCent is used to
2 estimate direct N₂O emissions due to mineral N available from the following sources: (1) application of synthetic
3 fertilizers; (2) application of livestock manure; (3) retention of crop residues in the field for N-fixing legumes and
4 non-legume crops and subsequent mineralization of N during microbial decomposition (i.e., leaving residues in the
5 field after harvest instead of burning or collecting residues); (4) mineralization of N from decomposition of soil
6 organic matter; and (5) asymbiotic fixation.

7 Management activity data from several sources supplement the activity data from the NRI. The USDA-NRCS
8 Conservation Effects and Assessment Project (CEAP) provides data on a variety of cropland management activities,
9 and is used to inform the inventory analysis about tillage practices, mineral fertilization, manure amendments,
10 cover crop management, as well as planting and harvest dates (USDA-NRCS 2018b; USDA-NRCS 2012). CEAP data
11 are collected at a subset of NRI survey locations, and currently provide management information from
12 approximately 2002 to 2006. These data are combined with other datasets in an imputation analysis that extend
13 the time series from 1990 to 2015. This imputation analysis is comprised of three steps: a) determine the trends in
14 management activity across the time series by combining information from several datasets (discussed below), b)
15 use an artificial neural network to determine the likely management practice at a given NRI survey location (Cheng
16 and Titterington 1994), and c) assign management practices from the CEAP survey to specific NRI locations using
17 predictive mean matching methods that are adapted to reflect the trending information (Little 1988, van Buuren
18 2012). The artificial neural network is a machine learning method that approximates nonlinear functions of inputs
19 and searches through a very large class of models to impute an initial value for management practices at specific
20 NRI survey locations. The predictive mean matching method identifies the most similar management activity
21 recorded in the CEAP survey that matches the prediction from the artificial neural network. The matching ensures
22 that imputed management activities are realistic for each NRI survey location, and not odd or physically
23 unrealizable results that could be generated by the artificial neural network. There are six complete imputations of
24 the management activity data using these methods.

25 To determine trends in mineral fertilization and manure amendments from 1979 to 2015, CEAP data are combined
26 with information on fertilizer use and rates by crop type for different regions of the United States from the USDA
27 Economic Research Service. The data collection program was known as the Cropping Practices Surveys through
28 1995 (USDA-ERS 1997), and is now part of data collection known as the Agricultural Resource Management
29 Surveys (ARMS) (USDA-ERS 2018). Additional data on fertilization practices are compiled through other sources
30 particularly the National Agricultural Statistics Service (USDA-NASS 1992, 1999, 2004). The donor survey data from
31 CEAP contain both mineral fertilizer rates and manure amendment rates, so that the selection of a donor via
32 predictive mean matching yields the joint imputation of both rates. This approach captures the relationship
33 between mineral fertilization and manure amendment practices for U.S. croplands based directly on the observed
34 patterns in the CEAP survey data.

35 To determine the trends in tillage management from 1979 to 2015, CEAP data are combined with Conservation
36 Technology Information Center data between 1989 and 2004 (CTIC 2004) and USDA-ERS Agriculture Resource
37 Management Surveys (ARMS) data from 2002 to 2015 (Claasen et al. 2018). The CTIC data are adjusted for long-
38 term adoption of no-till agriculture (Towery 2001). It is assumed that the majority of agricultural lands are
39 managed with full tillage prior to 1985.

40 For cover crops, CEAP data are combined with information from 2011 to 2016 in the USDA Census of Agriculture
41 (USDA-NASS 2012, 2017). It is assumed that cover crop management was minimal prior to 1990 and the rates
42 increased linearly over the decade to the levels of cover crop management in the CEAP survey.

43 The IPCC method considers crop residue N and N mineralized from soil organic matter as activity data. However,
44 they are not treated as activity data in DayCent simulations because residue production, symbiotic N fixation (e.g.,
45 legumes), mineralization of N from soil organic matter, and asymbiotic N fixation are internally generated by the
46 model as part of the simulation. In other words, DayCent accounts for the influence of symbiotic N fixation,
47 mineralization of N from soil organic matter and crop residue retained in the field, and asymbiotic N fixation on
48 N₂O emissions, but these are not model inputs.

49 The N₂O emissions from crop residues are reduced by approximately 3 percent (the assumed average burned
50 portion for crop residues in the United States) to avoid double counting associated with non-CO₂ greenhouse gas

1 emissions from agricultural residue burning. Estimated levels of residue burning are based on state inventory data
2 (ILENR 1993; Oregon Department of Energy 1995; Noller 1996; Wisconsin Department of Natural Resources 1993;
3 Cibrowski 1996).

4 Uncertainty in the emission estimates from DayCent is associated with input uncertainty due to missing
5 management data in the NRI survey that is imputed from other sources; model uncertainty due to incomplete
6 specification of C and N dynamics in the DayCent model parameters and algorithms; and sampling uncertainty
7 associated with the statistical design of the NRI survey. To assess input uncertainty, C and N dynamics at each NRI
8 survey location are simulated six times using the imputation product and other model driver data. Uncertainty in
9 parameterization and model algorithms are determined using a structural uncertainty estimator derived from
10 fitting a linear mixed-effect model (Ogle et al. 2007; Del Grosso et al. 2010). Sampling uncertainty is assessed using
11 NRI replicate sampling weights. These data are combined in a Monte Carlo stochastic simulation with 1,000
12 iterations for 1990 through 2015. For each iteration, there is a random selection of management data from the
13 imputation product (select one of the six imputations), random selection of parameter values and random effects
14 for the linear mixed-effect model (i.e., structural uncertainty estimator), and random selection of a set of survey
15 weights from the replicates associated with the NRI survey design.

16 Nitrous oxide emissions and 95 percent confidence intervals are estimated for each year between 1990 and 2015
17 using the DayCent model. However, note that the areas have been modified in the original NRI survey through a
18 process in which the Forest Inventory and Analysis (FIA) survey data and the National Land Cover Dataset (Yang et
19 al. 2018) are harmonized with the NRI data. This process ensures that the land use areas are consistent across all
20 land use categories (See Section 6.1, Representation of the U.S. Land Base for more information). Further
21 elaboration on the methodology and data used to estimate N₂O emissions from mineral soils are described in
22 Annex 3.12.

23 For the Tier 3 method, soil N₂O emissions from 2016 to 2019 associated with mineral soils in croplands are
24 estimated using a splicing method that accounts for uncertainty in the original inventory data and the splicing
25 method (See Box 5-4). Annual data are currently available through 2015 (USDA-NRCS 2018a), and the Inventory
26 time series will be updated in the future when new NRI data are released.

27 Nitrous oxide emissions from managed agricultural lands are the result of interactions among anthropogenic
28 activities (e.g., N fertilization, manure application, tillage) and other driving variables, such as weather and soil
29 characteristics. These factors influence key processes associated with N dynamics in the soil profile, including
30 immobilization of N by soil microbial organisms, decomposition of organic matter, plant uptake, leaching, runoff,
31 and volatilization, as well as the processes leading to N₂O production (nitrification and denitrification). It is not
32 possible to partition N₂O emissions into each anthropogenic activity directly from model outputs due to the
33 complexity of the interactions (e.g., N₂O emissions from synthetic fertilizer applications cannot be distinguished
34 from those resulting from manure applications). To approximate emissions by activity, the amount of mineral N
35 added to the soil, or made available through decomposition of soil organic matter and plant litter, as well as
36 asymbiotic fixation of N from the atmosphere, is determined for each N source and then divided by the total
37 amount of mineral N in the soil according to the DayCent model simulation. The percentages are then multiplied
38 by the total of direct N₂O emissions in order to approximate the portion attributed to N management practices.
39 This approach is only an approximation because it assumes that all N made available in soil has an equal
40 probability of being released as N₂O, regardless of its source, which is unlikely to be the case (Delgado et al. 2009).
41 However, this approach allows for further disaggregation of emissions by source of N, which is valuable for
42 reporting purposes and is analogous to the reporting associated with the IPCC (2006) Tier 1 method, in that it
43 associates portions of the total soil N₂O emissions with individual sources of N.

44 *Tier 1 Approach for Mineral Cropland Soils*

45 The IPCC (2006) Tier 1 methodology is used to estimate direct N₂O emissions for mineral cropland soils that are not
46 simulated by DayCent (e.g., DayCent has not been parametrized to simulate all crop types and some soil types such
47 as *Histosols*). For the Tier 1 method, estimates of direct N₂O emissions from N applications are based on mineral
48 soil N that is made available from the following practices: (1) the application of synthetic commercial fertilizers; (2)
49 application of managed manure and non-manure commercial organic fertilizers; and (3) decomposition and

1 mineralization of nitrogen from above- and below-ground crop residues in agricultural fields (i.e., crop biomass
2 that is not harvested). Non-manure commercial organic amendments are only included in the Tier 1 analysis
3 because these data are not available at the county-level, which is necessary for the DayCent simulations.
4 Consequently, all commercial organic fertilizer, as well as manure that is not added to crops in the DayCent
5 simulations, are included in the Tier 1 analysis. The following sources are used to derive activity data:

- 6 • A process-of-elimination approach is used to estimate synthetic N fertilizer additions for crop areas that
7 are not simulated by DayCent. The total amount of fertilizer used on farms has been estimated at the
8 county-level by the USGS using sales records from 1990 to 2012 (Brakebill and Gronberg 2017). For 2013
9 through 2015, county-level fertilizer used on-farms is adjusted based on annual fluctuations in total U.S.
10 fertilizer sales (AAPFCO 2013 through 2017).²³ After subtracting the portion of fertilizer applied to crops
11 and grasslands simulated by DayCent (see Tier 3 Approach for Mineral Cropland Soils and Direct N₂O
12 Emissions from Grassland Soils sections for information on data sources), the remainder of the total
13 fertilizer used on farms is assumed to be applied to crops that are not simulated by DayCent.
- 14 • Similarly, a process-of-elimination approach is used to estimate manure N additions for crops that are not
15 simulated by DayCent. The total amount of manure available for land application to soils has been
16 estimated with methods described in the Manure Management section (Section 5.2) and annex (Annex
17 3.11). The amount of manure N applied in the Tier 3 approach to crops and grasslands is subtracted from
18 total annual manure N available for land application (see Tier 3 Approach for Mineral Cropland Soils and
19 Direct N₂O Emissions from Grassland Soils sections for information on data sources). This difference is
20 assumed to be applied to crops that are not simulated by DayCent.
- 21 • Commercial organic fertilizer additions are based on organic fertilizer consumption statistics, which are
22 converted from mass of fertilizer to units of N using average organic fertilizer N content, which range
23 between 2.3 to 4.2 percent across the time series (TVA 1991 through 1994; AAPFCO 1995 through 2017).
24 Commercial fertilizers do include dried manure and biosolids (i.e., treated sewage sludge), but the
25 amounts are removed from the commercial fertilizer data to avoid double counting²⁴ with the manure N
26 dataset described above and the biosolids (i.e., treated sewage sludge) amendment data discussed later
27 in this section.
- 28 • Crop residue N is derived by combining amounts of above- and below-ground biomass, which are
29 determined based on NRI crop area data (USDA-NRCS 2018a), crop production yield statistics (USDA-NASS
30 2019), dry matter fractions (IPCC 2006), linear equations to estimate above-ground biomass given dry
31 matter crop yields from harvest (IPCC 2006), ratios of below-to-above-ground biomass (IPCC 2006), and N
32 contents of the residues (IPCC 2006). N inputs from residue were reduced by 3 percent to account for
33 average residue burning portions in the United States.

34 The total increase in soil mineral N from applied fertilizers and crop residues is multiplied by the IPCC (2006)
35 default emission factor to derive an estimate of direct N₂O emissions using the Tier 1 method. Further elaboration
36 on the methodology and data used to estimate N₂O emissions from mineral soils are described in Annex 3.12.

37 Soil N₂O emissions from 2016 to 2019 for Tier 1 mineral soil emissions are estimated using a splicing method that is
38 described in Box 5-4, with the exception of the crop residue N, which is only estimated with the data splicing
39 method for 2019. As with the Tier 3 method, the time series that is based on the splicing methods will be
40 recalculated in a future Inventory report when updated activity data are available.

²³ The fertilizer consumption data in AAPFCO are recorded in “fertilizer year” totals, (i.e., July to June), but are converted to calendar year totals. This is done by assuming that approximately 35 percent of fertilizer usage occurred from July to December and 65 percent from January to June (TVA 1992b).

²⁴ Commercial organic fertilizers include dried blood, tankage, compost, and other, but the dried manure and biosolids (i.e., treated sewage sludge) are also included in other datasets in this Inventory. Consequently, the proportions of dried manure and biosolids, which are provided in the reports (TVA 1991 through 1994; AAPFCO 1995 through 2017), are used to estimate the N amounts in dried manure and biosolids. To avoid double counting, the resulting N amounts for dried manure and biosolids are subtracted from the total N in commercial organic fertilizers before estimating emissions using the Tier 1 method.

1 *Tier 1 and 3 Approaches for Direct N₂O Emissions from Mineral Grassland Soils*

2 As with N₂O emissions from croplands, the Tier 3 process-based DayCent model and Tier 1 method described in
3 IPCC (2006) are combined to estimate emissions from non-federal grasslands and PRP manure N additions for
4 federal grasslands, respectively. Grassland includes pasture and rangeland that produce grass or mixed
5 grass/legume forage primarily for livestock grazing. Rangelands are extensive areas of native grassland that are not
6 intensively managed, while pastures are seeded grassland (possibly following tree removal) that may also have
7 additional management, such as irrigation, fertilization, or inter-seeding legumes. DayCent is used to simulate N₂O
8 emissions from NRI survey locations (USDA-NRCS 2018a) on non-federal grasslands resulting from manure
9 deposited by livestock directly onto pastures and rangelands (i.e., PRP manure), N fixation from legume seeding,
10 managed manure amendments (i.e., manure other than PRP manure such as Daily Spread or manure collected
11 from other animal waste management systems such as lagoons and digesters), and synthetic fertilizer application.
12 Other N inputs are simulated within the DayCent framework, including N input from mineralization due to
13 decomposition of soil organic matter and N inputs from senesced grass litter, as well as asymbiotic fixation of N
14 from the atmosphere. The simulations used the same weather, soil, and synthetic N fertilizer data as discussed
15 under the Tier 3 Approach in the Mineral Cropland Soils section. Mineral N fertilization rates are based on data
16 from the Carbon Sequestration Rural Appraisals (CSRA) conducted by the USDA-NRCS (USDA-NRCS, unpublished
17 data). The CSRA was a solicitation of expert knowledge from USDA-NRCS staff throughout the United States to
18 support the Inventory. Biological N fixation is simulated within DayCent, and therefore is not an input to the
19 model.

20 Manure N deposition from grazing animals in PRP systems (i.e., PRP manure N) is a key input of N to grasslands.
21 The amounts of PRP manure N applied on non-federal grasslands for each NRI survey location are based on the
22 amount of N excreted by livestock in PRP systems that is estimated in the Manure Management section (See
23 Section 5.2 and Annex 3.10). The total amount of N excreted in each county is divided by the grassland area to
24 estimate the N input rate associated with PRP manure. The resulting rates are a direct input into the DayCent
25 simulations. The N input is subdivided between urine and dung based on a 50:50 split. DayCent simulations of non-
26 federal grasslands accounted for approximately 61 percent of total PRP manure N in aggregate across the
27 country.²⁵ The remainder of the PRP manure N in each state is assumed to be excreted on federal grasslands, and
28 the N₂O emissions are estimated using the IPCC (2006) Tier 1 method.

29 Biosolids (i.e., treated sewage sludge) are assumed to be applied on grasslands because of the heavy metal content
30 and other pollutants in human waste that limit its use as an amendment to croplands. Application of biosolids is
31 estimated from data compiled by EPA (1993, 1999, 2003), McFarland (2001), and NEBRA (2007) (see Section 7.2
32 Wastewater Treatment for a detailed discussion of the methodology for estimating treated sewage sludge
33 available for land application application). Biosolids data are only available at the national scale, and it is not
34 possible to associate application with specific soil conditions and weather at NRI survey locations. Therefore,
35 DayCent could not be used to simulate the influence of biosolids on N₂O emissions from grassland soils, and
36 consequently, emissions from biosolids are estimated using the IPCC (2006) Tier 1 method.

37 Soil N₂O emission estimates from DayCent are adjusted using a structural uncertainty estimator accounting for
38 uncertainty in model algorithms and parameter values (Del Grosso et al. 2010). There is also sampling uncertainty
39 for the NRI survey that is propagated through the estimate with replicate sampling weights associated with the
40 survey. N₂O emissions for the PRP manure N deposited on federal grasslands and applied biosolids N are estimated
41 using the Tier 1 method by multiplying the N input by the default emission factor. Emissions from manure N are
42 estimated at the state level and aggregated to the entire country, but emissions from biosolids N are calculated
43 exclusively at the national scale. Further elaboration on the methodology and data used to estimate N₂O emissions
44 from mineral soils are described in Annex 3.12.

45 Soil N₂O emissions and 95 percent confidence intervals are estimated for each year between 1990 and 2015 based
46 on the Tier 1 and 3 methods, with the exception of biosolids (discussed below). Emissions from 2016 to 2019 are

²⁵ A small amount of PRP N (less than 1 percent) is deposited in grazed pasture that is in rotation with annual crops, and is reported in the grassland N₂O emissions.

1 estimated using a splicing method as described in Box 5-4. As with croplands, estimates for 2016 to 2019 will be
2 recalculated in a future Inventory when new NRI data are released by USDA. Biosolids application data are
3 compiled through 2019 in this Inventory, and therefore soil N₂O emissions and confidence intervals are estimated
4 using the Tier 1 method for all years in the time series without application of the splicing method.

5 *Tier 1 Approach for Drainage of Organic Soils in Croplands and Grasslands*

6 The IPCC (2006) Tier 1 method is used to estimate direct N₂O emissions due to drainage of organic soils in
7 croplands and grasslands at a state scale. State-scale estimates of the total area of drained organic soils are
8 obtained from the 2015 NRI (USDA-NRCS 2018a) using soils data from the Soil Survey Geographic Database
9 (SSURGO) (Soil Survey Staff 2019). Temperature data from the PRISM Climate Group (PRISM 2018) are used to
10 subdivide areas into temperate and tropical climates according to the climate classification from IPCC (2006). To
11 estimate annual emissions, the total temperate area is multiplied by the IPCC default emission factor for
12 temperate regions, and the total tropical area is multiplied by the IPCC default emission factor for tropical regions
13 (IPCC 2006). Annual NRI data are only available between 1990 and 2015, but the time series was adjusted using
14 data from the Forest Inventory and Analysis Program (USFS 2019) in order to estimate emissions from 2016 to
15 2018. The land representation data have not been updated for this Inventory so the amount of drained organic
16 soils is assumed to be the same in 2019 as the estimated areas in 2018, and consequently the emissions in 2019
17 are also assumed to be the same as 2018. Further elaboration on the methodology and data used to estimate N₂O
18 emissions from organic soils are described in Annex 3.12.

19 **Total Direct N₂O Emissions from Cropland and Grassland Soils**

20 Annual direct emissions from the Tier 1 and 3 approaches for mineral and drained organic soils occurring in both
21 croplands and grasslands are summed to obtain the total direct N₂O emissions from agricultural soil management
22 (see Table 5-16 and Table 5-17).

23 **Indirect N₂O Emissions Associated with Nitrogen Management in Cropland and 24 Grasslands**

25 Indirect N₂O emissions occur when mineral N applied or made available through anthropogenic activity is
26 transported from the soil either in gaseous or aqueous forms and later converted into N₂O. There are two
27 pathways leading to indirect emissions. The first pathway results from volatilization of N as NO_x and NH₃ following
28 application of synthetic fertilizer, organic amendments (e.g., manure, biosolids), and deposition of PRP manure.
29 Nitrogen made available from mineralization of soil organic matter and residue, including N incorporated into
30 crops and forage from symbiotic N fixation, and input of N from asymbiotic fixation also contributes to volatilized
31 N emissions. Volatilized N can be returned to soils through atmospheric deposition, and a portion of the deposited
32 N is emitted to the atmosphere as N₂O. The second pathway occurs via leaching and runoff of soil N (primarily in
33 the form of NO₃⁻) that is made available through anthropogenic activity on managed lands, mineralization of soil
34 organic matter and residue, including N incorporated into crops and forage from symbiotic N fixation, and inputs of
35 N into the soil from asymbiotic fixation. The NO₃⁻ is subject to denitrification in water bodies, which leads to N₂O
36 emissions. Regardless of the eventual location of the indirect N₂O emissions, the emissions are assigned to the
37 original source of the N for reporting purposes, which here includes croplands and grasslands.

38 *Tier 1 and 3 Approaches for Indirect N₂O Emissions from Atmospheric Deposition of Volatilized N*

39 The Tier 3 DayCent model and IPCC (2006) Tier 1 methods are combined to estimate the amount of N that is
40 volatilized and eventually emitted as N₂O. DayCent is used to estimate N volatilization for land areas whose direct
41 emissions are simulated with DayCent (i.e., most commodity and some specialty crops and most grasslands). The N
42 inputs included are the same as described for direct N₂O emissions in the Tier 3 Approach for Mineral Cropland
43 Soils and Direct N₂O Emissions from Grassland Soils sections. Nitrogen volatilization from all other areas is
44 estimated using the Tier 1 method with default IPCC fractions for N subject to volatilization (i.e., N inputs on

1 croplands not simulated by DayCent, PRP manure N excreted on federal grasslands, and biosolids [i.e., treated
2 sewage sludge] application on grasslands).

3 The IPCC (2006) default emission factor is multiplied by the amount of volatilized N generated from both DayCent
4 and Tier 1 methods to estimate indirect N₂O emissions occurring following re-deposition of the volatilized N (see
5 Table 5-19). Further elaboration on the methodology and data used to estimate indirect N₂O emissions are
6 described in Annex 3.12.

7 *Tier 1 and 3 Approaches for Indirect N₂O Emissions from Leaching/Runoff*

8 As with the calculations of indirect emissions from volatilized N, the Tier 3 DayCent model and IPCC (2006) Tier 1
9 method are combined to estimate the amount of N that is subject to leaching and surface runoff into water bodies,
10 and eventually emitted as N₂O. DayCent is used to simulate the amount of N transported from lands in the Tier 3
11 Approach. Nitrogen transport from all other areas is estimated using the Tier 1 method and the IPCC (2006) default
12 factor for the proportion of N subject to leaching and runoff associated with N applications on croplands that are
13 not simulated by DayCent, applications of biosolids on grasslands, and PRP manure N excreted on federal
14 grasslands.

15 For both the DayCent Tier 3 and IPCC (2006) Tier 1 methods, nitrate leaching is assumed to be an insignificant
16 source of indirect N₂O in cropland and grassland systems in arid regions, as discussed in IPCC (2006). In the United
17 States, the threshold for significant nitrate leaching is based on the potential evapotranspiration (PET) and rainfall
18 amount, similar to IPCC (2006), and is assumed to be negligible in regions where the amount of precipitation does
19 not exceed 80 percent of PET (Note: All irrigated systems are assumed to have significant amounts of leaching of N
20 even in drier climates).

21 For leaching and runoff data estimated by the Tier 3 and Tier 1 approaches, the IPCC (2006) default emission factor
22 is used to estimate indirect N₂O emissions that occur in groundwater and waterways (see Table 5-19). Further
23 elaboration on the methodology and data used to estimate indirect N₂O emissions are described in Annex 3.12.

24 Indirect soil N₂O emissions from 2016 to 2019 are estimated using the splicing method that is described in Box 5-4.
25 As with the direct N₂O emissions, the time series will be recalculated in a future Inventory report when new
26 activity data are compiled.

27 **Uncertainty and Time-Series Consistency**

28 Uncertainty is estimated for each of the following five components of N₂O emissions from agricultural soil
29 management: (1) direct emissions simulated by DayCent; (2) the components of indirect emissions (N volatilized
30 and leached or runoff) simulated by DayCent; (3) direct emissions estimated with the IPCC (2006) Tier 1 method;
31 (4) the components of indirect emissions (N volatilized and leached or runoff) estimated with the IPCC (2006) Tier
32 1 method; and (5) indirect emissions estimated with the IPCC (2006) Tier 1 method. Uncertainty in direct emissions
33 as well as the components of indirect emissions that are estimated from DayCent are derived from a Monte Carlo
34 Analysis (consistent with IPCC Approach 2), addressing uncertainties in model inputs and structure (i.e., algorithms
35 and parameterization) (Del Grosso et al. 2010). For 2016 to 2019, there is additional uncertainty propagated
36 through the Monte Carlo Analysis associated with the splicing method (See Box 5-4).

37 Simple error propagation methods (IPCC 2006) are used to derive confidence intervals for direct emissions
38 estimated with the IPCC (2006) Tier 1 method, the proportion of volatilization and leaching or runoff estimated
39 with the IPCC (2006) Tier 1 method, and indirect N₂O emissions. Uncertainty in the splicing method is also included
40 in the error propagation for 2016 to 2019 (see Box 5-4). Additional details on the uncertainty methods are
41 provided in Annex 3.12.

42 Table 5-20 shows the combined uncertainty for direct soil N₂O emissions. The estimated emissions ranges from 31
43 percent below to 31 percent above the 2019 emission estimate of 290.4 MMT CO₂ Eq. The combined uncertainty
44 for indirect soil N₂O emissions ranges from 71 percent below to 154 percent above the 2019 estimate of 54.2 MMT
45 CO₂ Eq.

Table 5-20: Quantitative Uncertainty Estimates of N₂O Emissions from Agricultural Soil Management in 2019 (MMT CO₂ Eq. and Percent)

Source	Gas	2019 Emission				
		Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate			
			(MMT CO ₂ Eq.)		(%)	
		Lower Bound	Upper Bound	Lower Bound	Upper Bound	
Direct Soil N ₂ O Emissions	N ₂ O	290.4	200.7	380.1	-31%	31%
Indirect Soil N ₂ O Emissions	N ₂ O	54.2	16.0	137.5	-71%	154%

Note: Due to lack of data, uncertainties in PRP manure N production, other organic fertilizer amendments, and biosolids (i.e., treated sewage sludge) amendments to soils are currently treated as certain; these sources of uncertainty will be included in future Inventory reports.

Additional uncertainty is associated with an incomplete estimation of N₂O emissions from managed croplands and grasslands in Hawaii and Alaska. The Inventory currently includes the N₂O emissions from mineral fertilizer and PRP N additions in Alaska and Hawaii, and drained organic soils in Hawaii. Land areas used for agriculture in Alaska and Hawaii are small relative to major crop commodity states in the conterminous United States, so the emissions are likely to be small for the other sources of N (e.g., crop residue inputs), which are not currently included in the Inventory.

Methodological recalculations were applied to the entire time series to ensure time-series consistency from 1990 through 2019. Details on the emission trends and methodologies through time are described in more detail in the Introduction and Methodology sections.

QA/QC and Verification

General (Tier 1) and category-specific (Tier 2) QA/QC activities were conducted consistent with the U.S. Inventory QA/QC plan outlined in Annex 8. DayCent results for N₂O emissions and NO₃⁻ leaching are compared with field data representing various cropland and grassland systems, soil types, and climate patterns (Del Grosso et al. 2005; Del Grosso et al. 2008), and further evaluated by comparing the model results to emission estimates produced using the IPCC (2006) Tier 1 method for the same sites. Nitrous oxide measurement data for cropland are available for 64 sites representing 796 different combinations of fertilizer treatments and cultivation practices, and measurement data for grassland are available for 13 sites representing 36 different management treatments. Nitrate leaching data are available for 12 sites, representing 279 different combinations of fertilizer treatments and tillage practices. In general, DayCent predicted N₂O emission and nitrate leaching for these sites reasonably well. See Annex 3.12 for more detailed information about the comparisons.

Spreadsheets containing input data and probability distribution functions required for DayCent simulations of croplands and grasslands and unit conversion factors have been checked, in addition to the program scripts that are used to run the Monte Carlo uncertainty analysis. Links between spreadsheets have also been checked, updated, and corrected when necessary. Spreadsheets containing input data, emission factors, and calculations required for the Tier 1 method have been checked and updated as needed.

Recalculations Discussion

One improvement has been implemented in this Inventory leading to the need for recalculations. This improvement was an update to the time series of PRP and manure N available for application to soils, in order to be consistent with the data generated for the Manure Management section of this Inventory. The surrogate data method was also applied to re-estimate N₂O emissions from 2016 to 2018. These changes resulted in an average increase in emissions of 0.1 percent from 1990 to 2018 relative to the previous Inventory.

1 Planned Improvements

2 A key improvement for a future Inventory will be to incorporate additional management activity data from the
3 USDA-NRCS Conservation Effects Assessment Project survey. This survey has compiled new data in recent years
4 that will be available for the Inventory analysis by next year. The latest land use data will also be incorporated from
5 the USDA National Resources Inventory and related management data from USDA-ERS ARMS surveys.

6 Several planned improvements are underway associated with improving the DayCent biogeochemical model.
7 These improvements include a better representation of plant phenology, particularly senescence events following
8 grain filling in crops. In addition, crop parameters associated with temperature and water stress effects on plant
9 production will be further improved in DayCent with additional model calibration. Model development is
10 underway to represent the influence of nitrification inhibitors and slow-release fertilizers (e.g., polymer-coated
11 fertilizers) on N₂O emissions. Experimental study sites will continue to be added for quantifying model structural
12 uncertainty. Studies that have continuous (daily) measurements of N₂O (e.g., Scheer et al. 2013) will be given
13 priority.

14 Improvements are underway to simulate crop residue burning in the DayCent model based on the amount of crop
15 residues burned according to the data that is used in the Field Burning of Agricultural Residues source category
16 (see Section 5.7). Alaska and Hawaii are not included for all sources in the current Inventory for agricultural soil
17 management, with the exception of N₂O emissions from drained organic soils in croplands and grasslands for
18 Hawaii, synthetic fertilizer and PRP N amendments for grasslands in Alaska and Hawaii. There is also an
19 improvement based on updating the Tier 1 emission factor for N₂O emissions from drained organic soils by using
20 the revised factor in the 2013 Supplement to the *2006 IPCC Guidelines for National Greenhouse Gas Inventories:
21 Wetlands* (IPCC 2013).

22 In addition, there is a planned improvement associated with implementation of the Tier 1 method. Specifically, soil
23 N₂O emissions will be estimated and reported for N mineralization from soil organic matter decomposition that is
24 accelerated with *Forest Land Converted to Cropland* and *Grassland Converted to Cropland*.

25 These improvements are expected to be completed for the next full Inventory analysis (i.e., 2022 submission to the
26 UNFCCC, 1990 through 2020 Inventory). However, the timeline may be extended if there are insufficient resources
27 to fund all or part of these planned improvements.

28 5.5 Liming (CRF Source Category 3G)

29 Crushed limestone (CaCO₃) and dolomite (CaMg(CO₃)₂) are added to soils by land managers to increase soil pH
30 (i.e., to reduce acidification). Carbon dioxide emissions occur as these compounds react with hydrogen ions in
31 soils. The rate of degradation of applied limestone and dolomite depends on the soil conditions, soil type, climate
32 regime, and whether limestone or dolomite is applied. Emissions from limestone and dolomite that are used in
33 industrial processes (e.g., cement production, glass production, etc.) are reported in the IPPU chapter. Emissions
34 from liming of soils have fluctuated between 1990 and 2019 in the United States, ranging from 2.2 MMT CO₂ Eq. to
35 6.0 MMT CO₂ Eq. across the entire time series. In 2019, liming of soils in the United States resulted in emissions of
36 2.4 MMT CO₂ Eq. (0.7 MMT C), representing a 52 percent decrease in emissions since 1990 (see Table 5-21 and
37 Table 5-22). The trend is driven by variation in the amount of limestone and dolomite applied to soils over the time
38 period.

39 **Table 5-21: Emissions from Liming (MMT CO₂ Eq.)**

Source	1990	2005	2015	2016	2017	2018	2019
Limestone	4.1	3.9	3.5	2.8	2.9	2.0	2.2
Dolomite	0.6	0.4	0.3	0.3	0.2	0.2	0.2
Total	4.7	4.3	3.7	3.1	3.1	2.2	2.4

Note: Totals may not sum due to independent rounding.

1 **Table 5-22: Emissions from Liming (MMT C)**

Source	1990	2005	2015	2016	2017	2018	2019
Limestone	1.1	1.1	0.9	0.8	0.8	0.6	0.6
Dolomite	0.2	0.1	0.1	0.1	0.1	0.1	0.1
Total	1.3	1.2	1.0	0.8	0.8	0.6	0.7

Note: Totals may not sum due to independent rounding.

2 Methodology

3 Carbon dioxide emissions from application of limestone and dolomite to soils were estimated using a Tier 2
 4 methodology consistent with IPCC (2006). The annual amounts of limestone and dolomite, which are applied to
 5 soils (see Table 5-23), were multiplied by CO₂ emission factors from West and McBride (2005). These country-
 6 specific emission factors (0.059 metric ton C/metric ton limestone, 0.064 metric ton C/metric ton dolomite) are
 7 lower than the IPCC default emission factors because they account for the portion of carbonates that are
 8 transported from soils through hydrological processes and eventually deposited in ocean basins (West and
 9 McBride 2005). This analysis of lime dissolution is based on studies in the Mississippi River basin, where the vast
 10 majority of lime application occurs in the United States (West 2008). Moreover, much of the remaining lime
 11 application is occurring under similar precipitation regimes, and so the emission factors are considered a
 12 reasonable approximation for all lime application in the United States (West 2008) (See Box 5-5).

13 The annual application rates of limestone and dolomite were derived from estimates and industry statistics
 14 provided in the *Minerals Yearbook* (Tepordei 1993 through 2006; Willett 2007a, 2007b, 2009, 2010, 2011a, 2011b,
 15 2013a, 2014, 2015, 2016, 2017, 2020a), as well as preliminary data that will eventually be published in the
 16 *Minerals Yearbook* for the latter part of the time series (Willett 2019, 2020b). Data for the final year of the
 17 inventory is based on the *Mineral Industry Surveys*, as discussed below (USGS 2020). The U.S. Geological Survey
 18 (USGS; U.S. Bureau of Mines prior to 1997) compiled production and use information through surveys of crushed
 19 stone manufacturers. However, manufacturers provided different levels of detail in survey responses so the
 20 estimates of total crushed limestone and dolomite production and use were divided into three components: (1)
 21 production by end-use, as reported by manufacturers (i.e., “specified” production); (2) production reported by
 22 manufacturers without end-uses specified (i.e., “unspecified” production); and (3) estimated additional production
 23 by manufacturers who did not respond to the survey (i.e., “estimated” production).

24 **Box 5-5: Comparison of the Tier 2 U.S. Inventory Approach and IPCC (2006) Default Approach**

Emissions from liming of soils were estimated using a Tier 2 methodology based on emission factors specific to the United States that are lower than the IPCC (2006) emission default factors. Most lime application in the United States occurs in the Mississippi River basin, or in areas that have similar soil and rainfall regimes as the Mississippi River basin. Under these conditions, a significant portion of dissolved agricultural lime leaches through the soil into groundwater. Groundwater moves into channels and is transported to larger rivers and eventually the ocean where CaCO₃ precipitates to the ocean floor (West and McBride 2005). The U.S.-specific emission factors (0.059 metric ton C/metric ton limestone and 0.064 metric ton C/metric ton dolomite) are about half of the IPCC (2006) emission factors (0.12 metric ton C/metric ton limestone and 0.13 metric ton C/metric ton dolomite). For comparison, the 2019 U.S. emission estimate from liming of soils is 2.4 MMT CO₂ Eq. using the country-specific factors. In contrast, emissions would be estimated at 5.0 MMT CO₂ Eq. using the IPCC (2006) default emission factors.

25
 26 Data on “specified” limestone and dolomite amounts were used directly in the emission calculation because the
 27 end use is provided by the manufacturers and can be used to directly determine the amount applied to soils.
 28 However, it is not possible to determine directly how much of the limestone and dolomite is applied to soils for

1 manufacturer surveys in the “unspecified” and “estimated” categories. For these categories, the amounts of
 2 crushed limestone and dolomite applied to soils were determined by multiplying the percentage of total
 3 “specified” limestone and dolomite production that is applied to soils, by the total amounts of “unspecified” and
 4 “estimated” limestone and dolomite production. In other words, the proportion of total “unspecified” and
 5 “estimated” crushed limestone and dolomite that was applied to soils is proportional to the amount of total
 6 “specified” crushed limestone and dolomite that was applied to soils.

7 In addition, data were not available for 1990, 1992, and 2019 on the fractions of total crushed stone production
 8 that were limestone and dolomite, and on the fractions of limestone and dolomite production that were applied to
 9 soils. To estimate the 1990 and 1992 data, a set of average fractions were calculated using the 1991 and 1993
 10 data. These average fractions were applied to the quantity of "total crushed stone produced or used" reported for
 11 1990 and 1992 in the 1994 *Minerals Yearbook* (Tepordei 1996). To estimate 2019 data, 2018 fractions were applied
 12 to a 2019 estimate of total crushed stone presented in the USGS *Mineral Industry Surveys: Crushed Stone and Sand
 13 and Gravel in the First Quarter of 2020* (USGS 2020).

14 The primary source for limestone and dolomite activity data is the *Minerals Yearbook*, published by the Bureau of
 15 Mines through 1996 and by the USGS from 1997 to the present. In 1994, the “Crushed Stone” chapter in the
 16 *Minerals Yearbook* began rounding (to the nearest thousand metric tons) quantities for total crushed stone
 17 produced or used. It then reported revised (rounded) quantities for each of the years from 1990 to 1993. In order
 18 to minimize the inconsistencies in the activity data, these revised production numbers have been used in all of the
 19 subsequent calculations.

20 **Table 5-23: Applied Minerals (MMT)**

Mineral	1990	2005	2015	2016	2017	2018	2019
Limestone	19.0	18.1	16.0	13.0	13.4	9.4	10.2
Dolomite	2.4	1.9	1.2	1.1	0.8	0.9	1.0

21 **Uncertainty and Time-Series Consistency**

22 Uncertainty regarding the amount of limestone and dolomite applied to soils was estimated at ±15 percent with
 23 normal densities (Tepordei 2003; Willett 2013b). Analysis of the uncertainty associated with the emission factors
 24 included the fraction of lime dissolved by nitric acid versus the fraction that reacts with carbonic acid, and the
 25 portion of bicarbonate that leaches through the soil and is transported to the ocean. Uncertainty regarding the
 26 time associated with leaching and transport was not addressed in this analysis, but is assumed to be a relatively
 27 small contributor to the overall uncertainty (West 2005). The probability distribution functions for the fraction of
 28 lime dissolved by nitric acid and the portion of bicarbonate that leaches through the soil were represented as
 29 triangular distributions between ranges of zero and 100 percent of the estimates. The uncertainty surrounding
 30 these two components largely drives the overall uncertainty.

31 A Monte Carlo (Approach 2) uncertainty analysis was applied to estimate the uncertainty in CO₂ emissions from
 32 liming. The results of the Approach 2 quantitative uncertainty analysis are summarized in Table 5-24. Carbon
 33 dioxide emissions from carbonate lime application to soils in 2019 were estimated to be between -0.27 and 4.61
 34 MMT CO₂ Eq. at the 95 percent confidence level. This confidence interval represents a range of 111 percent below
 35 to 88 percent above the 2019 emission estimate of 2.4 MMT CO₂ Eq. Note that there is a small probability of a
 36 negative emissions value leading to a net uptake of CO₂ from the atmosphere. Net uptake occurs due to the
 37 dominance of the carbonate lime dissolving in carbonic acid rather than nitric acid (West and McBride 2005).

38
 39
 40

Table 5-24: Approach 2 Quantitative Uncertainty Estimates for CO₂ Emissions from Liming (MMT CO₂ Eq. and Percent)

Source	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a (MMT CO ₂ Eq.) (%)			
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Liming	CO ₂	2.4	(0.27)	4.61	-111%	88%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

Methodological recalculations were applied to the entire time series to ensure time-series consistency from 1990 through 2019. Details on the emission trends and methodologies through time are described in more detail in the Introduction and Methodology sections.

QA/QC and Verification

A source-specific QA/QC plan for liming has been developed and implemented, consistent with the U.S. Inventory QA/QC plan outlined in Annex 8. The quality control effort focused on the Tier 1 procedures for this Inventory. No errors were found.

Recalculations Discussion

An adjustment was made in the current Inventory to improve the results; limestone and dolomite application data for 2018 were updated with the recently published data from USGS (2020), rather than approximated by a ratio method, which was used in the previous Inventory. With this revision in the activity data, the emissions decreased by 28.6 percent for 2018 relative to the previous Inventory.

5.6 Urea Fertilization (CRF Source Category 3H)

The use of urea (CO(NH₂)₂) as a fertilizer leads to greenhouse gas emissions through the release of CO₂ that was fixed during the production of urea. In the presence of water and urease enzymes, urea that is applied to soils as fertilizer is converted into ammonium (NH₄⁺), hydroxyl ion (OH), and bicarbonate (HCO₃⁻). The bicarbonate then evolves into CO₂ and water. Emissions from urea fertilization in the United States is 5.3 MMT CO₂ Eq. (1.5 MMT C) in 2019 (Table 5-25 and Table 5-26). Carbon dioxide emissions have increased by 121 percent between 1990 and 2019 due to an increasing amount of urea that is applied to soils. The variation in emissions across the time series is driven by differences in the amounts of fertilizer applied to soils each year. Carbon dioxide emissions associated with urea that is used for non-agricultural purposes, are reported in the IPPU chapter (Section 4.6).

Table 5-25: CO₂ Emissions from Urea Fertilization (MMT CO₂ Eq.)

Source	1990	2005	2015	2016	2017	2018	2019
Urea Fertilization	2.4	3.5	4.7	4.9	5.1	5.2	5.3

Table 5-26: CO₂ Emissions from Urea Fertilization (MMT C)

Source	1990	2005	2015	2016	2017	2018	2019
Urea Fertilization	0.7	1.0	1.3	1.3	1.4	1.4	1.5

Methodology

Carbon dioxide emissions from the application of urea to agricultural soils were estimated using the IPCC (2006) Tier 1 methodology. The method assumes that C in the urea is released after application to soils and converted to CO₂. The annual amounts of urea applied to croplands (see Table 5-27) were derived from the state-level fertilizer sales data provided in *Commercial Fertilizer* reports (TVA 1991, 1992, 1993, 1994; AAPFCO 1995 through 2018).²⁶ These amounts were multiplied by the default IPCC (2006) emission factor (0.20 metric tons of C per metric ton of urea), which is equal to the C content of urea on an atomic weight basis. The calculations were made using a Monte Carlo analysis as described in the Uncertainty section below.

Fertilizer sales data are reported in fertilizer years (July previous year through June current year) so a calculation was performed to convert the data to calendar years (January through December). According to monthly fertilizer use data (TVA 1992b), 35 percent of total fertilizer used in any fertilizer year is applied between July and December of the previous calendar year, and 65 percent is applied between January and June of the current calendar year.

Fertilizer sales data for the 2016 through 2019 fertilizer years were not available for this Inventory. Therefore, urea application in the 2016 through 2019 fertilizer years were estimated using a linear, least squares trend of consumption over the data from the previous five years (2011 through 2015) at the state scale. A trend of five years was chosen as opposed to a longer trend as it best captures the current inter-state and inter-annual variability in consumption. State-level estimates of CO₂ emissions from the application of urea to agricultural soils were summed to estimate total emissions for the entire United States. The fertilizer year data is then converted into calendar year (Table 5-27) data using the method described above.

Table 5-27: Applied Urea (MMT)

	1990	2005	2015	2016	2017	2018	2019
Urea Fertilizer ^a	3.3	4.8	6.4	6.7	6.9	7.1	7.3

^a These numbers represent amounts applied to all agricultural land, including *Cropland Remaining Cropland, Land Converted to Cropland, Grassland Remaining Grassland, Land Converted to Grassland, Settlements Remaining Settlements, Land Converted to Settlements, Forest Land Remaining Forest Land and Land Converted to Forest Land*, as it is not currently possible to apportion the data by land-use category.

Uncertainty and Time-Series Consistency

An Approach 2 Monte Carlo analysis is conducted as described by the IPCC (2006). The largest source of uncertainty is the default emission factor, which assumes that 100 percent of the C in CO(NH₂)₂ applied to soils is emitted as CO₂. The uncertainty surrounding this factor incorporates the possibility that some of the C may not be emitted to the atmosphere, and therefore the uncertainty range is set from 50 percent emissions to the maximum emission value of 100 percent using a triangular distribution. In addition, urea consumption data have uncertainty that is represented as a normal density. Due to the highly skewed distribution of the resulting emissions from the Monte Carlo uncertainty analysis, the estimated emissions are based on the analytical solution to the equation, and the confidence interval is approximated based on the values at 2.5 and 97.5 percentiles.

Carbon dioxide emissions from urea fertilization of agricultural soils in 2019 are estimated to be between 3.06 and 5.51 MMT CO₂ Eq. at the 95 percent confidence level. This indicates a range of 43 percent below to 3 percent above the 2019 emission estimate of 5.3 MMT CO₂ Eq. (Table 5-28).

²⁶ The amount of urea consumed for non-agricultural purposes in the United States is reported in the Industrial Processes and Product Use chapter, Section 4.6 Urea Consumption for Non-Agricultural Purposes.

Table 5-28: Quantitative Uncertainty Estimates for CO₂ Emissions from Urea Fertilization (MMT CO₂ Eq. and Percent)

Source	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
			(MMT CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Urea Fertilization	CO ₂	5.3	3.06	5.51	-43%	+3%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

There are additional uncertainties that are not quantified in this analysis. There is uncertainty surrounding the assumptions underlying conversion of fertilizer years to calendar years. These uncertainties are negligible over multiple years because an over- or under-estimated value in one calendar year is addressed with a corresponding increase or decrease in the value for the subsequent year. In addition, there is uncertainty regarding the fate of C in urea that is incorporated into solutions of urea ammonium nitrate (UAN) fertilizer. Emissions of CO₂ from UAN applications to soils are not estimated in the current Inventory (see Planned Improvements).

Methodological recalculations were applied to the entire time series to ensure time-series consistency from 1990 through 2019. Details on the emission trends and methodologies are described in the Introduction and Methodology sections.

QA/QC and Verification

A source-specific QA/QC plan for Urea Fertilization has been developed and implemented, consistent with the U.S. Inventory QA/QC plan. One quality control issue was raised by the expert review team (ERT) from the UNFCCC for this emission source. In the previous (i.e., 1990 through 2018) Inventory, estimates of CO₂ emissions were based on the results from the Monte Carlo uncertainty analysis. Specifically, the mode from the Monte Carlo uncertainty analysis was used as the most probable estimate of emissions. The mode differs from the analytical solution to the equation due to the pattern in the probability distribution for CO₂ emissions from the Monte Carlo uncertainty analysis, which combined a normal density for the urea application data with the right triangle distribution for the emission factor. For this Inventory, the analytical solution has been adopted as the estimate of CO₂ emissions for urea fertilization to be consistent with recommendations from ERT. The ERT considered the analytical solution to be more representative of the emissions than the mode from the Monte Carlo uncertainty analysis.

Recalculations Discussion

Emissions estimates were derived directly from the Monte Carlo uncertainty analysis in the previous Inventory as discussed in the QA/QC and Verification section. For this Inventory, the entire time series was recalculated using the analytical solution rather than the mode from the Monte Carlo uncertainty analysis. This change in emission estimates averaged about 15 percent higher across the time series compared to the previous Inventory.

Planned Improvements

A key planned improvement is to incorporate Urea Ammonium Nitrate (UAN) in the estimation of Urea CO₂ emissions. Activity data for UAN have been identified, but additional information is needed to fully incorporate this type of fertilizer into the analysis, which will be completed in a future Inventory.

5.7 Field Burning of Agricultural Residues (CRF Source Category 3F)

Crop production creates large quantities of agricultural crop residues, which farmers manage in a variety of ways. For example, crop residues can be left in the field and possibly incorporated into the soil with tillage; collected and used as fuel, animal bedding material, supplemental animal feed, or construction material; composted and applied to soils; transported to landfills; or burned in the field. Field burning of crop residues is not considered a net source of CO₂ emissions because the C released to the atmosphere as CO₂ during burning is reabsorbed during the next growing season by the crop. However, crop residue burning is a net source of CH₄, N₂O, CO, and NO_x, which are released during combustion.

In the United States, field burning of agricultural residues commonly occurs in southeastern states, the Great Plains, and the Pacific Northwest (McCarty 2011). The primary crops that are managed with residue burning include corn, cotton, lentils, rice, soybeans, and wheat (McCarty 2009). In 2019, CH₄ and N₂O emissions from field burning of agricultural residues were 0.4 MMT CO₂ Eq. (17 kt) and 0.2 MMT CO₂ Eq. (1 kt), respectively (Table 5-29 and Table 5-30). Annual emissions of CH₄ and N₂O have increased from 1990 to 2019 by 14 percent and 16 percent, respectively. The increase in emissions over time is partly due to higher yielding crop varieties with larger amounts of residue production and fuel loads, but also linked with an increase in the area burned for some of the crop types.

Table 5-29: CH₄ and N₂O Emissions from Field Burning of Agricultural Residues (MMT CO₂ Eq.)

Gas/Crop Type	1990	2005	2015	2016	2017	2018	2019
CH₄	0.4						
Maize	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Rice	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Wheat	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Barley	+	+	+	+	+	+	+
Oats	+	+	+	+	+	+	+
Other Small Grains	+	+	+	+	+	+	+
Sorghum	+	+	+	+	+	+	+
Cotton	+	+	+	+	+	+	+
Grass Hay	+	+	+	+	+	+	+
Legume Hay	+	+	+	+	+	+	+
Peas	+	+	+	+	+	+	+
Sunflower	+	+	+	+	+	+	+
Tobacco	+	+	+	+	+	+	+
Vegetables	+	+	+	+	+	+	+
Chickpeas	+	+	+	+	+	+	+
Dry Beans	+	+	+	+	+	+	+
Lentils	+	+	+	+	+	+	+
Peanuts	+	+	+	+	+	+	+
Soybeans	+	+	+	+	+	+	+
Potatoes	+	+	+	+	+	+	+
Sugarbeets	+	+	+	+	+	+	+
N₂O	0.2						
Maize	+	+	0.1	0.1	0.1	0.1	0.1
Rice	+	+	+	+	+	+	+
Wheat	0.1	0.1	0.1	0.1	0.1	0.1	0.1

Barley	+	+	+	+	+	+	+
Oats	+	+	+	+	+	+	+
Other Small Grains	+	+	+	+	+	+	+
Sorghum	+	+	+	+	+	+	+
Cotton	+	+	+	+	+	+	+
Grass Hay	+	+	+	+	+	+	+
Legume Hay	+	+	+	+	+	+	+
Peas	+	+	+	+	+	+	+
Sunflower	+	+	+	+	+	+	+
Tobacco	+	+	+	+	+	+	+
Vegetables	+	+	+	+	+	+	+
Chickpeas	+	+	+	+	+	+	+
Dry Beans	+	+	+	+	+	+	+
Lentils	+	+	+	+	+	+	+
Peanuts	+	+	+	+	+	+	+
Soybeans	+	+	+	+	+	+	+
Potatoes	+	+	+	+	+	+	+
Sugarbeets	+	+	+	+	+	+	+
Total	0.5	0.6	0.6	0.6	0.6	0.6	0.6

+ Does not exceed 0.05 MMT CO₂ Eq.

Note: Totals may not sum due to independent rounding.

1

2 **Table 5-30: CH₄, N₂O, CO, and NO_x Emissions from Field Burning of Agricultural Residues**
3 **(kt)**

Gas/Crop Type	1990	2005	2015	2016	2017	2018	2019
CH₄	15	17	18	17	17	17	17
Maize	2	4	5	5	5	5	5
Rice	3	3	3	2	3	2	3
Wheat	6	6	5	5	5	5	5
Barley	+	+	+	+	+	+	+
Oats	+	+	+	+	+	+	+
Other Small Grains	+	+	+	+	+	+	+
Sorghum	+	+	+	+	+	+	+
Cotton	1	2	1	1	1	1	1
Grass Hay	+	+	+	+	+	+	+
Legume Hay	+	+	+	+	+	+	+
Peas	+	+	+	+	+	+	+
Sunflower	+	+	+	+	+	+	+
Tobacco	+	+	+	+	+	+	+
Vegetables	+	+	+	+	+	+	+
Chickpeas	+	+	+	+	+	+	+
Dry Beans	+	+	+	+	+	+	+
Lentils	+	+	+	+	+	+	+
Peanuts	+	+	+	+	+	+	+
Soybeans	1	2	2	2	2	2	2
Potatoes	+	+	+	+	+	+	+
Sugarbeets	+	+	+	+	+	+	+
N₂O	1						

Maize	+		+		+	+	+	+	+
Rice	+		+		+	+	+	+	+
Wheat	+		+		+	+	+	+	+
Barley	+		+		+	+	+	+	+
Oats	+		+		+	+	+	+	+
Other Small Grains	+		+		+	+	+	+	+
Sorghum	+		+		+	+	+	+	+
Cotton	+		+		+	+	+	+	+
Grass Hay	+		+		+	+	+	+	+
Legume Hay	+		+		+	+	+	+	+
Peas	+		+		+	+	+	+	+
Sunflower	+		+		+	+	+	+	+
Tobacco	+		+		+	+	+	+	+
Vegetables	+		+		+	+	+	+	+
Chickpeas	+		+		+	+	+	+	+
Dry Beans	+		+		+	+	+	+	+
Lentils	+		+		+	+	+	+	+
Peanuts	+		+		+	+	+	+	+
Soybeans	+		+		+	+	+	+	+
Potatoes	+		+		+	+	+	+	+
Sugarbeets	+		+		+	+	+	+	+
CO	315		363		342	340	339	338	337
NO_x	13		15		14	14	14	14	14

+ Does not exceed 0.5 kt.

Note: Totals may not sum due to independent rounding.

1 Methodology

2 A country-specific Tier 2 method is used to estimate greenhouse gas emissions from field burning of agricultural
3 residues from 1990 to 2014 (for more details comparing the country-specific approach to the IPCC (2006) default
4 approach, see Box 5-6), and a data splicing method with a linear extrapolation is applied to complete the emissions
5 time series from 2015 to 2019. The following equation is used to estimate the amounts of C and N released
6 (R_i , where i is C or N) from burning.

7

8

$$R_i = CP \times RCR \times DMF \times F_i \times FB \times CE$$

9

10

$$FB = \frac{AB}{CAH}$$

11 where,

- 12 Crop Production (CP) = Annual production of crop, by state, kt crop production
13 Residue: Crop Ratio (RCR) = Amount of residue produced per unit of crop production, kt residue/kt crop
14 production
15 Dry Matter Fraction (DMF) = Amount of dry matter per unit of residue biomass for a crop, kt residue dry
16 matter/ kt residue biomass
17 Fraction C or N (F_i) = Fraction of C or N per unit of dry matter for a crop, kt C or N /kt residue dry
18 matter
19 Fraction Burned (FB) = Proportion of residue biomass consumed, unitless

- 1 Combustion Efficiency (CE) = Proportion of C or N released with respect to the total amount of C or N
 2 available in the burned material, respectively, unitless
 3 Area Burned (AB) = Total area of crop burned, by state, ha
 4 Crop Area Harvested (CAH) = Total area of crop harvested, by state, ha
 5

6 Crop production data are available by state and year from USDA (2019) for twenty-one crops that are burned in
 7 the conterminous United States, including maize, rice, wheat, barley, oats, other small grains, sorghum, cotton,
 8 grass hay, legume hay, peas, sunflower, tobacco, vegetables, chickpeas, dry beans, lentils, peanuts, soybeans,
 9 potatoes, and sugarbeets.²⁷ Crop area data are based on the 2015 National Resources Inventory (NRI) (USDA-NRCS
 10 2018). In order to estimate total crop production, the crop yield data from USDA Quick Stats crop yields is
 11 multiplied by the NRI crop areas. The production data for the crop types are presented in Table 5-31. Alaska and
 12 Hawaii are not included in the current analysis, but there is a planned improvement to estimate residue burning
 13 emissions for these two states in a future Inventory.

14 The amount of elemental C or N released through oxidation of the crop residues is used in the following equation
 15 to estimate the amount of CH₄, CO, N₂O, and NO_x emissions (E_g , where g is the specific gas, i.e., CH₄, CO, N₂O, and
 16 NO_x) from the Field Burning of Agricultural Residues:

$$17 \quad E_g = R_i \times EF_g \times CF$$

18 where,

- 19 Emission ratio (EF_g) = emission ratio by gas, g CH₄-C or CO-C/g C released, or g N₂O-N or NO_x-
 20 N/g N released
 21 Conversion Factor (CF) = conversion by molecular weight ratio of CH₄-C to C (16/12), CO-C to C
 22 (28/12), N₂O-N to N (44/28), or NO_x-N to N (30/14)
 23

24 **Box 5-6: Comparison of Tier 2 U.S. Inventory Approach and IPCC (2006) Default Approach**

Emissions from Field Burning of Agricultural Residues are calculated using a Tier 2 methodology that is based on the method developed by the IPCC/UNEP/OECD/IEA (1997). The rationale for using the IPCC/UNEP/OECD/IEA (1997) approach rather than the method provided in the 2006 IPCC Guidelines is as follows: (1) the equations from both guidelines rely on the same underlying variables (though the formats differ); (2) the IPCC (2006) equation was developed to be broadly applicable to all types of biomass burning, and, thus, is not specific to agricultural residues; (3) the IPCC (2006) method provides emission factors based on the dry matter content rather emission rates related to the amount of C and N in the residues; and (4) the IPCC (2006) default factors are provided only for four crops (corn, rice, sugarcane, and wheat) while this Inventory includes emissions from twenty-one crops.

A comparison of the methods in the current Inventory and the default IPCC (2006) approach was undertaken for 2014 to determine the difference in estimates between the two approaches. To estimate greenhouse gas emissions from field burning of agricultural residues using the IPCC (2006) methodology, the following equation—cf. IPCC (2006) Equation 2.27—was used with default factors and country-specific values for mass of fuel.

$$Emissions (kt) = AB \times (M_B \times C_f) \times G_{ef} \times 10^{-6}$$

where,

- Area Burned (AB) = Total area of crop burned (ha)

²⁷ Sugarcane and Kentucky bluegrass (produced on farms for turf grass installations) may have small areas of burning that are not captured in the sample of locations that were used in the remote sensing analysis.

Mass of Fuel ($M_B \times C_f$) = IPCC (2006) default carbon fractions with fuel biomass consumption U.S.- Specific Values using NASS Statistics²⁸ (metric tons dry matter burnt ha⁻¹)
Emission Factor (G_{ef}) = IPCC (2006) emission factor (g kg⁻¹ dry matter burnt)

The IPCC (2006) Tier 1 method approach resulted in 33 percent lower emissions of CH₄ and 53 percent lower emissions of N₂O compared to this Inventory. In summary, the IPCC/UNEP/OECD/IEA (1997) method is considered more appropriate for U.S. conditions because it is more flexible for incorporating country-specific data. Emissions are estimated based on specific C and N content of the fuel, which is converted into CH₄, CO, N₂O and NO_x, compared to IPCC (2006) approach that is based on dry matter rather than elemental composition.

1

2 **Table 5-31: Agricultural Crop Production (kt of Product)**

Crop	1990	2005	2013	2014
Maize	296,065	371,256	436,565	453,524
Rice	9,543	11,751	10,894	12,380
Wheat	79,805	68,077	67,388	62,602
Barley	9,281	5,161	4,931	5,020
Oats	5,969	2,646	1,806	2,042
Other Small Grains	2,651	2,051	1,902	2,492
Sorghum	23,687	14,382	18,680	18,436
Cotton	4,605	6,106	3,982	4,396
Grass Hay	44,150	49,880	45,588	46,852
Legume Hay	90,360	91,819	79,669	82,844
Peas	51	660	599	447
Sunflower	1,015	1,448	987	907
Tobacco	1,154	337	481	542
Vegetables	0	1,187	1,844	2,107
Chickpeas	0	5	0	0
Dry Beans	467	1,143	1,110	1,087
Lentils	0	101	72	76
Peanuts	1,856	2,176	2,072	2,735
Soybeans	56,612	86,980	94,756	110,560
Potatoes	18,924	20,026	20,234	19,175
Sugarbeets	24,951	25,635	31,890	31,737

Note: The amount of crop production has not been compiled for 2015 to 2019 so a data splicing method is used to estimate emissions for this portion of the time series.

3 The area burned is determined based on an analysis of remote sensing products (McCarty et al. 2009, 2010, 2011).
4 The presence of fires has been analyzed at 3600 survey locations in the NRI from 1990 to 2002 with LANDFIRE data
5 products developed from 30 m Landsat imagery (LANDFIRE 2014), and from 2003 through 2014 using 1 km
6 Moderate Resolution Imaging Spectroradiometer imagery (MODIS) Global Fire Location Product (MCD14ML) using
7 combined observations from Terra and Aqua satellites (Giglio et al. 2006). A sample of states are included in the
8 analysis with high, medium and low burning rates for agricultural residues, including Arkansas, California, Florida,
9 Indiana, Iowa and Washington. The area burned is determined directly from the analysis for these states.

²⁸ NASS yields are used to derive mass of fuel values because IPCC (2006) only provides default values for 4 of the 21 crops included in the Inventory.

1 For other states within the conterminous United States, the area burned for the 1990 through 2014 portion of the
 2 time series is estimated from a logistical regression model that has been developed from the data collected from
 3 the remote sensing products for the six states. The logistical regression model is used to predict occurrence of fire
 4 events. Several variables are tested in the logistical regression including a) the historical level of burning in each
 5 state (high, medium or low levels of burning) based on an analysis by McCarty et al. (2011), b) year that state laws
 6 limit burning of fields, in addition to c) mean annual precipitation and mean annual temperature from a 4-
 7 kilometer gridded product from the PRISM Climate Group (2015). A K-fold model fitting procedure is used due to
 8 low frequency of burning and likelihood that outliers could influence the model fit. Specifically, the model is
 9 trained with a random selection of sample locations and evaluated with the remaining sample. This process is
 10 repeated ten times to select a model that is most common among the set of ten, and avoid models that appear to
 11 be influenced by outliers due to the random draw of survey locations for training the model. In order to address
 12 uncertainty, a Monte Carlo analysis is used to sample the parameter estimates for the logistical regression model
 13 and produce one thousand estimates of burning for each crop in the remaining forty-two states included in this
 14 Inventory. State-level area burned data are divided by state-level crop area data to estimate the percent of crop
 15 area burned by crop type for each state. Table 5-32 shows the resulting percentage of crop residue burned at the
 16 national scale by crop type. State-level estimates are also available upon request.

17 **Table 5-32: U.S. Average Percent Crop Area Burned by Crop (Percent)**

Crop	1990	2005	2013	2014
Maize	0	0	0	0
Rice	8%	8%	4%	6%
Wheat	1%	2%	2%	1%
Barley	1%	0	1%	1%
Oats	1%	1%	2%	1%
Other Small Grains	1%	1%	1%	1%
Sorghum	1%	1%	1%	1%
Cotton	1%	1%	1%	1%
Grass Hay	0	0	0	0
Legume Hay	0	0	0	0
Peas	0	0	0	0
Sunflower	0	0	0	0
Tobacco	2%	2%	3%	3%
Vegetables	0	0	0	0
Chickpeas	0	1%	0	0
Dry Beans	1%	1%	0	0
Lentils	0	0	0	0
Peanuts	3%	3%	3%	3%
Soybeans	0	0	1%	1%
Potatoes	0	0	0	0
Sugarbeets	0	0	0	0

18 Additional parameters are needed to estimate the amount of burning, including residue: crop ratios, dry matter
 19 fractions, carbon fractions, nitrogen fractions and combustion efficiency. Residue: crop product mass ratios,
 20 residue dry matter fractions, and the residue N contents are obtained from several sources (IPCC 2006 and sources
 21 at bottom of Table 5-33). The residue C contents for all crops are based on IPCC (2006) default value for
 22 herbaceous biomass. The combustion efficiency is assumed to be 90 percent for all crop types
 23 (IPCC/UNEP/OECD/IEA 1997). See Table 5-33 for a summary of the crop-specific conversion factors. Emission ratios
 24 and mole ratio conversion factors for all gases are based on the *Revised 1996 IPCC Guidelines*
 25 (IPCC/UNEP/OECD/IEA 1997) (see Table 5-34).

1 **Table 5-33: Parameters for Estimating Emissions from Field Burning of Agricultural Residues**

Crop	Residue/Crop Ratio	Dry Matter Fraction	Carbon Fraction	Nitrogen Fraction	Combustion Efficiency (Fraction)
Maize	0.707	0.56	0.47	0.01	0.90
Rice	1.340	0.89	0.47	0.01	0.90
Wheat	1.725	0.89	0.47	0.01	0.90
Barley	1.181	0.89	0.47	0.01	0.90
Oats	1.374	0.89	0.47	0.01	0.90
Other Small Grains	1.777	0.88	0.47	0.01	0.90
Sorghum	0.780	0.60	0.47	0.01	0.90
Cotton	7.443	0.93	0.47	0.01	0.90
Grass Hay	0.208	0.90	0.47	0.02	0.90
Legume Hay	0.290	0.67	0.47	0.01	0.90
Peas	1.677	0.91	0.47	0.01	0.90
Sunflower	1.765	0.88	0.47	0.01	0.90
Tobacco	0.300	0.87	0.47	0.01	0.90
Vegetables	0.708	0.08	0.47	0.01	0.90
Chickpeas	1.588	0.91	0.47	0.01	0.90
Dry Beans	0.771	0.90	0.47	0.01	0.90
Lentils	1.837	0.91	0.47	0.02	0.90
Peanuts	1.600	0.94	0.47	0.02	0.90
Soybeans	1.500	0.91	0.47	0.01	0.90
Potatoes	0.379	0.25	0.47	0.02	0.90
Sugarbeets	0.196	0.22	0.47	0.02	0.90

Notes:

Chickpeas: IPCC (2006), Table 11.2; values are for Beans & pulses.

Cotton: Combined sources (Heitholt et al. 1992; Halevy 1976; Wells and Meredith 1984; Sadras and Wilson 1997; Pettigrew and Meredith 1997; Torbert and Reeves 1994; Gerik et al. 1996; Brouder and Cassmen 1990; Fritschi et al. 2003; Pettigrew et al. 2005; Bouquet and Breitenbeck 2000; Mahroni and Aharonov 1964; Bange and Milroy 2004; Hollifield et al. 2000; Mondino et al. 2004; Wallach et al. 1978).

Lentils: IPCC (2006), Table 11.2; Beans & pulses.

Peas: IPCC (2006), Table 11.2; values are for Beans & pulses.

Peanuts: IPCC (2006); Table 11.2; Root ratio and belowground N content values are for Root crops, other.

Sugarbeets: IPCC (2006); Table 11.2; values are for Tubers.

Sunflower: IPCC (2006), Table 11.2; values are for Grains.

Sugarcane: combined sources (Wiedenfels 2000, Dua and Sharma 1976; Singels & Bezuidenhout 2002; Stirling et al. 1999; Sitompul et al. 2000).

Tobacco: combined sources (Beyaert 1996; Moustakas and Ntzanis 2005; Crafts-Brandner et al. 1994; Hopkinson 1967; Crafts-Brandner et al. 1987).

Vegetables (Combination of carrots, lettuce/cabbage, melons, onions, peppers and tomatoes):

Carrots: McPharlin et al. (1992); Gibberd et al. (2003); Reid and English (2000); Peach et al. (2000); see IPCC Tubers for R:S and N fraction.

Lettuce, cabbage: combined sources (Huett and Dettman 1991; De Pinheiro Henriques & Marcelis 2000; Huett and Dettman 1989; Peach et al. 2000; Kage et al. 2003; Tan et al. 1999; Kumar et al. 1994; MacLeod et al. 1971; Jacobs et al. 2004; Jacobs et al. 2001; Jacobs et al. 2002); values from IPCC Grains used for N fraction.

Melons: Valantin et al. (1999); squash for R:S; IPCC Grains for N fraction.

Onion: Peach et al. (2000), Halvorson et al. (2002); IPCC (2006) Tubers for N fraction.

Peppers: combined sources (Costa and Gianquinto 2002; Marcussi et al. 2004; Tadesse et al. 1999; Diaz-Perez et al. 2008); IPCC Grains for N fraction.

Tomatoes: Scholberg et al. (2000a,b); Akindoye et al. (2005); values for AGR-N and BGR-N are from Grains.

1 **Table 5-34: Greenhouse Gas Emission Ratios and Conversion Factors**

Gas	Emission Ratio	Conversion Factor
CH ₄ :C	0.005 ^a	16/12
CO:C	0.060 ^a	28/12
N ₂ O:N	0.007 ^b	44/28
NO _x :N	0.121 ^b	30/14

^a Mass of C compound released (units of C) relative to mass of total C released from burning (units of C).

^b Mass of N compound released (units of N) relative to mass of total N released from burning (units of N).

2 For this Inventory, new activity data on the burned areas have not been analyzed for 2015 to 2019. To complete
 3 the emissions time series, a linear extrapolation of the trend is applied to estimate the emissions in the last five
 4 years of the inventory. Specifically, a linear regression model with autoregressive moving-average (ARMA) errors is
 5 used to estimate the trend in emissions over time from 1990 through 2014, and the trend is used to approximate
 6 the CH₄, N₂O, CO and NO_x for the last five years in the time series from 2015 to 2019 (Brockwell and Davis 2016).
 7 The Tier 2 method described previously will be applied to recalculate the emissions for the last five years in the
 8 time series (2015 to 2019) in a future Inventory.

9 **Uncertainty and Time-Series Consistency**

10 Emissions are estimated using a linear regression model with autoregressive moving-average (ARMA) errors for
 11 2019. The linear regression ARMA model produced estimates of the upper and lower bounds to quantify
 12 uncertainty (Table 5-35), and the results are summarized in Table 5-35. Methane emissions from field burning of
 13 agricultural residues in 2019 are between 0.35 and 0.50 MMT CO₂ Eq. at a 95 percent confidence level. This
 14 indicates a range of 18 percent below and 18 percent above the 2019 emission estimate of 0.43 MMT CO₂ Eq.
 15 Nitrous oxide emissions are between 0.16 and 0.22 MMT CO₂ Eq., or approximately 17 percent below and 17
 16 percent above the 2019 emission estimate of 0.19 MMT CO₂ Eq.

17 **Table 5-35: Approach 2 Quantitative Uncertainty Estimates for CH₄ and N₂O Emissions from**
 18 **Field Burning of Agricultural Residues (MMT CO₂ Eq. and Percent)**

Source	Gas	2019 Emission Estimate (MMT CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a (MMT CO ₂ Eq.) (%)			
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Field Burning of Agricultural Residues	CH ₄	0.4	0.35	0.50	-18%	18%
Field Burning of Agricultural Residues	N ₂ O	0.2	0.16	0.22	-17%	17%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

19 Due to data limitations, there are additional uncertainties in agricultural residue burning, particularly the potential
 20 omission of burning associated with Kentucky bluegrass (produced on farms for turf grass installation) and
 21 sugarcane.

22 Methodological recalculations were applied to the entire time series to ensure time-series consistency from 1990
 23 through 2019. Details on the emission trends and methodologies through time are described in the Introduction
 24 and Methodology sections.

1 QA/QC and Verification

2 A source-specific QA/QC plan for field burning of agricultural residues is implemented with Tier 1 analyses,
3 consistent with the U.S. Inventory QA/QC plan outlined in Annex 8. The previous Inventory included a term for
4 burning efficiency that is not found in the IPCC/UNEP/OECD/IEA (1997) method. This term has been removed
5 based on a QA/QC initiated by the UN Expert Review Team. In addition, the combustion efficiency term has been
6 set to 90 percent to be consistent with the Tier 1 method in IPCC/UNEP/OECD/IEA (1997).

7 Recalculations Discussion

8 Methodological recalculations are associated with two methodological revisions, a) removing the burning
9 efficiency term and b) adopting the combustion efficiency value in IPCC/UNEP/OECD/IEA (1997) (See QA/QC and
10 Verification Section for more information). As a result of these two revisions, the emissions increased on average
11 across the time series by 10 percent and 9 percent for CH₄ and N₂O, respectively. The absolute increases in
12 emissions are 0.4 MMT CO₂ Eq. and 0.2 MMT CO₂ Eq. for CH₄ and N₂O, respectively.

13 Planned Improvements

14 The key planned improvement is to estimate the emissions associated with field burning of agricultural residues in
15 the states of Alaska and Hawaii. In addition, a new method is in development that will directly link agricultural
16 residue burning with the Tier 3 methods that are used in several other source categories, including Agricultural Soil
17 Management, *Cropland Remaining Cropland*, and *Land Converted to Cropland* chapters of the Inventory. The
18 method is based on simulating burning events directly within the DayCent process-based model framework using
19 information derived from remote sensing fire products as described in the Methodology section. This
20 improvement will lead to greater consistency in the methods for across sources, ensuring mass balance of C and N
21 in the Inventory analysis.