Andrew Hawley

Puget Soundkeeper Alliance, Friends of Toppenish Creek, Sierra Club, Waterkeeper Alliance, Center for Food Safety, and Western Environmental Law Center submit these preliminary comments on the reissuance of the Washington State Concentrated Animal Feeding Operation Permits.

Puget Soundkeeper Alliance • Friends of Toppenish Creek • Sierra Club • Waterkeeper Alliance • Center for Food Safety • Western Environmental Law Center

October 24, 2021

Via Electronic Mail and Form Submission only

Chelsea Morris
Permit Writer
Washington State Department of Ecology
Via email to: chelsea.morris@ecy.wa.gov
And submitted online at: http://wq.ecology.commentinput.com/?id=AmHth

Dear Ms Morris:

Puget Soundkeeper Alliance, Friends of Toppenish Creek, Sierra Club, Waterkeeper Alliance, Center for Food Safety, and Western Environmental Law Center, and their tens of thousands of members, supporters, and volunteers throughout the State of Washington, submit these preliminary comments on the reissuance of the Washington State Concentrated Animal Feeding Operation (CAFO) Permits.

We are committed to protecting communities across Washington State from the impacts caused by CAFOs that are currently allowed to pollute surface waters and groundwater. It is past time for Ecology to do its job to restore these waters. Until the agency has done so, it will continue to deny Washingtonians the fundamental right to a safe and healthy environment, clean drinking water, and swimmable, fishable waterways.

Further, we are dedicated to recognizing and working to remedy the historic and ongoing structural racism that has led to cumulative environmental burdens, from toxic drinking water to threats from climate change, as well as the resulting disproportionate health outcomes experienced by people of color, Indigenous people, Tribal members and other marginalized communities across the state. To make real progress toward environmental justice, Ecology must take this opportunity to curb the discharges of pollutants from CAFOs as required under the law.

It is Ecology's job to ensure that safeguards are in place to protect our waters from constant pollution threats. Yet, for almost a decade Ecology has ignored our repeated calls to improve the oversight and regulation of the state's CAFOs. The urgency of our calls arises directly out of the significant, ongoing public health and environmental impacts from these facilities.

Throughout this time, we have made clear that Ecology's CAFO Permits fall short of federal and state regulatory requirements to protect public health and the environment from these impacts. By failing to require the use of modern pollution control measures, protect water quality, impose sufficient monitoring requirement, allow for meaningful public participation, cover all applicable facilities, protect overburdened communities, and account for a changing climate,

Ecology's failure to comply with federal and state law to properly regulate CAFOs places communities at risk.

We believe sustainable agriculture is a vital part of Washington's economy and way of life. We also believe farmers can be good stewards of our water resources if they follow responsible practices for reducing pollution from manure and fertilizer. That said, no one, including large-scale CAFOs should get a pass on polluting our waterways. We all have a responsibility to keep our waters clean for people, and wildlife such as shellfish, orcas, and salmon.

CAFOs Are Harming Our Communities

For far too long, Ecology has failed to require CAFOs to implement basic scientifically-proven and affordable best practices to prevent damaging water pollution from CAFOs statewide. These requirements, like those for other industry operations discharging waste to public waters, protect the environment and public health from dangerous pollutants. As a result of Ecology's failure, the direct and indirect impacts of CAFOs on the environment are making our communities unhealthy, unsafe, and less prosperous.

These avoidable impacts are all too predictable. For example, nitrate contamination threatens drinking water in communities with high concentrations of CAFOs. Ecology and the United States Geological Survey report that 29 percent of sampled wells in the Sumas Blaine aquifer in Whatcom County and over 20 percent of wells in the Yakima Valley exceed the nitrate maximum contaminant level. Nitrates are difficult for residents to detect because they are odorless, colorless, and flavorless, and they can cause multiple adverse health outcomes such as methemoglobinemia ("blue baby syndrome"), cardiovascular harm, strokes, reproductive problems such as miscarriages, thyroid problems, and some cancers.[1] Boiling water just makes the problem worse, and for many environmentally overburdened communities, such as those in Yakima County, the costs of remedial measures such as filtration or bottled water are too high, forcing Washington residents—disproportionately Indigenous and people of color—to sacrifice health for private profit.

In addition to the impact on drinking water, the discharge of pollutants from CAFOs significantly impacts the water quality of the state's rivers, streams, and marine waters. For example, the discharge of nutrients and toxic pollution from facilities such as CAFOs into Puget Sound and its tributaries is creating a water quality crisis. Perhaps the most immediate and pressing problem with the Sound's water quality is dangerously low levels of dissolved oxygen caused by excessive nutrients from over-application of manure and fertilizers. As Ecology itself

---

[1] *See, e.g.,* Environmental Working Group, America's Nitrate Habit Is Costly and Dangerous. Prevention Is the Solution, But Voluntary Actions Fall Short, Oct. 2, 2018 (availabele at https://www.ewg.org/research/nitratecost/); NRDC, CAFOs: What We Don't Know Is Hurting Us, Sept. 2019 (available at https://www.nrdc.org/sites/default/files/cafos-dont-know-hurting-us-report.pdf); National Cancer Institute, Cancer Trends Progress Report, Nitrates (available at https://progressreport.cancer.gov/prevention/nitrate); Agency for Toxic Substances and Disease Registry, Public Health Statement for Nitrate and Nitrite (available at https://www.atsdr.cdc.gov/ToxProfiles/tp204-c1.pdf).

stated a dozen years ago, "[f]ish need oxygen" yet "[t]here are many areas in Puget Sound with very low levels of dissolved oxygen."[2]

Ecology itself reports that the presence of excess nutrients in the water—i.e., nitrogen and phosphorus—is causing dissolved oxygen levels to drop to these critically low levels in some parts of Puget Sound.[3] Ecology knows that low oxygen levels in Puget Sound are "bad news for aquatic life" such as shellfish, salmon, Southern Resident orcas and other species.[4] Yet Ecology's approach to CAFO management ignores the clear connection between ongoing pollution from these operations, the Sound's failure to meet water quality standards, and the threats to these species.

In addition to these direct threats, CAFOs are a significant contributor to the climate crisis. Dairies, especially those that confine cows and use manure lagoons, drive climate change by emitting greenhouse gases such as nitrous oxide and methane. These pollutants are less abundant than the well-known greenhouse gas, carbon dioxide (CO_2), but they are much more potent: methane has a Global Warming Potential (GWP) 84-87 times that of CO_2 over 20 years, and nitrous oxide has a GWP of 265–298 times that of CO_2 over 100 years.[5] Livestock production is the dominant source of methane in the United States,[6] and manure management is the fastest growing major source of methane, with emissions from dairies increasing by 117 percent between 1990 and 2019.[7]

Washington is already experiencing the catastrophic effects of climate change through dwindling snowpack and freshwater resources, unprecedented and deadly heatwaves, and increased wildfire. Those most vulnerable to climate change are people of color, Indigenous people, Tribes, and others subject to disproportionate impacts from historic and ongoing systemic and structural racism. Importantly, impacts to water quality caused by and exacerbated by a changing climate, such as increased temperature, lower dissolved oxygen, and nuisance algal growth, are the same impacts caused by discharges from CAFOs. As a result, CAFO pollution both causes water pollution and makes it significantly worse by driving the climate crisis.

---

[2] *See Ecology, Public Notice South Puget Sound Dissolved Oxygen Study (2006).*

[3] *See* Ecology, Puget Sound and the Straits Dissolved Oxygen Assessment Impacts of Current and Future Human Nitrogen Sources and Climate Change through 2070, at 98–101 (2014).

[4] *See* Ecology, Nutrient pollution modeling shows different futures for Puget Sound, Blog, https://ecology.wa.gov/Blog/Posts/September-2021/Latest-Salish-Sea-modeling-results-bring-us-closer (last visited Oct. 20, 2021).

[5] https://www.epa.gov/ghgemissions/understanding-global-warming-potentials (last visited Oct. 24, 2021).

[6] https://www.epa.gov/ghgemissions/overview-greenhouse-gases (last visited Oct. 24. 2021).

[7] Environmental Protection Agency, Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2019, 5-12 (April 14, 2021).

The Washington Court of Appeals Ordered Ecology to Remedy Failures to Comply with the Law

It was for these reasons, and many others, that several organizations challenged the previous iterations of Ecology's general permits for CAFOs. The organizations challenged Ecology's repeated failures to follow state and federal law regarding the minimum steps CAFOs must take to prevent the release of pollutants and protect water quality, and Ecology's failure to impose adequate monitoring requirements, allow for public oversight, and address climate change.

In June of 2021, the Washington State Court of Appeals agreed and invalidated Ecology's permit because it failed to comply with the law in several important ways. First, the court held that Ecology did not follow the state statute requiring a determination of what modern pollution controls were reasonable to control the discharge of nutrients, bacteria, and other pollutants before issuing the permit. Second, the court found that the permit did not limit the discharge of pollutants adequately to protect the health of nearby waterways, as required under state and federal law. Third, the court found the permit did not include sufficient monitoring of both surface waters and groundwater to determine both whether the permit was working and whether the permittees were complying with their obligations. Fourth, Ecology failed to require site-specific Nutrient Management Plans that meet federal standards as required to ensure meaningful evaluation of, and public participation in, the development of the measures meant to protect local waterbodies and communities. Finally, the Court held that under state law Ecology must consider the impacts of climate change when developing the permit.

In reissuing the CAFO Permits, Ecology must, at a minimum, strengthen the Permit to redress the deficiencies identified by the Appellate Court in its opinion.[8] These include but are not limited to:

- Determine the most current methodology that can be reasonably required for preventing, controlling, or abating the pollutants associated with a discharge from each part of the facilities, including but not limited to, existing manure lagoons and compost areas.
- Impose effluent limits to ensure that each facility implements such adequate discharge controls for their facility based on its size and environmental impact.
- Impose the effluent limits necessary to ensure that each facility does not cause or contribute to a violation of state surface water and groundwater quality standards. Specifically, Ecology must develop water quality based effluent limitations that protect all waters of the state from production area and land application area discharges, including but not limited to discharges from lagoons, compost areas, land application fields, and tile drains.
- Protect groundwater from pollutant discharges and leaching from manure storage lagoons, compost areas, and land application fields.
- Require surface water monitoring that is sufficient to ensure the permit terms are adequate to comply with the requirements of state and federal law regarding the implementation of the required pollution control technology and the protection of water quality, and that the permittees are complying with the permit's terms and conditions.

---

[8] https://www.courts.wa.gov/opinions/pdf/D2%2052952-1-II%20Published%20Opinion.pdf

- Require groundwater monitoring that is sufficient to ensure the permit terms are adequate to comply with the requirements of state and federal law regarding the implementation of the required pollution control technology and the protection of water quality, and that the permittees are complying with the permit's terms and conditions.
- Require the development of a site-specific Nutrient Management Plan that demonstrates how the CAFO will comply with state law, federal law, and all of the permit terms, conduct agency review of the Nutrient Management Plan for compliance and make affirmative determinations of its adequacy, and provide the public access to review and comment on the site-specific Nutrient Management Plan and Ecology's review of that plan before plan approval and before issuance of any permit.
- Quantify the toll of the state's CAFO operations on the climate crisis and the impacts of climate change on water quality in the writing of the Permits.

Ecology Must Comply with State and Federal Law by Implementing a Protective Permit and by Requiring CAFOs Be Permitted

While these changes are essential, more must be done to fully address CAFO pollution. With this third iteration of the CAFO general permit, Ecology must move towards, at long last, eliminating once and for *all* the discharge of pollution from these facilities.

This is, of course, the true goal of both state and federal law. Indeed, the Washington Water Pollution Control Act declares the "public policy of the state of Washington to maintain the highest possible standards to insure the purity of all waters of the state consistent with public health and public enjoyment thereof, the propagation and protection of wildlife, birds, game, fish and other aquatic life, and the industrial development of the state." RCW 90.48.010. Thus, "[c]onsistent with this policy, the state of Washington will exercise its powers, as fully and as effectively as possible, to retain and secure high quality for all waters of the state [and] work[] cooperatively with the federal government in a joint effort *to extinguish the sources of water quality degradation*." *Id*. (emphasis added). The Clean Water Act, in turn, is designed "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters," 33 U.S.C. § 1251(a), with the goal of not just reducing, but eliminating, all water pollution. *Waterkeeper Alliance, Inc. v. U.S. E.P.A*, 399 F.3d 486, 490 (2d Cir. 2005) (citing 33 U.S.C. § 1251(a)(1)). The time has come for Ecology to put these goals into practice.

Moreover, and relatedly, Ecology must ensure universal coverage under the permits. Even the most protective general permit is of little use if all applicable facilities are not covered, which is the current situation. Ecology must use its authority and resources to ensure that all large, medium, or small CAFOs that are discharging to the state's waters are covered under the permit. Ecology's current approach of assuming a facility is not discharging is unsupported by the facts or science, and places communities at risk, particularly those already overburdened by environmental harms and unable to secure the basic protections afforded under the law.

Ecology Must Commit to an Open and Transparent Process for Developing the Permit

Ecology must engage the public in an open and transparent process as it works to make the necessary improvements to its permitting approach. Given the significant impacts these facilities have on surrounding communities and communities downstream of CAFOs, Ecology must actively solicit information and comments from these communities.

On this score, Ecology is not off to a particularly good start. We are concerned generally about what information Ecology is giving the public and the forums it is providing for impacted members of communities to learn about, and comment on, the CAFO permitting process. We are not reassured, at least so far, that Ecology is on a path to comply with the court's mandate that it provide the public with access during the permit writing process.

To begin with, while we appreciate Ecology's efforts in holding two listening sessions, it was apparent that the communities directly impacted by CAFOs did not feel that these sessions were a viable forum for discussing the changes that should be made to the permits. This could have been for several reasons, but the platform Ecology uses for virtual public hearings disadvantages the public because it does not provide information about who is commenting, nor does it allow participants to have a sense of the number of people in attendance. To ask people in the community to speak directly to a room with an unknown audience, where only staff members are visible, is likely to dissuade members of the communities most impacted by CAFOs from commenting.

Moreover, virtual listening sessions are only useful for those who know about the comment period, understand the process and the opportunity it presents, have access to technology to comment, and are available during the scheduled time to submit comments. Ecology has not done a particularly effective job at conveying this information nor has it provided the sort of community outreach and support necessary to ensure actual public access to this process.

Ecology must go further than it has to this point to ensure that those most impacted by CAFOs have the opportunity to be heard. Ecology must continue to use these traditional tools, such as webinars and passive comment periods, for receiving public input, but must also actively engage the community in a conversation regarding CAFOs. While this will undoubtedly be more difficult in light of the ongoing pandemic, Ecology owes it to those it is mandated to protect, the people of Washington, to make every effort to hear their story.

Conclusion

We appreciate your consideration of these preliminary comments. We look forward to working with you to develop permits that are based on science, comply with all applicable legal requirements, and that protect the communities who have been put directly in harm's way by ongoing, un- or under-regulated pollution from these facilities. Should you have any questions or concerns for Commenters, please contact Andrew Hawley at 206-487-7250, or hawley@westernlaw.org.

Sincerely,

Alyssa Barton
Policy Manager
Puget Soundkeeper Alliance

Jean Mendoza
Executive Director
Friends of Toppenish Creek

Emily Knobbe
Policy Manager
Center for Food Safety

Kelly Hunter Foster
Clean Water Defense Campaign Manager
Waterkeeper Alliance

Margie Van Cleve
Conservation Chair
Washington State Chapter of the Sierra Club

Andrew Hawley
Staff Attorney
Western Environmental Law Center

5. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. This chapter provides an assessment of methane (CH_4) and nitrous oxide (N_2O) 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 (CO_2) emissions from liming and urea fertilization (see Figure 5-1). Additional CO_2, CH_4 and N_2O fluxes from agriculture-related land-use and land-use conversion activities, such as cultivation of cropland, grassland fires, aquaculture, 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 CH_4 and N_2O emissions from stationary on-farm energy use are reported in the Energy chapter under the Industrial sector emissions. Methane and N_2O 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**
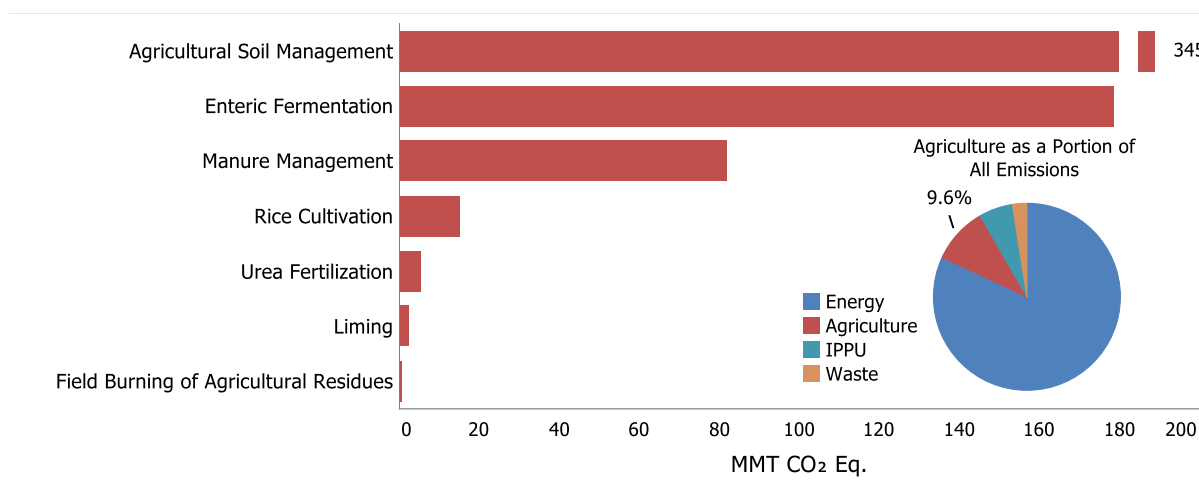
In 2019, the Agriculture sector was responsible for emissions of 628.6 MMT CO_2 Eq.,[1] or 9.6 percent of total U.S. greenhouse gas emissions. Methane emissions from enteric fermentation and manure management represent 27.1 percent and 9.5 percent of total CH_4 emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were the largest emitters of CH_4. Rice cultivation and field burning of agricultural residues were minor sources of CH_4. Emissions of N_2O 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. N_2O emissions, accounting for 75.4 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 CO_2 equivalent values based on the *IPCC Fourth Assessment Report* (AR4) GWP values. See the Introduction chapter for more information.

of agricultural residues were also small sources of N_2O emissions. Urea fertilization and liming accounted for 0.10 percent and 0.05 percent of total CO_2 emissions from anthropogenic activities, respectively.

Table 5- and Table 5-2 present emission estimates for the Agriculture sector. Between 1990 and 2019, CO_2 and CH_4 emissions from agricultural activities increased by 9.9 percent and 17.5 percent, respectively, while N_2O emissions from agricultural activities fluctuated from year to year, but increased by 10.4 percent overall. Trends in sources of agricultural emissions over the 1990 to 2019 time series are shown in Figure 5-2.

Figure 5-2: Trends in Agriculture Chapter Greenhouse Gas Emission Sources
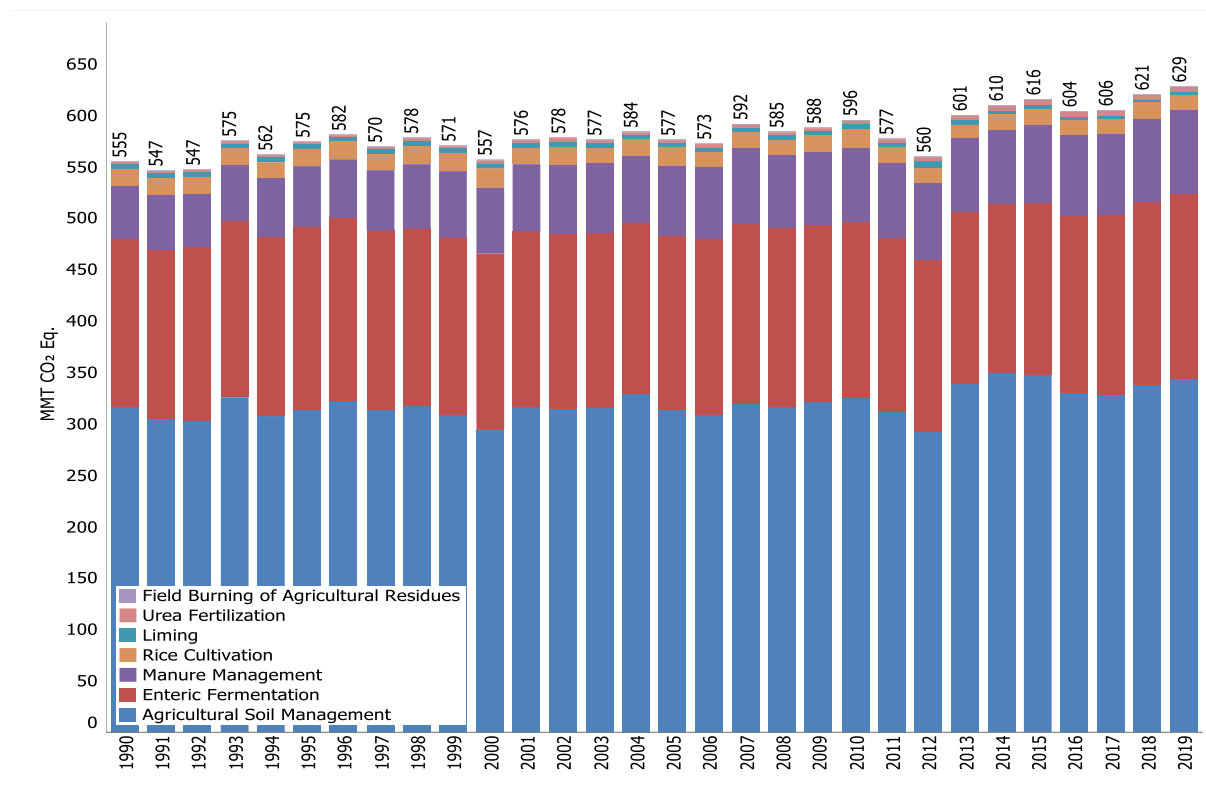
Each year, some emission estimates in the Agriculture sector of the Inventory are recalculated and revised with improved methods and/or data. In general, recalculations are made to the U.S. greenhouse gas emission estimates either to incorporate new methodologies or, most commonly, to update recent historical data. These improvements are implemented consistently across the previous Inventory's time series (i.e., 1990 through 2018) to ensure that the trend is accurate. This year's notable updates include (1) Enteric Fermentation: default national emission factors were updated for sheep and goats; (2) Field Burning of Agricultural Residues: updated parameters within the methodology for combustion efficiency; (3) Urea Fertilization: updated methodology based on the analytical solution from the Monte Carlo analysis; (4) Rice Cultivation: correction in splicing method; (5) Liming: updated activity data from USGS; and (6) Agricultural Soil Management: using surrogate date method to update the time series of PRP and manure N available for application to soils. In total, the improvements made to the Agriculture sector in this Inventory increased greenhouse gas emissions by 2.5 MMT CO_2 Eq. (0.4 percent) in 2018. For more information on specific methodological updates, please see the Recalculations discussions within the respective source category sections of this chapter.

Emissions reported in the Agriculture chapter include those from all states; however, for Hawaii and Alaska some agricultural practices that can increase nitrogen availability in the soil, and thus cause N_2O emissions, are not included (see chapter sections on "Uncertainty and Time-Series Consistency" and "Planned Improvements" for more details). In addition, U.S. Territories and the District of Columbia are not estimated with the exception of Urea Fertilization in Puerto Rico due to incomplete data. EPA continues to review available data on an ongoing basis to include emissions from territories in future inventories to the extent they are occurring. Many U.S.

territories in the Pacific Islands have no permanent populations and therefore EPA assumes no agriculture activities are occuring. See Annex 5 for more information on EPA's assessment of the sources not included in this inventory.

Table 5-1: Emissions from Agriculture (MMT CO_2 Eq.)

Gas/Source	1990	2005	2015	2016	2017	2018	2019
CO_2	**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
CH_4	**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
N_2O	**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.

Table 5-2: Emissions from Agriculture (kt)

Gas/Source	1990	2005	2015	2016	2017	2018	2019
CO_2	**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
CH_4	**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
N_2O	**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.

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 format 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

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 CH_4 as a byproduct, which can be exhaled or eructated by the animal. The amount of CH_4 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 CH_4 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 CH_4 emissions per unit of body mass among all animal types.

Non-ruminant animals (e.g., swine, horses, and mules and asses) also produce CH_4 emissions through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants emit significantly less CH_4 on a per-animal-mass basis than ruminants because the capacity of the large intestine to produce CH_4 is lower.

In addition to the type of digestive system, an animal's feed quality and feed intake also affect CH_4 emissions. In general, lower feed quality and/or higher feed intake leads to higher CH_4 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 CH_4 emissions in 2019 were 178.6 MMT CO_2 Eq. (7,142 kt). Beef cattle remain the largest contributor of CH_4 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: CH_4 Emissions from Enteric Fermentation (MMT CO_2 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

---

[2] CO_2 emissions from livestock are not estimated because annual net CO_2 emissions are assumed to be zero – the CO_2 photosynthesized by plants is returned to the atmosphere as respired CO_2 (IPCC 2006).

[3] Enteric fermentation emissions from poultry are not estimated because no IPCC method has been developed for determining enteric fermentation CH_4 emissions from poultry; at this time, developing of a country-specific method would require a disproportionate amount of resources given the small magnitude of this source category. Enteric fermentation emissions from camels are not estimated because there is no significant population of camels in the United States. Given the insignificance of estimated camel emissions in terms of the overall level and trend in national emissions, there are no immediate improvement plans to include this emissions category in the Inventory. See Annex 5 for more information on significance of estimated camel emissions.

Livestock Type	1990		2005		2015	2016	2017	2018	2019
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
Goats	0.6		0.7		0.6	0.6	0.6	0.6	0.6
American Bison	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 CO_2 Eq.

Table 5-4: CH_4 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
Goats	23		26		24	24	24	24	24
American Bison	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.

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

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

Methodology

Livestock enteric fermentation emission estimate methodologies fall into two categories: cattle and other domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics,

account for the majority of enteric fermentation CH_4 emissions from livestock in the United States. A more detailed methodology (i.e., IPCC Tier 2) was therefore applied to estimate emissions for all cattle. Emission estimates for other domesticated animals (horses, sheep, swine, goats, American bison, and mules and asses) were estimated using the IPCC Tier 1 approach, as suggested by the *2006 IPCC Guidelines* (see the Planned Improvements section).

While the large diversity of animal management practices cannot be precisely characterized and evaluated, significant scientific literature exists that provides the necessary data to estimate cattle emissions using the IPCC Tier 2 approach. The Cattle Enteric Fermentation Model (CEFM), developed by EPA and used to estimate cattle CH_4 emissions from enteric fermentation, incorporates this information and other analyses of livestock population, feeding practices, and production characteristics. For the current Inventory, CEFM results for 1990 through 2017 were carried over from the 1990 to 2017 Inventory (i.e., 2019 Inventory submission) to focus resources on CEFM improvements, and a simplified approach was used to estimate 2018 and 2019 enteric emissions from cattle.

See Annex 3.10 for more detailed information on the methodology and data used to calculate CH_4 emissions from enteric fermentation. In addition, variables and the resulting emissions are also available at the state level in Annex 3.10.

1990 to 2017 Inventory Methodology for Cattle

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

- Dairy Cattle

  o Calves
  o Heifer Replacements
  o Cows

- Beef Cattle

  o Calves
  o Heifer Replacements
  o Heifer and Steer Stockers
  o Animals in Feedlots (Heifers and Steer)
  o Cows
  o Bulls

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

Diet characteristics were estimated by region for dairy, grazing beef, and feedlot beef cattle. These diet characteristics were used to calculate digestible energy (DE) values (expressed as the percent of gross energy intake digested by the animal) and CH_4 conversion rates (Y_m) (expressed as the fraction of gross energy converted to CH_4) for each regional population category. The IPCC recommends Y_m ranges of 3.0±1.0 percent for feedlot cattle and 6.5±1.0 percent for other well-fed cattle consuming temperate-climate feed types (IPCC 2006). Given the availability of detailed diet information for different regions and animal types in the United States, DE and Y_m values unique to the United States were developed. The diet characterizations and estimation of DE and Y_m values were based on information from state agricultural extension specialists, a review of published forage quality studies and scientific literature, expert opinion, and modeling of animal physiology.

The diet characteristics for dairy cattle were based on Donovan (1999) and an extensive review of nearly 20 years of literature from 1990 through 2009. Estimates of DE were national averages based on the feed components of the diets observed in the literature for the following year groupings: 1990 through 1993, 1994 through 1998, 1999

through 2003, 2004 through 2006, 2007, and 2008 onward.[4] Base year Y_m values by region were estimated using Donovan (1999). As described in ERG (2016), a ruminant digestion model (COWPOLL, as selected in Kebreab et al. 2008) was used to evaluate Y_m for each diet evaluated from the literature, and a function was developed to adjust regional values over time based on the national trend. Dairy replacement heifer diet assumptions were based on the observed relationship in the literature between dairy cow and dairy heifer diet characteristics.

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

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

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

To estimate CH_4 emissions, the population was divided into state, age, sub-type (i.e., dairy cows and replacements, beef cows and replacements, heifer and steer stockers, heifers and steers in feedlots, bulls, beef calves 4 to 6 months, and dairy calves 4 to 6 months), and production (i.e., pregnant, lactating) groupings to more fully capture differences in CH_4 emissions from these animal types. The transition matrix was used to simulate the age and weight structure of each sub-type on a monthly basis in order to more accurately reflect the fluctuations that occur throughout the year. Cattle diet characteristics were then used in conjunction with Tier 2 equations from IPCC (2006) to produce CH_4 emission factors for the following cattle types: dairy cows, beef cows, dairy replacements, beef replacements, steer stockers, heifer stockers, steer feedlot animals, heifer feedlot animals, bulls, and calves. To estimate emissions from cattle, monthly population data from the transition matrix were multiplied by the calculated emission factor for each cattle type. More details are provided in Annex 3.10.

2018 and 2019 Inventory Methodology for Cattle

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

---

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

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

CEFM Cattle Category	USDA-NASS *QuickStats* 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

Non-Cattle Livestock

Emission estimates for other animal types were based on average emission factors (Tier 1 default IPCC emission factors) representative of entire populations of each animal type. Methane emissions from these animals accounted for a minor portion of total CH_4 emissions from livestock in the United States from 1990 through 2019. Additionally, the variability in emission factors for each of these other animal types (e.g., variability by age, production system, and feeding practice within each animal type) is less than that for cattle.

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

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

Uncertainty and Time-Series Consistency

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

A total of 185 primary input variables (177 for cattle and 8 for non-cattle) were identified as key input variables for the uncertainty analysis. A normal distribution was assumed for almost all activity- and emission factor-related input variables. Triangular distributions were assigned to three input variables (specifically, cow-birth ratios for the

three most recent years included in the 2001 model run) to ensure only positive values would be simulated. For some key input variables, the uncertainty ranges around their estimates (used for inventory estimation) were collected from published documents and other public sources; others were based on expert opinion and best estimates. In addition, both endogenous and exogenous correlations between selected primary input variables were modeled. The exogenous correlation coefficients between the probability distributions of selected activity-related variables were developed through expert judgment.

Among the individual cattle sub-source categories, beef cattle account for the largest amount of CH_4 emissions, as well as the largest degree of uncertainty in the emission estimates—due mainly to the difficulty in estimating the diet characteristics for grazing members of this animal group. Among non-cattle, horses represent the largest percent of uncertainty in the previous uncertainty analysis because the Food and Agricultural Organization of the United Nations (FAO) population estimates used for horses at that time had a higher degree of uncertainty than for the USDA population estimates used for swine, goats, and sheep. The horse populations are now from the same USDA source as the other animal types, and therefore the uncertainty range around horses is likely overestimated. Cattle calves, American bison, mules and asses were excluded from the initial uncertainty estimate because they were not included in emission estimates at that time.

The uncertainty ranges associated with the activity data-related input variables were plus or minus 10 percent or lower. However, for many emission factor-related input variables, the lower- and/or the upper-bound uncertainty estimates were over 20 percent. The results of the quantitative uncertainty analysis are summarized in Table 5-6. Based on this analysis, enteric fermentation CH_4 emissions in 2019 were estimated to be between 158.9 and 210.7 MMT CO_2 Eq. at a 95 percent confidence level, which indicates a range of 11 percent below to 18 percent above the 2019 emission estimate of 178.6 MMT CO_2 Eq.

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

Source	Gas	2019 Emission Estimate (MMT CO₂ Eq.)	Uncertainty Range Relative to Emission Estimate[a, b, c]			
			(MMT CO₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Enteric Fermentation	CH₄	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.

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

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

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

Recalculations Discussion

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

Planned Improvements

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

- Further research to improve the estimation of dry matter intake (as gross energy intake) using data from appropriate production systems;

- Updating input variables that are from older data sources, such as beef births by month, beef and dairy annual calving rates, and beef cow lactation rates;

- Investigating the availability of data for dairy births by month, to replace the current assumption that births are evenly distributed throughout the year;

- Updating the diet data to incorporate monthly or annual milk fat data in place of the fixed IPCC default value of 4 percent milk fat. EPA has investigated the availability of data across the time series and plans to incorporate annual U.S. milk fat values into the CEFM calculations in the next (i.e., 1990 to 2020) Inventory, as opposed to using a default 4 percent milk fat across the entire time series;

- Investigating the availability of annual data for the DE, Y_m, and crude protein values of specific diet and feed components for grazing and feedlot animals;

- Further investigation on additional sources or methodologies for estimating DE for dairy cattle, given the many challenges in characterizing dairy cattle diets;

- Further evaluation of the assumptions about weights and weight gains for beef cows, such that trends beyond 2007 are updated, rather than held constant;

- Further evaluation of the estimated weight for dairy cows (i.e., 1,500 lbs) that is based solely on Holstein cows as mature dairy cow weight is likely slightly overestimated, based on knowledge of the breeds of dairy cows in the United States.

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

- Potentially updating to a Tier 2 methodology for other animal types (i.e., sheep, swine, goats, horses); efforts to move to Tier 2 will consider the emissions significance of livestock types;

- Investigation of methodologies and emission factors for including enteric fermentation emission estimates from poultry;

- Comparison of the current CEFM processing of animal population data to estimates developed using annual average populations to determine if the model could be simplified to use annual population data;

- Comparison of the current CEFM with other models that estimate enteric fermentation emissions for quality assurance and verification;

- Investigation of recent research implications suggesting that certain parameters in enteric models may be simplified without significantly diminishing model accuracy;

- Recent changes that have been implemented to the CEFM warrant an assessment of the current uncertainty analysis; therefore, a revision of the quantitative uncertainty surrounding emission estimates from this source category will be initiated. EPA plans to perform this uncertainty analysis following the completed updates to the CEFM; and

- Analysis and integration of a more representative spatial distribution of animal populations by state, particularly for poultry animal populations.

EPA received comments during recent Public Review periods of the Inventory regarding the CEFM model and data and assumptions used to calculate enteric fermentation cattle emissions. Many of the comments received are consistent with potential planned improvement options listed above. EPA is continuously investigating these recommendations and potential improvements and working with USDA and other experts to utilize the best available data and methods for estimating emissions. Many of these improvements are major updates and may take multiple years to implement in full. In addition, EPA received comments during the Public Review period of the current (1990 through 2019) and previous (1990 through 2018) Inventory regarding the use of alternate metrics for weighting non-CO_2 emissions such as methane that differ from those required in reporting under the UNFCCC to facilitate comparability as described in Box 5-1.

5.2 Manure Management (CRF Source Category 3B)

The treatment, storage, and transportation of livestock manure can produce anthropogenic CH_4 and N_2O emissions.[5] Methane is produced by the anaerobic decomposition of manure and nitrous oxide is produced from direct and indirect pathways through the processes of nitrification and denitrification; in addition, there are many underlying factors that can affect these resulting emissions from manure management, as described below.

When livestock manure is stored or treated in systems that promote anaerobic conditions (e.g., as a liquid/slurry in lagoons, ponds, tanks, or pits), the decomposition of the volatile solids component in the manure tends to produce CH_4. When manure is handled as a solid (e.g., in stacks or drylots) or deposited on pasture, range, or paddock lands, it tends to decompose aerobically and produce CO_2 and little or no CH_4. Ambient temperature, moisture, and manure storage or residency time affect the amount of CH_4 produced because they influence the growth of the bacteria responsible for CH_4 formation. For non-liquid-based manure systems, moist conditions (which are a function of rainfall and humidity) can promote CH_4 production. Manure composition, which varies by animal diet, growth rate, and animal type (particularly the different animal digestive systems), also affects the amount of CH_4 produced. In general, the greater the energy content of the feed, the greater the potential for CH_4 emissions. However, some higher-energy feeds also are more digestible than lower quality forages, which can result in less overall waste excreted from the animal.

As previously stated, N_2O emissions are produced through both direct and indirect pathways. Direct N_2O emissions are produced as part of the nitrogen (N) cycle through the nitrification and denitrification of the N in livestock dung

---

[5] CO_2 emissions from livestock are not estimated because annual net CO_2 emissions are assumed to be zero – the CO_2 photosynthesized by plants is returned to the atmosphere as respired CO_2 (IPCC 2006).

and urine.[6] There are two pathways for indirect N_2O emissions. The first is the result of the volatilization of N in manure (as NH_3 and NO_x) and the subsequent deposition of these gases and their products (NH_4^+ and NO_3^-) onto soils and the surface of lakes and other waters. The second pathway is the runoff and leaching of N from manure into the groundwater below, into riparian zones receiving drain or runoff water, or into the ditches, streams, rivers, and estuaries into which the land drainage water eventually flows.

The production of direct N_2O emissions from livestock manure depends on the composition of the manure (manure includes both feces and urine), the type of bacteria involved in the process, and the amount of oxygen and liquid in the manure system. For direct N_2O emissions to occur, the manure must first be handled aerobically where organic N is mineralized or decomposed to NH_4 which is then nitrified to NO_3 (producing some N_2O as a byproduct) (nitrification). Next, the manure must be handled anaerobically where the nitrate is then denitrified to N_2O and N_2 (denitrification). NO_x can also be produced during denitrification (Groffman et al. 2000; Robertson and Groffman 2015). These emissions are most likely to occur in dry manure handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to saturation. A very small portion of the total N excreted is expected to convert to N_2O in the waste management system (WMS).

Indirect N_2O emissions are produced when nitrogen is lost from the system through volatilization (as NH_3 or NO_x) or through runoff and leaching. The vast majority of volatilization losses from these operations are NH_3. Although there are also some small losses of NO_x, there are no quantified estimates available for use, so losses due to volatilization are only based on NH_3 loss factors. Runoff losses would be expected from operations that house animals or store manure in a manner that is exposed to weather. Runoff losses are also specific to the type of animal housed on the operation due to differences in manure characteristics. Little information is known about leaching from manure management systems as most research focuses on leaching from land application systems. Since leaching losses are expected to be minimal, leaching losses are coupled with runoff losses and the runoff/leaching estimate provided in this chapter does not account for any leaching losses.

Estimates of CH_4 emissions from manure management in 2019 were 62.4 MMT CO_2 Eq. (2,495 kt); in 1990, emissions were 37.1 MMT CO_2 Eq. (1,485 kt). This represents a 68 percent increase in emissions from 1990. Emissions increased on average by 0.8 MMT CO_2 Eq. (2 percent) annually over this period. The majority of this increase is due to swine and dairy cow manure, where emissions increased 49 and 117 percent, respectively. From 2018 to 2019, there was a 1 percent increase in total CH_4 emissions from manure management, due to an increase in animal populations.

Although a large quantity of managed manure in the United States is handled as a solid, producing little CH_4, the general trend in manure management, particularly for dairy cattle and swine (which are both shifting towards larger facilities), is one of increasing use of liquid systems. Also, new regulations controlling the application of manure nutrients to land have shifted manure management practices at smaller dairies from daily spread systems to storage and management of the manure on site. In many cases, manure management systems with the most substantial methane emissions are those associated with confined animal management operations where manure is handled in liquid-based systems. Nitrous oxide emissions from manure management vary significantly between the types of management system used and can also result in indirect emissions due to other forms of nitrogen loss from the system (IPCC 2006).

While national dairy animal populations have decreased since 1990, some states have seen increases in their dairy cattle populations as the industry becomes more concentrated in certain areas of the country and the number of animals contained on each facility increases. These areas of concentration, such as California, New Mexico, and Idaho, tend to utilize more liquid-based systems to manage (flush or scrape) and store manure. Thus, the shift toward larger dairy cattle and swine facilities since 1990 has translated into an increasing use of liquid manure management systems, which have higher potential CH_4 emissions than dry systems. This significant shift in both

---

[6] Direct and indirect N_2O 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.

the dairy cattle and swine industries was accounted for by incorporating state and WMS-specific CH_4 conversion factor (MCF) values in combination with the 1992, 1997, 2002, 2007, 2012, and 2017 farm-size distribution data reported in the U.S. Department of Agriculture (USDA) *Census of Agriculture* (USDA 2019d).

In 2019, total N_2O emissions from manure management were estimated to be 19.6 MMT CO_2 Eq. (66 kt); in 1990, emissions were 14.0 MMT CO_2 Eq. (47 kt). These values include both direct and indirect N_2O emissions from manure management. Nitrous oxide emissions have increased since 1990. Small changes in N_2O emissions from individual animal groups exhibit the same trends as the animal group populations, with the overall net effect that N_2O emissions showed a 40 percent increase from 1990 to 2019 and a 0.9 percent increase from 2018 through 2019. Overall shifts toward liquid systems have driven down the emissions per unit of nitrogen excreted as dry manure handling systems have greater aerobic conditions that promote N_2O emissions.

Table 5-7 and Table 5-8 provide estimates of CH_4 and N_2O emissions from manure management by animal category.[7]

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

Gas/Animal Type	1990	2005	2015	2016	2017	2018	2019
CH₄ᵃ	**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	+	+	+	+	+	+	+
N₂Oᵇ	**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 Bisonᶜ	NA	NA	NA	NA	NA	NA	NA
Total	**51.1**	**67.9**	**75.4**	**77.7**	**78.5**	**81.1**	**82.0**

Notes: N_2O 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 CO_2 Eq.

NA (Not Available)

ᵃ Accounts for CH_4 reductions due to capture and destruction of CH_4 at facilities using anaerobic digesters.

ᵇ Includes both direct and indirect N_2O emissions.

ᶜ There are no American bison N_2O emissions from managed systems; American bison are maintained entirely on pasture, range, and paddock.

---

[7] Manure management emissions from camels are not estimated because there is no significant population of camels in the United States. Given the insignificance of estimated camel emissions in terms of the overall level and trend in national emissions, there are no immediate improvement plans to include this emissions category in the Inventory. See Annex 5 for more information on significance of estimated camel emissions.

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	+	+	+	+	0	+	+
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
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	NA	NA	NA	NA	NA	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)

[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.

Methodology

The methodologies presented in IPCC (2006) form the basis of the CH₄ and N₂O emission estimates for each animal type, including Tier 1, Tier 2, and use of the CEFM previously described for Enteric Fermentation. This combination of Tier 1 and Tier 2 methods was applied to all livestock animal types. This section presents a summary of the methodologies used to estimate CH₄ and N₂O emissions from manure management. For the current Inventory, time-series results were carried over from the 1990 to 2018 Inventory (i.e., 2020 submission) and a simplified approach was used to estimate manure management emissions for 2019.

See Annex 3.11 for more detailed information on the methodology (including detailed formulas and emission factors), data used to calculate CH₄ and N₂O emissions, and emission results (including input variables and results at the state-level) from manure management.

Methane Calculation Methods

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

- Animal population data (by animal type and state);
- Typical animal mass (TAM) data (by animal type);
- Portion of manure managed in each WMS, by state and animal type;
- Volatile solids (VS) production rate (by animal type and state or United States);
- Methane producing potential (B₀) of the volatile solids (by animal type); and

- Methane conversion factors (MCF), the extent to which the CH_4 producing potential is realized for each type of WMS (by state and manure management system, including the impacts of any biogas collection efforts).

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

- Annual animal population data for 1990 through 2018 for all livestock types, except goats, horses, mules and asses, and American bison were obtained from the USDA-NASS. For cattle, the USDA populations were utilized in conjunction with birth rates, detailed feedlot placement information, and slaughter weight data to create the transition matrix in the Cattle Enteric Fermentation Model (CEFM) that models cohorts of individual animal types and their specific emission profiles. The key variables tracked for each of the cattle population categories are described in Section 5.1 and in more detail in Annex 3.10. Goat population data for 1992, 1997, 2002, 2007, 2012, and 2017; horse and mule and ass population data for 1987, 1992, 1997, 2002, 2007, 2012, and 2017; and American bison population for 2002, 2007, 2012, and 2017 were obtained from the *Census of Agriculture* (USDA 2019d). American bison population data for 1990 through 1999 were obtained from the National Bison Association (1999).

- The TAM is an annual average weight that was obtained for animal types other than cattle from information in USDA's *Agricultural Waste Management Field Handbook* (USDA 1996), the American Society of Agricultural Engineers, Standard D384.1 (ASAE 1998) and others (Meagher 1986; EPA 1992; Safley 2000; ERG 2003b; IPCC 2006; ERG 2010a). For a description of the TAM used for cattle, see Annex 3.10.

- WMS usage was estimated for swine and dairy cattle for different farm size categories using state and regional data from USDA (USDA APHIS 1996; Bush 1998; Ott 2000; USDA 2016c) and EPA (ERG 2000a; EPA 2002a and 2002b; ERG 2018, ERG 2019). For beef cattle and poultry, manure management system usage data were not tied to farm size but were based on other data sources (ERG 2000a; USDA APHIS 2000; UEP 1999). For other animal types, manure management system usage was based on previous estimates (EPA 1992). American bison WMS usage was assumed to be the same as not on feed (NOF) cattle, while mules and asses were assumed to be the same as horses.

- VS production rates for all cattle except for calves were calculated by head for each state and animal type in the CEFM. VS production rates by animal mass for all other animals were determined using data from USDA's *Agricultural Waste Management Field Handbook* (USDA 1996 and 2008; ERG 2010b and 2010c) and data that was not available in the most recent *Handbook* were obtained from the American Society of Agricultural Engineers, Standard D384.1 (ASAE 1998) or the *2006 IPCC Guidelines* (IPCC 2006). American bison VS production was assumed to be the same as NOF bulls.

- B_0 was determined for each animal type based on literature values (Morris 1976; Bryant et al. 1976; Hashimoto 1981; Hashimoto 1984; EPA 1992; Hill 1982; Hill 1984).

- MCFs for dry systems were set equal to default IPCC factors based on state climate for each year (IPCC 2006). MCFs for liquid/slurry, anaerobic lagoon, and deep pit systems were calculated based on the forecast performance of biological systems relative to temperature changes as predicted in the van't Hoff-Arrhenius equation which is consistent with IPCC (2006) Tier 2 methodology.

- Data from anaerobic digestion systems with CH_4 capture and combustion were obtained from the EPA AgSTAR Program, including information available in the AgSTAR project database (EPA 2019). Anaerobic digester emissions were calculated based on estimated methane production and collection and destruction efficiency assumptions (ERG 2008).

- For all cattle except for calves, the estimated amount of VS (kg per animal-year) managed in each WMS for each animal type, state, and year were taken from the CEFM, assuming American bison VS production to be the same as NOF bulls. For animals other than cattle, the annual amount of VS (kg per year) from manure excreted in each WMS was calculated for each animal type, state, and year. This calculation multiplied the animal population (head) by the VS excretion rate (kg VS per 1,000 kg animal mass per

day), the TAM (kg animal mass per head) divided by 1,000, the WMS distribution (percent), and the number of days per year (365.25).

The estimated amount of VS managed in each WMS was used to estimate the CH_4 emissions (kg CH_4 per year) from 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 (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 were summed to determine the total U.S. CH_4 emissions. See details in Step 5 of Annex 3.11.

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

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

- EPA multiplied the national populations by the animal-specific 2018 implied emission factors[8] for CH_4 to calculate national-level 2019 CH_4 emissions estimates by animal type. These methods were utilized in order to maintain time-series consistency as referenced in Volume 1, Chapter 5 of the *2006 IPCC Guidelines*.

Nitrous Oxide Calculation Methods

The following inputs were used in the calculation of direct and indirect manure management N_2O emissions for 1990 through 2018:

- Animal population data (by animal type and state);
- TAM data (by animal type);
- Portion of manure managed in each WMS (by state and animal type);
- Total Kjeldahl N excretion rate (Nex);
- Direct N_2O emission factor (EF_{WMS});
- Indirect N_2O emission factor for volatilization ($EF_{volatilization}$);
- Indirect N_2O emission factor for runoff and leaching ($EF_{runoff/leach}$);
- Fraction of N loss from volatilization of NH_3 and NO_x ($Frac_{gas}$); and
- Fraction of N loss from runoff and leaching ($Frac_{runoff/leach}$).

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

- Nex for all cattle except for calves were calculated by head for each state and animal type in the CEFM. Nex rates by animal mass for all other animals were determined using data from USDA's *Agricultural Waste Management Field Handbook* (USDA 1996 and 2008; ERG 2010b and 2010c) and data from the American Society of Agricultural Engineers, Standard D384.1 (ASAE 1998) and IPCC (2006). American bison Nex were assumed to be the same as NOF bulls.[9]

- All N_2O emission factors (direct and indirect) were taken from IPCC (2006).

---

[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] Nex 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 N_2O emissions from unmanaged WMS are not included in the Manure Management source category, there are no N_2O emissions from American bison included in the Manure Management source category.

- Country-specific estimates for the fraction of N loss from volatilization ($Frac_{gas}$) and runoff and leaching ($Frac_{runoff/leach}$) were developed. $Frac_{gas}$ values were based on WMS-specific volatilization values as estimated from EPA's *National Emission Inventory - Ammonia Emissions from Animal Agriculture Operations* (EPA 2005). $Frac_{runoff/leaching}$ values were based on regional cattle runoff data from EPA's Office of Water (EPA 2002b; see Annex 3.11).

To estimate N_2O emissions for cattle (except for calves), the estimated amount of N excreted (kg per animal-year) that is managed in each WMS for each animal type, state, and year were taken from the CEFM. For calves and other animals, the amount of N excreted (kg per year) in manure in each WMS for each animal type, state, and year was calculated. The population (head) for each state and animal was multiplied by TAM (kg animal mass per head) divided by 1,000, the nitrogen excretion rate (Nex, in kg N per 1,000 kg animal mass per day), WMS distribution (percent), and the number of days per year.

Direct N_2O emissions were calculated by multiplying the amount of N excreted (kg per year) in each WMS by the 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. These emissions were summed over state, animal, and WMS to determine the total direct N_2O emissions (kg of N_2O per year). See details in Step 6 of Annex 3.11.

Indirect N_2O emissions from volatilization (kg N_2O per year) were then calculated by multiplying the amount of N excreted (kg per year) in each WMS by the fraction of N lost through volatilization ($Frac_{gas}$) divided by 100, the emission factor for volatilization ($EF_{volatilization}$, in kg N_2O per kg N), and the conversion factor of N_2O-N to N_2O. Indirect N_2O emissions from runoff and leaching (kg N_2O per year) were then calculated by multiplying the amount of N excreted (kg per year) in each WMS by the fraction of N lost through runoff and leaching ($Frac_{runoff/leach}$) divided by 100, and the emission factor for runoff and leaching ($EF_{runoff/leach}$, in kg N_2O per kg N), and the conversion factor of N_2O-N to N_2O. The indirect N_2O emissions from volatilization and runoff and leaching were summed to determine the total indirect N_2O emissions. See details in Step 6 of Annex 3.11.

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

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

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

- The national populations were multiplied by the animal-specific 2018 implied emission factors for N_2O (which combines both direct and indirect N_2O) to calculate national-level 2019 N_2O emissions estimates by animal type. These methods were utilized in order to maintain time-series consistency as referenced in Volume 1, Chapter 5 of the *2006 IPCC Guidelines*.

Uncertainty and Time-Series Consistency

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

The results of the Approach 2 quantitative uncertainty analysis are summarized in Table 5-9. Manure management 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, which indicates a range of 18 percent below to 20 percent above the actual 2019 emission estimate of 62.4 MMT 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 Eq. (or approximately 16 percent below and 24 percent above the actual 2019 emission estimate of 19.6 MMT CO_2 Eq.).

Table 5-9: Approach 2 Quantitative Uncertainty Estimates for CH_4 and N_2O (Direct and Indirect) Emissions from Manure Management (MMT CO_2 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_2O	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 N_2O emissions from managed systems and CH_4 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, B_0, 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 CH_4 emission factors for manure management (kg of CH_4 per head per year) were compared against the default IPCC (2006) values.[10] Table 5-10 presents the implied emission factors of kg of CH_4 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

---

[10] CH_4 implied emission factors were not calculated for 2019 due to the simplified emissions estimation approach used to estimate emissions for that year; therefore, those values are consistent with 2018.

trend towards larger farm sizes; large farms are more likely to manage manure as a liquid and therefore produce more CH_4 emissions.

Table 5-10: IPCC (2006) Implied Emission Factor Default Values Compared with Calculated Values for CH_4 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: CH_4 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.

In addition, default IPCC (2006) emission factors for N_2O were compared to the U.S. Inventory implied N_2O emission factors. Default N_2O emission factors from the *2006 IPCC Guidelines* were used to estimate N_2O emission from each WMS in conjunction with U.S.-specific Nex values. The implied emission factors differed from the U.S. Inventory values due to the use of U.S.-specific Nex values and differences in populations present in each WMS throughout the time series.

Recalculations Discussion

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

Planned Improvements

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

- Continuing to investigate new sources of WMS data. EPA is working with the USDA Natural Resources Conservation Service to collect data for potential improvements to the Inventory. EPA expects the next WMS systems to be updated for the next (i.e., 1990 to 2020) Inventory submission include poultry and beef cattle.

- Updating the B_0 data used in the Inventory, as data become available. EPA is conducting outreach with counterparts from USDA as to available data and research on B_0.

- Revising the methodology for population distribution to states where USDA population data are withheld due to disclosure concerns. These updates will be made in collaboration with the EPA National Emissions Inventory staff to improve consistency across U.S. inventories. EPA plans to incorporate these updates into the next (i.e., 1990 to 2020) Inventory submission.

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

- Revising the anaerobic digestion estimates to estimate CH_4 emissions *reductions* due to the use of anaerobic digesters (the Inventory currently estimates only emissions from anaerobic digestion systems).

- Investigating improved emissions estimate methodologies for swine pit systems with less than one month of storage (the recently updated swine WMS data included this WMS category).

- Comparing CH_4 and N_2O emission estimates with estimates from other models and more recent studies and compare the results to the Inventory.

- Comparing manure management emission estimates with on-farm measurement data to identify opportunities for improved estimates.

- Comparing VS and Nex data to literature data to identify opportunities for improved estimates.

- Improving collaboration with the Enteric Fermentation source category estimates. For future inventories, it may be beneficial to have the CEFM and Manure Management calculations in the same model, as they rely on much of the same activity data and they depend on each other's outputs to properly calculate emissions.

- Revising the uncertainty analysis to address changes that have been implemented to the CH_4 and N_2O estimates. EPA plans to line up the timing of performing the updated Manure Management uncertainty analysis with the uncertainty analysis for Enteric Fermentation.

5.3 Rice Cultivation (CRF Source Category 3C)

Most of the world's rice is grown on flooded fields (Baicich 2013) that create anaerobic conditions leading to CH_4 production through a process known as methanogenesis. Approximately 60 to 90 percent of the CH_4 produced by methanogenic bacteria in flooded rice fields is oxidized in the soil and converted to CO_2 by methanotrophic bacteria. The remainder is emitted to the atmosphere (Holzapfel-Pschorn et al. 1985; Sass et al. 1990) or transported as dissolved CH_4 into groundwater and waterways (Neue et al. 1997). Methane is transported to the atmosphere primarily through the rice plants, but some CH_4 also escapes via ebullition (i.e., bubbling through the water) and to a much lesser extent by diffusion through the water (van Bodegom et al. 2001).

Water management is arguably the most important factor affecting CH_4 emissions in rice cultivation, and improved water management has the largest potential to mitigate emissions (Yan et al. 2009). Upland rice fields are not flooded, and therefore do not produce CH_4, but large amounts of CH_4 can be emitted in continuously irrigated fields, which is the most common practice in the United States (USDA 2012). Single or multiple aeration events with drainage of a field during the growing season can significantly reduce these emissions (Wassmann et al. 2000a), but drainage may also increase N_2O emissions. Deepwater rice fields (i.e., fields with flooding depths greater than one meter, such as natural wetlands) tend to have fewer living stems reaching the soil, thus reducing the amount of CH_4 transport to the atmosphere through the plant compared to shallow-flooded systems (Sass 2001).

Other management practices also influence CH_4 emissions from flooded rice fields including rice residue straw management and application of organic amendments, in addition to cultivar selection due to differences in the

amount of root exudates[11] among rice varieties (Neue et al. 1997). These practices influence the amount of organic matter available for methanogenesis, and some practices, such as mulching rice straw or composting organic amendments, can reduce the amount of labile carbon and limit CH_4 emissions (Wassmann et al. 2000b). Fertilization practices also influence CH_4 emissions, particularly the use of fertilizers with sulfate (Wassmann et al. 2000b; Linquist et al. 2012), which can reduce CH_4 emissions. Other environmental variables also impact the methanogenesis process such as soil temperature and soil type. Soil temperature regulates the activity of methanogenic bacteria, which in turn affects the rate of CH_4 production. Soil texture influences decomposition of soil organic matter, but is also thought to have an impact on oxidation of CH_4 in the soil (Sass et al. 1994).

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

A combination of Tier 1 and 3 methods are used to estimate CH_4 emissions from rice cultivation across most of the time series, while a surrogate data method has been applied to estimate national emissions for 2016 to 2019 in this Inventory due to lack of data in the later years of the time series. National emission estimates based on surrogate data will be recalculated in a future Inventory with the Tier 1 and 3 methods as data becomes available.

Overall, rice cultivation is a minor source of CH_4 emissions in the United States relative to other source categories (see Table 5-11, Table 5-12, and Figure 5-3). Most emissions occur in Arkansas, California, Louisiana, Mississippi, Missouri and Texas. In 2019, CH_4 emissions from rice cultivation were 15.1 MMT CO_2 Eq. (602 kt). Annual emissions fluctuate between 1990 and 2019, which is largely due to differences in the amount of rice harvested areas over time, which has been decreasing over the past two decades. Consequently, emissions in 2019 are six percent lower than emissions in 1990.

Table 5-11: CH_4 Emissions from Rice Cultivation (MMT CO_2 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 CO_2 Eq.

---

[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.

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.

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		134		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.

Figure 5-3: Annual CH₄ Emissions from Rice Cultivation, 2015
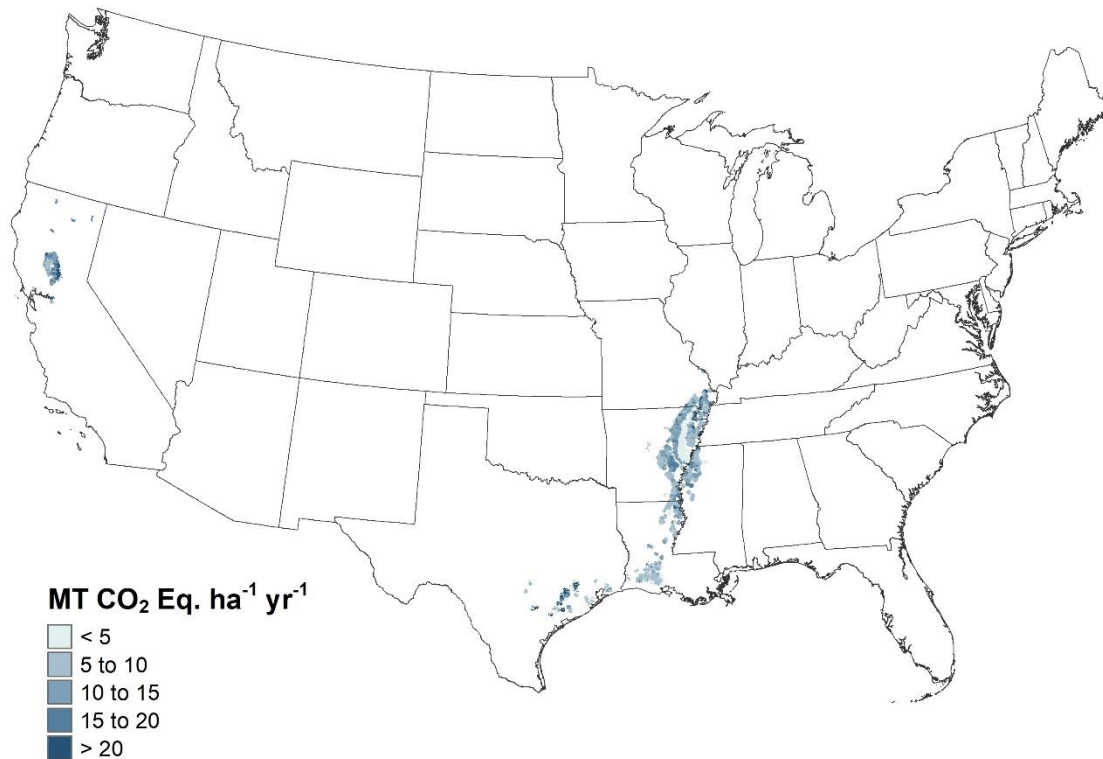
Note: Only national-scale emissions are estimated for 2016 through 2019 in this Inventory using the surrogate data method

described in the Methodology section; therefore, the fine-scale emission patterns in this map are based on the estimates for 2015.

Methodology

The methodology used to estimate CH_4 emissions from rice cultivation is based on a combination of IPCC Tier 1 and 3 approaches. The Tier 3 method utilizes the DayCent process-based model to estimate CH_4 emissions from rice cultivation (Cheng et al. 2013), and has been tested in the United States (see Annex 3.12) and Asia (Cheng et al. 2013, 2014). The model simulates hydrological conditions and thermal regimes, organic matter decomposition, root exudation, rice plant growth and its influence on oxidation of CH_4, as well as CH_4 transport through the plant and via ebullition (Cheng et al. 2013). The method captures the influence of organic amendments and rice straw management on methanogenesis in the flooded soils, and ratooning of rice crops with a second harvest during the growing season. In addition to CH_4 emissions, DayCent simulates soil C stock changes and N_2O emissions (Parton et al. 1987 and 1998; Del Grosso et al. 2010), and allows for a seamless set of simulations for crop rotations that include both rice and non-rice crops.

The Tier 1 method is applied to estimate CH_4 emissions from rice when grown in rotation with crops that are not simulated by DayCent, such as vegetable crops. The Tier 1 method is also used for areas converted between agriculture (i.e., cropland and grassland) and other land uses, such as forest land, wetland, and settlements. In addition, the Tier 1 method is used to estimate CH_4 emissions from organic soils (i.e., Histosols) and from areas with very gravelly, cobbly, or shaley soils (greater than 35 percent by volume). The Tier 3 method using DayCent has not been fully tested for estimating emissions associated with these crops and rotations, land uses, as well as organic soils or cobbly, gravelly, and shaley mineral soils.

The Tier 1 method for estimating CH_4 emissions from rice production utilizes a default base emission rate and scaling factors (IPCC 2006). The base emission rate represents emissions for continuously flooded fields with no organic amendments. Scaling factors are used to adjust the base emission rate for water management and organic amendments that differ from continuous flooding with no organic amendments. The method accounts for pre-season and growing season flooding; types and amounts of organic amendments; and the number of rice production seasons within a single year (i.e., single cropping, ratooning, etc.). The Tier 1 analysis is implemented in the Agriculture and Land Use National Greenhouse Gas Inventory (ALU) software (Ogle et al. 2016).[12]

Rice cultivation areas are based on cropping and land use histories recorded in the USDA National Resources Inventory (NRI) survey (USDA-NRCS 2018). The NRI is a statistically-based sample of all non-federal land, and includes 489,178 survey locations in agricultural land for the conterminous United States and Hawaii of which 1,960 include one or more years of rice cultivation. The Tier 3 method is used to estimate CH_4 emissions from 1,655 of the NRI survey locations, and the remaining 305 survey locations are estimated with the Tier 1 method. Each NRI survey location is associated with an "expansion factor" that allows scaling of CH_4 emission to the entire land base with rice cultivation (i.e., each expansion factor represents the amount of area with the same land-use/management history as the survey location). Land-use and some management information in the NRI (e.g., crop type, soil attributes, and irrigation) were collected on a 5-year cycle beginning in 1982, along with cropping 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 1994 to 1997). The NRI program began collecting annual data in 1998, with data currently available through 2015 (USDA-NRCS 2018). The current Inventory only uses NRI data through 2015 because newer data are not available, but will be incorporated when additional years of data are released by USDA-NRCS. The harvested rice areas in each state are presented in Table 5-13.

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

---

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

State							
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**

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.

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

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%

[a] Arkansas: *1990–2000* (Slaton 1999 through 2001); *2001–2011* (Wilson 2002 through 2007, 2009 through 2012); *2012–2013* (Hardke 2013, 2014). Estimates of ratooning for Missouri and Mississippi are based on the data from Arkansas.
[b] Florida - Ratoon: *1990–2000* (Schueneman 1997, 1999 through 2001); *2001* (Deren 2002); *2002–2003* (Kirstein 2003 through 2004, 2006); *2004* (Cantens 2004 through 2005); *2005–2013* (Gonzalez 2007 through 2014).
[c] Louisiana: *1990–2013* (Linscombe 1999, 2001 through 2014).
[d] Texas: *1990–2002* (Klosterboer 1997, 1999 through 2003); *2003–2004* (Stansel 2004 through 2005); *2005* (Texas Agricultural Experiment Station 2006); *2006–2013* (Texas Agricultural Experiment Station 2007 through 2014).

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

Winter flooding is another key practice associated with water management in rice fields, and the impact of winter flooding on CH_4 emissions is addressed in the Tier 3 and Tier 1 analyses. Flooding is used to prepare fields for the next growing season, and to create waterfowl habitat (Young 2013; Miller et al. 2010; Fleskes et al. 2005). Fitzgerald et al. (2000) suggests that as much as 50 percent of the annual emissions may occur during winter flooding. Winter flooding is a common practice with an average of 34 percent of fields managed with winter flooding in California (Miller et al. 2010; Fleskes et al. 2005), and approximately 21 percent of the fields managed with winter flooding in Arkansas (Wilson and Branson 2005 and 2006; Wilson and Runsick 2007 and 2008; Wilson

et al. 2009 and 2010; Hardke and Wilson 2013 and 2014; Hardke 2015). No data are available on winter flooding for Texas, Louisiana, Florida, Missouri, or Mississippi. For these states, the average amount of flooding is assumed to be similar to Arkansas. In addition, the amount of flooding is assumed to be relatively constant over the Inventory time series.

A surrogate data method is used to estimate emissions from 2016 to 2019 associated with the rice CH_4 emissions for Tier 1 and 3 methods. Specifically, a linear regression model with autoregressive moving-average (ARMA) errors was used to estimate the relationship between the surrogate data and emissions data from 1990 through 2015, which were derived using the Tier 1 and 3 methods (Brockwell and Davis 2016). Surrogate data are based on rice commodity statistics from USDA-NASS.[13] See Box 5-2 for more information about the surrogate data method.

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

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

where Y is the response variable (e.g., CH_4 emissions), $X\beta$ 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 CH_4 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).

Uncertainty and Time-Series Consistency

Sources of uncertainty in the Tier 3 method include management practices, uncertainties in model structure (i.e., algorithms and parameterization), and variance associated with the NRI sample. Sources of uncertainty in the IPCC (2006) Tier 1 method include the emission factors, management practices, and variance associated with the NRI sample. A Monte Carlo analysis was used to propagate uncertainties in the Tier 1 and 3 methods. For 2016 to 2019, there is additional uncertainty propagated through the Monte Carlo analysis associated with the surrogate data method (See Box 5-2 for information about propagating uncertainty with the surrogate data method). The uncertainties from the Tier 1 and 3 approaches are combined to produce the final CH_4 emissions estimate using simple error propagation (IPCC 2006). Additional details on the uncertainty methods are provided in Annex 3.12.

---

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

Rice cultivation CH_4 emissions in 2019 were estimated to be between 3.8 and 37.5 MMT CO_2 Eq. at a 95 percent confidence level, which indicates a range of 75 percent below to 149 percent above the 2019 emission estimate of 15.1 MMT CO_2 Eq. (see Table 5-15).

Table 5-15: Approach 2 Quantitative Uncertainty Estimates for CH_4 Emissions from Rice Cultivation (MMT CO_2 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**	**CH_4**	**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 CH_4 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 CH_4 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 CH_4 emissions of 2.2 MMT CO_2 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 N_2O 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 soils[16] (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 N_2O occur when N is transported from a site and is subsequently converted to N_2O; 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 (NH_4^+) to nitrate (NO_3^-), and denitrification is the anaerobic microbial reduction of nitrate to N_2. Nitrous oxide is a gaseous intermediate product in the reaction sequence of nitrification and denitrification.

[15] Asymbiotic N fixation is the fixation of atmospheric N_2 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 N_2O emissions from these soils.

[17] These processes entail volatilization of applied or mineralized N as NH_3 and NO_x, transformation of these gases in the atmosphere (or upon deposition), and deposition of the N primarily in the form of particulate NH_4^+, nitric acid (HNO_3), and NO_x. In addition, hydrological processes lead to leaching and runoff of NO_3^- that is converted to N_2O in aquatic systems, e.g., wetlands, rivers, streams and lakes. Note: N_2O emissions are not estimated for aquatic systems associated with N inputs from terrestrial systems in order to avoid double-counting.

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

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

N Volatilization

N Deposition to All Soils and Water Bodies

Synthetic N Fertilizers
Synthetic N fertilizer applied to soil

Organic Amendments
Includes both commercial and non-commercial fertilizers (i.e., animal manure, compost, sewage sludge, tankage, etc.)

Urine and Dung from Grazing Animals
Manure deposited on pasture, range, and paddock

Crop Residues
Includes above- and belowground residues for all crops (non-N and N-fixing (and from perennial forage crops and pastures following renewal)

Mineralization of Soil Organic Matter
Includes N converted to mineral form upon decomposition of soil organic matter

Asymbiotic Fixation
Fixation of atmospheric N₂ by bacteria living in soils that do not have a direct relationship with plants

N Flows:

N Inputs to Managed Soils

Direct N₂O Emissions

N Volatilization and Deposition

Indirect N₂O Emissions

Histosol Cultivation

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.

Leaching   Runoff

Surface Water

Groundwater

Agricultural soils produce the majority of N$_2$O emissions in the United States. Estimated emissions in 2019 are 344.6 MMT CO$_2$ Eq. (1,156 kt) (see Table 5-16 and Table 5-17). Annual N$_2$O emissions from agricultural soils are 9 percent greater in the 2019 compared to 1990, but emissions fluctuated between 1990 and 2019 due to inter-annual variability largely associated with weather patterns, synthetic fertilizer use, and crop production. From 1990 to 2019, cropland accounted for 68 percent of total direct emissions on average, while grassland accounted for 32 percent. On average, 79 percent of indirect emissions are from croplands and 21 percent from grasslands. Estimated direct and indirect N$_2$O emissions by sub-source category are shown in Table 5-18 and Table 5-19.

Table 5-16: N$_2$O Emissions from Agricultural Soils (MMT CO$_2$ 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.

Table 5-17: N$_2$O 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.

Table 5-18: Direct N$_2$O Emissions from Agricultural Soils by Land Use Type and N Input Type (MMT CO$_2$ Eq.)

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

	1990		2005		2015	2016	2017	2018	2019
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 CO_2 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.

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.

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

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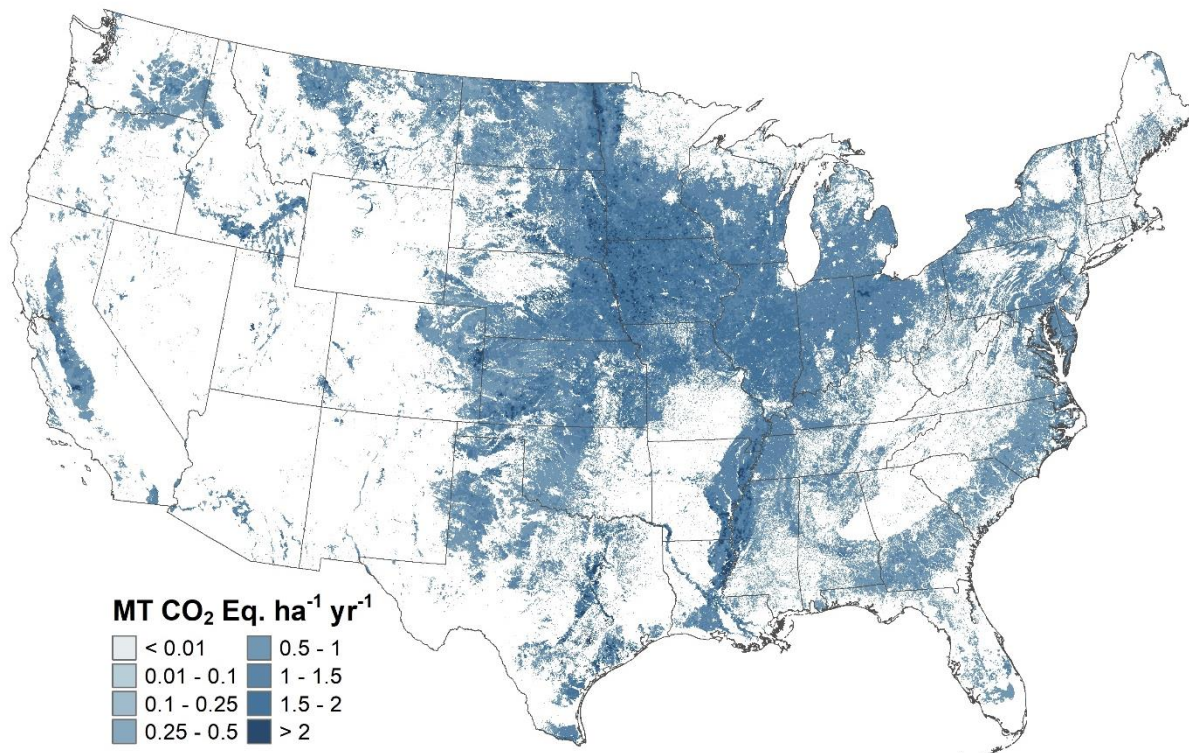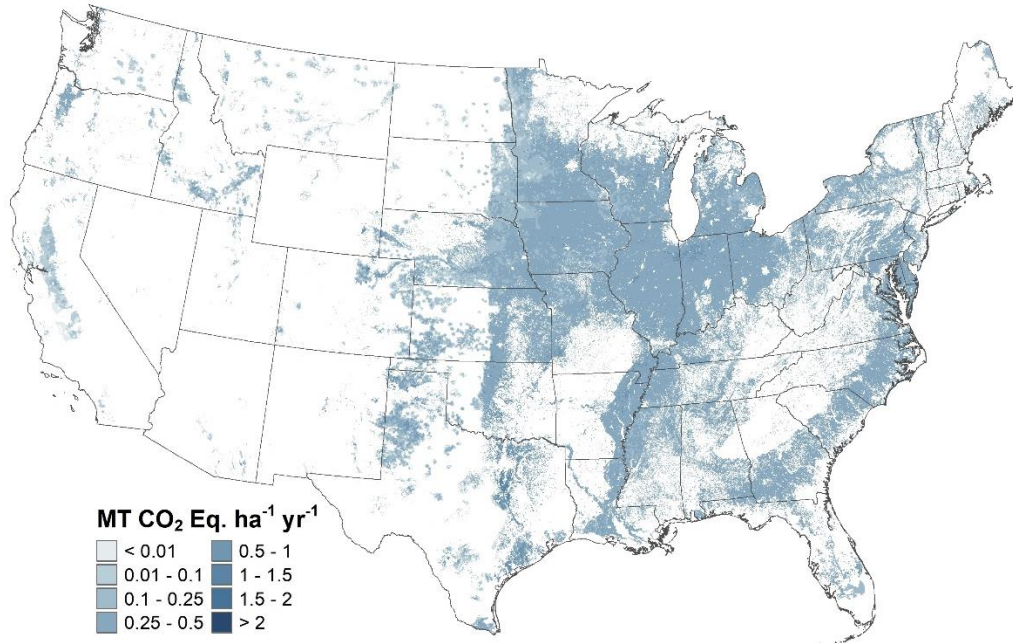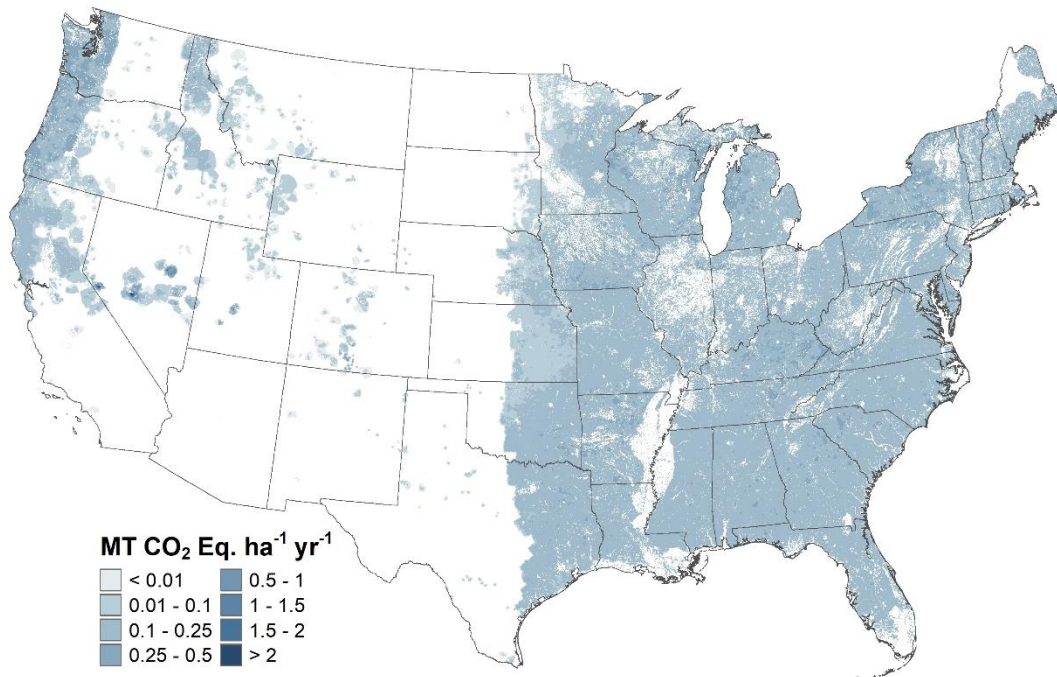

Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale emission patterns in this map are based on Inventory data from 2015.

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

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

Figure 5-6: Grasslands, 2015 Annual Direct N₂O Emissions Estimated Using the Tier 3 DayCent Model
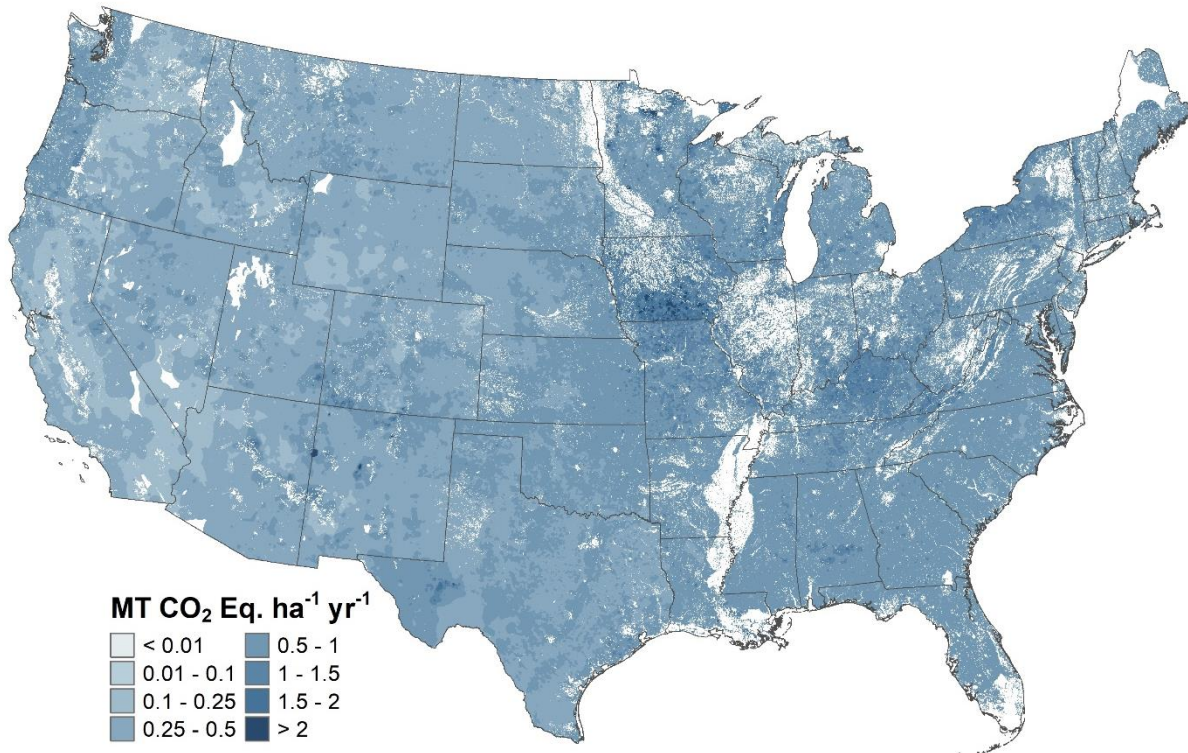
Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale emission patterns in this map are based on Inventory data from 2015.

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

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

Figure 5-7: Croplands, 2015 Annual Indirect N₂O Emissions from Volatilization Using the Tier 3 DayCent Model
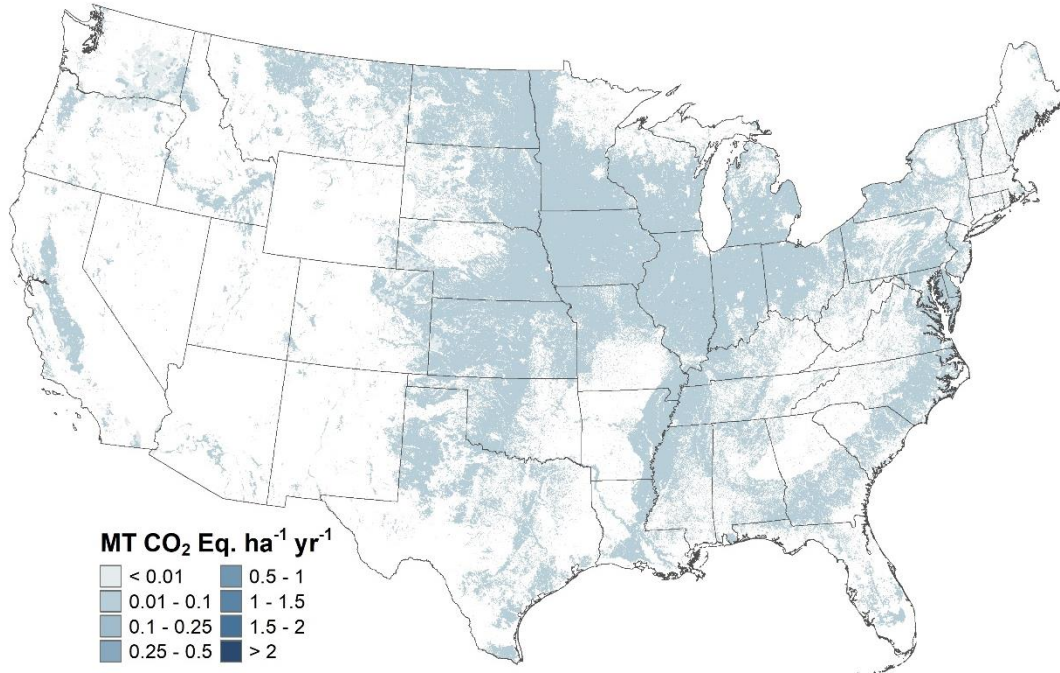
Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale emission patterns in this map are based on Inventory data from 2015.

Figure 5-8: Grasslands, 2015 Annual Indirect N₂O Emissions from Volatilization Using the Tier 3 DayCent Model
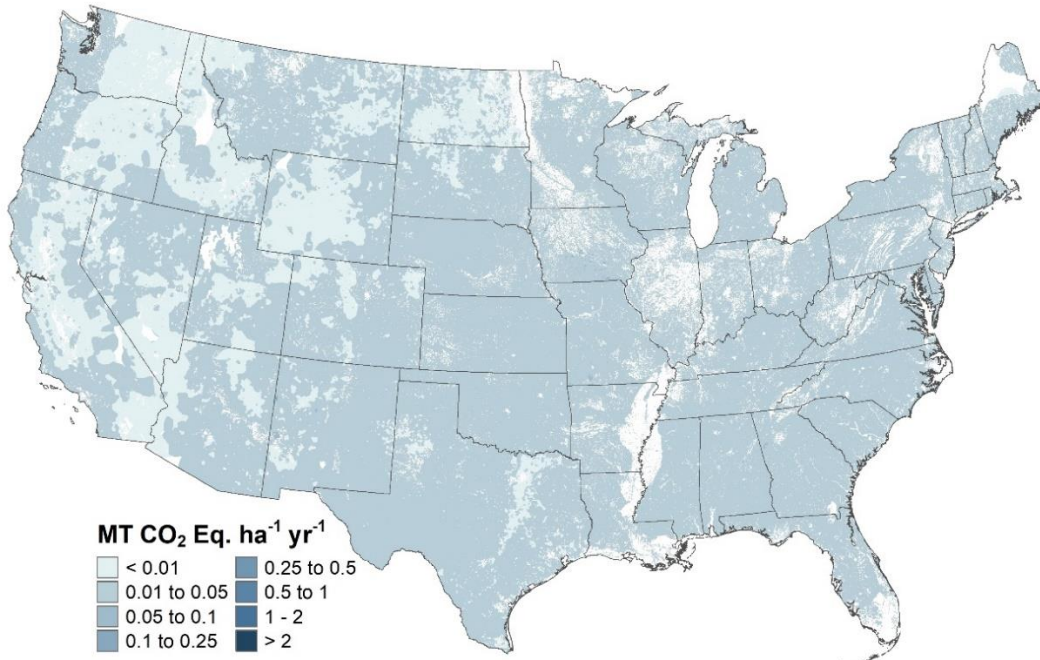
Note: Only national-scale emissions are estimated for 2016 to 2019 using a splicing method, and therefore the fine-scale emission patterns in this map are based on Inventory data from 2015.

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

MT CO$_2$ Eq. ha^{-1} yr^{-1}

< 0.01	0.5 - 1
0.01 - 0.1	1 - 1.5
0.1 - 0.25	1.5 - 2
0.25 - 0.5	> 2

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

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

MT CO$_2$ Eq. ha^{-1} yr^{-1}

< 0.01	0.5 - 1
0.01 - 0.1	1 - 1.5
0.1 - 0.25	1.5 - 2
0.25 - 0.5	> 2

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

Methodology

The *2006 IPCC Guidelines* (IPCC 2006) divide emissions from the agricultural soil management source category into five components, including (1) direct emissions from N additions to cropland and grassland mineral soils from synthetic fertilizers, biosolids (i.e., treated sewage sludge), crop residues (legume N-fixing and non-legume crops), and organic amendments; (2) direct emissions from soil organic matter mineralization due to land use and management change; (3) direct emissions from drainage of organic soils in croplands and grasslands; (4) direct emissions from soils due to manure deposited by livestock on PRP grasslands; and (5) indirect emissions from soils and water from N additions and manure deposition to soils that lead to volatilization, leaching, or runoff of N and subsequent conversion to N_2O.

In this source category, the United States reports on all croplands, as well as all managed grasslands, whereby anthropogenic greenhouse gas emissions are estimated consistent with the managed land concept (IPCC 2006), including direct and indirect N_2O emissions from asymbiotic fixation[18] and mineralization of N associated with decomposition of soil organic matter and residues. One recommendation from IPCC (2006) that has not been completely adopted is the estimation of emissions from grassland pasture renewal, which involves occasional plowing to improve forage production in pastures. Currently no data are available to address pasture renewal.

Direct N_2O Emissions

The methodology used to estimate direct N_2O emissions from agricultural soil management in the United States is based on a combination of IPCC Tier 1 and 3 approaches, along with application of a splicing method for latter years in the Inventory time series (IPCC 2006; Del Grosso et al. 2010) where data are not yet available. A Tier 3 process-based model (DayCent) is used to estimate direct emissions from a variety of crops that are grown on mineral (i.e., non-organic) soils, as well as the direct emissions from non-federal grasslands except for applications of biosolids (i.e., treated sewage sludge) (Del Grosso et al. 2010). The Tier 3 approach has been specifically designed and tested to estimate N_2O emissions in the United States, accounting for more of the environmental and management influences on soil N_2O emissions than the IPCC Tier 1 method (see Box 5-3 for further elaboration). Moreover, the Tier 3 approach addresses direct N_2O emissions and soil C stock changes from mineral cropland soils in a single analysis. Carbon and N dynamics are linked in plant-soil systems through biogeochemical processes of microbial decomposition and plant production (McGill and Cole 1981). Coupling the two source categories (i.e., agricultural soil C and N_2O) in a single inventory analysis ensures that there is consistent activity data and treatment of the processes, and interactions are considered between C and N cycling in soils.

The Tier 3 approach is based on the crop and land use histories recorded in the USDA National Resources Inventory (NRI) (USDA-NRCS 2018a). The NRI is a statistically-based sample of all non-federal land,[19] and includes 349,464 points on agricultural land for the conterminous United States that are included in the Tier 3 method. The Tier 1 approach is used to estimate the emissions from 175,527 locations in the NRI survey across the time series, which are designated as cropland or grassland (discussed later in this section). Each survey location is associated with an "expansion factor" that allows scaling of N_2O emissions from NRI points to the entire country (i.e., each expansion factor represents the amount of area with the same land-use/management history as the survey location). Each NRI survey location was sampled on a 5-year cycle from 1982 until 1997. For cropland, data were collected in 4 out of 5 years in the cycle (i.e., 1979 through 1982, 1984 through 1987, 1989 through 1992, and 1994 through 1997). In 1998, the NRI program began collecting annual data, which are currently available through 2015 (USDA-NRCS 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.

DayCent is used to estimate N_2O emissions associated with production of alfalfa hay, barley, corn, cotton, grass hay, grass-clover hay, oats, peanuts, potatoes, rice, sorghum, soybeans, sugar beets, sunflowers, tobacco and wheat, but is not applied to estimate N_2O emissions from other crops or rotations with other crops,[20] such as sugarcane, some vegetables, and perennial/horticultural crops. Areas that are converted between agriculture (i.e., cropland and grassland) and other land uses, such as forest land, wetland and settlements, are not simulated with DayCent. DayCent is also not used to estimate emissions from land areas with very gravelly, cobbly, or shaley soils in the topsoil (greater than 35 percent by volume in the top 30 cm of the soil profile), or to estimate emissions from drained organic soils (Histosols). The Tier 3 method has not been fully tested for estimating N_2O emissions associated with these crops and rotations, land uses, as well as organic soils or cobbly, gravelly, and shaley mineral soils. In addition, federal grassland areas are not simulated with DayCent due to limited activity data on land use histories. For areas that are not included in the DayCent simulations, Tier 1 methods are used to estimate emissions, including (1) direct emissions from N inputs for crops on mineral soils that are not simulated by DayCent; (2) direct emissions from PRP N additions on federal grasslands; (3) direct emissions for land application of biosolids (i.e., treated sewage sludge) to soils; and (4) direct emissions from drained organic soils in croplands and grasslands.

A splicing method is used to estimate soil N_2O emissions from 2016 to 2019 at the national scale because new NRI activity data are not available for those years. Specifically, linear regression models with autoregressive moving-average (ARMA) errors (Brockwell and Davis 2016) are used to estimate the relationship between surrogate data and the 1990 to 2015 emissions that are derived using the Tier 3 method. Surrogate data for these regression models includes corn and soybean yields from USDA-NASS statistics,[21] and weather data from the PRISM Climate Group (PRISM 2018). For the Tier 1 method, a linear-time series model is used to estimate emissions from 2016 to 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/>.

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

<div style="border:1px solid #000">

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

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

where Y is the response variable (e.g., soil nitrous oxide), $X\beta$ 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 $X\beta$ 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).

</div>

Tier 3 Approach for Mineral Cropland Soils

The DayCent biogeochemical model (Parton et al. 1998; Del Grosso et al. 2001 and 2011) is used to estimate direct N_2O emissions from mineral cropland soils that are managed for production of a wide variety of crops (see list in previous section) based on the crop histories in the 2015 NRI (USDA-NRCS 2018a). Crops simulated by DayCent are grown on approximately 85 percent of total cropland area in the United States. The model simulates net primary productivity (NPP) using the NASA-CASA production algorithm MODIS Enhanced Vegetation Index (EVI) products, MOD13Q1 and MYD13Q1[22] (Potter et al. 1993, 2007). The model simulates soil temperature and water dynamics, 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.

soil attributes from the Soil Survey Geographic Database (SSURGO) (Soil Survey Staff 2019). DayCent is used to estimate direct N_2O emissions due to mineral N available from the following sources: (1) application of synthetic fertilizers; (2) application of livestock manure; (3) retention of crop residues in the field for N-fixing legumes and non-legume crops and subsequent mineralization of N during microbial decomposition (i.e., leaving residues in the field after harvest instead of burning or collecting residues); (4) mineralization of N from decomposition of soil organic matter; and (5) asymbiotic fixation.

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

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

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

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

The IPCC method considers crop residue N and N mineralized from soil organic matter as activity data. However, they are not treated as activity data in DayCent simulations because residue production, symbiotic N fixation (e.g., legumes), mineralization of N from soil organic matter, and asymbiotic N fixation are internally generated by the model as part of the simulation. In other words, DayCent accounts for the influence of symbiotic N fixation, mineralization of N from soil organic matter and crop residue retained in the field, and asymbiotic N fixation on N_2O emissions, but these are not model inputs.

The N_2O emissions from crop residues are reduced by approximately 3 percent (the assumed average burned portion for crop residues in the United States) to avoid double counting associated with non-CO_2 greenhouse gas

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

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

Nitrous oxide emissions and 95 percent confidence intervals are estimated for each year between 1990 and 2015 using the DayCent model. However, note that the areas have been modified in the original NRI survey through a process in which the Forest Inventory and Analysis (FIA) survey data and the National Land Cover Dataset (Yang et al. 2018) are harmonized with the NRI data. This process ensures that the land use areas are consistent across all land use categories (See Section 6.1, Representation of the U.S. Land Base for more information). Further elaboration on the methodology and data used to estimate N_2O emissions from mineral soils are described in Annex 3.12.

For the Tier 3 method, soil N_2O emissions from 2016 to 2019 associated with mineral soils in croplands are estimated using a splicing method that accounts for uncertainty in the original inventory data and the splicing method (See Box 5-4). Annual data are currently available through 2015 (USDA-NRCS 2018a), and the Inventory time series will be updated in the future when new NRI data are released.

Nitrous oxide emissions from managed agricultural lands are the result of interactions among anthropogenic activities (e.g., N fertilization, manure application, tillage) and other driving variables, such as weather and soil characteristics. These factors influence key processes associated with N dynamics in the soil profile, including immobilization of N by soil microbial organisms, decomposition of organic matter, plant uptake, leaching, runoff, and volatilization, as well as the processes leading to N_2O production (nitrification and denitrification). It is not possible to partition N_2O emissions into each anthropogenic activity directly from model outputs due to the complexity of the interactions (e.g., N_2O emissions from synthetic fertilizer applications cannot be distinguished from those resulting from manure applications). To approximate emissions by activity, the amount of mineral N added to the soil, or made available through decomposition of soil organic matter and plant litter, as well as asymbiotic fixation of N from the atmosphere, is determined for each N source and then divided by the total amount of mineral N in the soil according to the DayCent model simulation. The percentages are then multiplied by the total of direct N_2O emissions in order to approximate the portion attributed to N management practices. This approach is only an approximation because it assumes that all N made available in soil has an equal probability of being released as N_2O, regardless of its source, which is unlikely to be the case (Delgado et al. 2009). However, this approach allows for further disaggregation of emissions by source of N, which is valuable for reporting purposes and is analogous to the reporting associated with the IPCC (2006) Tier 1 method, in that it associates portions of the total soil N_2O emissions with individual sources of N.

Tier 1 Approach for Mineral Cropland Soils

The IPCC (2006) Tier 1 methodology is used to estimate direct N_2O emissions for mineral cropland soils that are not simulated by DayCent (e.g., DayCent has not been parametrized to simulate all crop types and some soil types such as *Histosols*). For the Tier 1 method, estimates of direct N_2O emissions from N applications are based on mineral soil N that is made available from the following practices: (1) the application of synthetic commercial fertilizers; (2) application of managed manure and non-manure commercial organic fertilizers; and (3) decomposition and

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

- A process-of-elimination approach is used to estimate synthetic N fertilizer additions for crop areas that are not simulated by DayCent. The total amount of fertilizer used on farms has been estimated at the county-level by the USGS using sales records from 1990 to 2012 (Brakebill and Gronberg 2017). For 2013 through 2015, county-level fertilizer used on-farms is adjusted based on annual fluctuations in total U.S. fertilizer sales (AAPFCO 2013 through 2017).[23] After subtracting the portion of fertilizer applied to crops and grasslands simulated by DayCent (see Tier 3 Approach for Mineral Cropland Soils and Direct N_2O Emissions from Grassland Soils sections for information on data sources), the remainder of the total fertilizer used on farms is assumed to be applied to crops that are not simulated by DayCent.

- Similarly, a process-of-elimination approach is used to estimate manure N additions for crops that are not simulated by DayCent. The total amount of manure available for land application to soils has been estimated with methods described in the Manure Management section (Section 5.2) and annex (Annex 3.11). The amount of manure N applied in the Tier 3 approach to crops and grasslands is subtracted from total annual manure N available for land application (see Tier 3 Approach for Mineral Cropland Soils and Direct N_2O Emissions from Grassland Soils sections for information on data sources). This difference is assumed to be applied to crops that are not simulated by DayCent.

- Commercial organic fertilizer additions are based on organic fertilizer consumption statistics, which are converted from mass of fertilizer to units of N using average organic fertilizer N content, which range between 2.3 to 4.2 percent across the time series (TVA 1991 through 1994; AAPFCO 1995 through 2017). Commercial fertilizers do include dried manure and biosolids (i.e., treated sewage sludge), but the amounts are removed from the commercial fertilizer data to avoid double counting[24] with the manure N dataset described above and the biosolids (i.e., treated sewage sludge) amendment data discussed later in this section.

- Crop residue N is derived by combining amounts of above- and below-ground biomass, which are determined based on NRI crop area data (USDA-NRCS 2018a), crop production yield statistics (USDA-NASS 2019), dry matter fractions (IPCC 2006), linear equations to estimate above-ground biomass given dry matter crop yields from harvest (IPCC 2006), ratios of below-to-above-ground biomass (IPCC 2006), and N contents of the residues (IPCC 2006). N inputs from residue were reduced by 3 percent to account for average residue burning portions in the United States.

The total increase in soil mineral N from applied fertilizers and crop residues is multiplied by the IPCC (2006) default emission factor to derive an estimate of direct N_2O emissions using the Tier 1 method. Further elaboration on the methodology and data used to estimate N_2O emissions from mineral soils are described in Annex 3.12.

Soil N_2O emissions from 2016 to 2019 for Tier 1 mineral soil emissions are estimated using a splicing method that is described in Box 5-4, with the exception of the crop residue N, which is only estimated with the data splicing method for 2019. As with the Tier 3 method, the time series that is based on the splicing methods will be 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.

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

As with N_2O emissions from croplands, the Tier 3 process-based DayCent model and Tier 1 method described in IPCC (2006) are combined to estimate emissions from non-federal grasslands and PRP manure N additions for federal grasslands, respectively. Grassland includes pasture and rangeland that produce grass or mixed grass/legume forage primarily for livestock grazing. Rangelands are extensive areas of native grassland that are not intensively managed, while pastures are seeded grassland (possibly following tree removal) that may also have additional management, such as irrigation, fertilization, or inter-seeding legumes. DayCent is used to simulate N_2O emissions from NRI survey locations (USDA-NRCS 2018a) on non-federal grasslands resulting from manure deposited by livestock directly onto pastures and rangelands (i.e., PRP manure), N fixation from legume seeding, managed manure amendments (i.e., manure other than PRP manure such as Daily Spread or manure collected from other animal waste management systems such as lagoons and digesters), and synthetic fertilizer application. Other N inputs are simulated within the DayCent framework, including N input from mineralization due to decomposition of soil organic matter and N inputs from senesced grass litter, as well as asymbiotic fixation of N from the atmosphere. The simulations used the same weather, soil, and synthetic N fertilizer data as discussed under the Tier 3 Approach in the Mineral Cropland Soils section. Mineral N fertilization rates are based on data from the Carbon Sequestration Rural Appraisals (CSRA) conducted by the USDA-NRCS (USDA-NRCS, unpublished data). The CSRA was a solicitation of expert knowledge from USDA-NRCS staff throughout the United States to support the Inventory. Biological N fixation is simulated within DayCent, and therefore is not an input to the model.

Manure N deposition from grazing animals in PRP systems (i.e., PRP manure N) is a key input of N to grasslands. The amounts of PRP manure N applied on non-federal grasslands for each NRI survey location are based on the amount of N excreted by livestock in PRP systems that is estimated in the Manure Management section (See Section 5.2 and Annex 3.10). The total amount of N excreted in each county is divided by the grassland area to estimate the N input rate associated with PRP manure. The resulting rates are a direct input into the DayCent simulations. The N input is subdivided between urine and dung based on a 50:50 split. DayCent simulations of non-federal grasslands accounted for approximately 61 percent of total PRP manure N in aggregate across the country.[25] The remainder of the PRP manure N in each state is assumed to be excreted on federal grasslands, and the N_2O emissions are estimated using the IPCC (2006) Tier 1 method.

Biosolids (i.e., treated sewage sludge) are assumed to be applied on grasslands. Application of biosolids is estimated from data compiled by EPA (1993, 1999, 2003), McFarland (2001), and NEBRA (2007) (see Section 7.2 Wastewater Treatment for a detailed discussion of the methodology for estimating treated sewage sludge available for land application application). Biosolids data are only available at the national scale, and it is not possible to associate application with specific soil conditions and weather at NRI survey locations. Therefore, DayCent could not be used to simulate the influence of biosolids on N_2O emissions from grassland soils, and consequently, emissions from biosolids are estimated using the IPCC (2006) Tier 1 method.

Soil N_2O emission estimates from DayCent are adjusted using a structural uncertainty estimator accounting for uncertainty in model algorithms and parameter values (Del Grosso et al. 2010). There is also sampling uncertainty for the NRI survey that is propagated through the estimate with replicate sampling weights associated with the survey. N_2O emissions for the PRP manure N deposited on federal grasslands and applied biosolids N are estimated using the Tier 1 method by multiplying the N input by the default emission factor. Emissions from manure N are estimated at the state level and aggregated to the entire country, but emissions from biosolids N are calculated exclusively at the national scale. Further elaboration on the methodology and data used to estimate N_2O emissions from mineral soils are described in Annex 3.12.

Soil N_2O emissions and 95 percent confidence intervals are estimated for each year between 1990 and 2015 based on the Tier 1 and 3 methods, with the exception of biosolids (discussed below). Emissions from 2016 to 2019 are estimated using a splicing method as described in Box 5-4. As with croplands, estimates for 2016 to 2019 will be

---

[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 N_2O emissions.

recalculated in a future Inventory when new NRI data are released by USDA. Biosolids application data are compiled through 2019 in this Inventory, and therefore soil N_2O emissions and confidence intervals are estimated using the Tier 1 method for all years in the time series without application of the splicing method.

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

The IPCC (2006) Tier 1 method is used to estimate direct N_2O emissions due to drainage of organic soils in croplands and grasslands at a state scale. State-scale estimates of the total area of drained organic soils are obtained from the 2015 NRI (USDA-NRCS 2018a) using soils data from the Soil Survey Geographic Database (SSURGO) (Soil Survey Staff 2019). Temperature data from the PRISM Climate Group (PRISM 2018) are used to subdivide areas into temperate and tropical climates according to the climate classification from IPCC (2006). To estimate annual emissions, the total temperate area is multiplied by the IPCC default emission factor for temperate regions, and the total tropical area is multiplied by the IPCC default emission factor for tropical regions (IPCC 2006). Annual NRI data are only available between 1990 and 2015, but the time series was adjusted using data from the Forest Inventory and Analysis Program (USFS 2019) in order to estimate emissions from 2016 to 2018. The land representation data have not been updated for this Inventory so the amount of drained organic soils is assumed to be the same in 2019 as the estimated areas in 2018, and consequently the emissions in 2019 are also assumed to the same as 2018. Further elaboration on the methodology and data used to estimate N_2O emissions from organic soils are described in Annex 3.12.

Total Direct N_2O Emissions from Cropland and Grassland Soils

Annual direct emissions from the Tier 1 and 3 approaches for mineral and drained organic soils occurring in both croplands and grasslands are summed to obtain the total direct N_2O emissions from agricultural soil management (see Table 5-16 and Table 5-17).

Indirect N_2O Emissions Associated with Nitrogen Management in Cropland and Grasslands

Indirect N_2O emissions occur when mineral N applied or made available through anthropogenic activity is transported from the soil either in gaseous or aqueous forms and later converted into N_2O. There are two pathways leading to indirect emissions. The first pathway results from volatilization of N as NO_x and NH_3 following application of synthetic fertilizer, organic amendments (e.g., manure, biosolids), and deposition of PRP manure. Nitrogen made available from mineralization of soil organic matter and residue, including N incorporated into crops and forage from symbiotic N fixation, and input of N from asymbiotic fixation also contributes to volatilized N emissions. Volatilized N can be returned to soils through atmospheric deposition, and a portion of the deposited N is emitted to the atmosphere as N_2O. The second pathway occurs via leaching and runoff of soil N (primarily in the form of NO_3^-) that is made available through anthropogenic activity on managed lands, mineralization of soil organic matter and residue, including N incorporated into crops and forage from symbiotic N fixation, and inputs of N into the soil from asymbiotic fixation. The NO_3^- is subject to denitrification in water bodies, which leads to N_2O emissions. Regardless of the eventual location of the indirect N_2O emissions, the emissions are assigned to the original source of the N for reporting purposes, which here includes croplands and grasslands.

Tier 1 and 3 Approaches for Indirect N_2O Emissions from Atmospheric Deposition of Volatilized N

The Tier 3 DayCent model and IPCC (2006) Tier 1 methods are combined to estimate the amount of N that is volatilized and eventually emitted as N_2O. DayCent is used to estimate N volatilization for land areas whose direct emissions are simulated with DayCent (i.e., most commodity and some specialty crops and most grasslands). The N inputs included are the same as described for direct N_2O emissions in the Tier 3 Approach for Mineral Cropland Soils and Direct N_2O Emissions from Grassland Soils sections. Nitrogen volatilization from all other areas is estimated using the Tier 1 method with default IPCC fractions for N subject to volatilization (i.e., N inputs on

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

The IPCC (2006) default emission factor is multiplied by the amount of volatilized N generated from both DayCent and Tier 1 methods to estimate indirect N_2O emissions occurring following re-deposition of the volatilized N (see Table 5-19). Further elaboration on the methodology and data used to estimate indirect N_2O emissions are described in Annex 3.12.

Tier 1 and 3 Approaches for Indirect N_2O Emissions from Leaching/Runoff

As with the calculations of indirect emissions from volatilized N, the Tier 3 DayCent model and IPCC (2006) Tier 1 method are combined to estimate the amount of N that is subject to leaching and surface runoff into water bodies, and eventually emitted as N_2O. DayCent is used to simulate the amount of N transported from lands in the Tier 3 Approach. Nitrogen transport from all other areas is estimated using the Tier 1 method and the IPCC (2006) default factor for the proportion of N subject to leaching and runoff associated with N applications on croplands that are not simulated by DayCent, applications of biosolids on grasslands, and PRP manure N excreted on federal grasslands.

For both the DayCent Tier 3 and IPCC (2006) Tier 1 methods, nitrate leaching is assumed to be an insignificant source of indirect N_2O in cropland and grassland systems in arid regions, as discussed in IPCC (2006). In the United States, the threshold for significant nitrate leaching is based on the potential evapotranspiration (PET) and rainfall amount, similar to IPCC (2006), and is assumed to be negligible in regions where the amount of precipitation does not exceed 80 percent of PET (Note: All irrigated systems are assumed to have significant amounts of leaching of N even in drier climates).

For leaching and runoff data estimated by the Tier 3 and Tier 1 approaches, the IPCC (2006) default emission factor is used to estimate indirect N_2O emissions that occur in groundwater and waterways (see Table 5-19). Further elaboration on the methodology and data used to estimate indirect N_2O emissions are described in Annex 3.12.

Indirect soil N_2O emissions from 2016 to 2019 are estimated using the splicing method that is described in Box 5-4. As with the direct N_2O emissions, the time series will be recalculated in a future Inventory report when new activity data are compiled.

Uncertainty and Time-Series Consistency

Uncertainty is estimated for each of the following five components of N_2O emissions from agricultural soil management: (1) direct emissions simulated by DayCent; (2) the components of indirect emissions (N volatilized and leached or runoff) simulated by DayCent; (3) direct emissions estimated with the IPCC (2006) Tier 1 method; (4) the components of indirect emissions (N volatilized and leached or runoff) estimated with the IPCC (2006) Tier 1 method; and (5) indirect emissions estimated with the IPCC (2006) Tier 1 method. Uncertainty in direct emissions as well as the components of indirect emissions that are estimated from DayCent are derived from a Monte Carlo Analysis (consistent with IPCC Approach 2), addressing uncertainties in model inputs and structure (i.e., algorithms and parameterization) (Del Grosso et al. 2010). For 2016 to 2019, there is additional uncertainty propagated through the Monte Carlo Analysis associated with the splicing method (See Box 5-4).

Simple error propagation methods (IPCC 2006) are used to derive confidence intervals for direct emissions estimated with the IPCC (2006) Tier 1 method, the proportion of volatilization and leaching or runoff estimated with the IPCC (2006) Tier 1 method, and indirect N_2O emissions. Uncertainty in the splicing method is also included in the error propagation for 2016 to 2019 (see Box 5-4). Additional details on the uncertainty methods are provided in Annex 3.12.

Table 5-20 shows the combined uncertainty for direct soil N_2O emissions. The estimated emissions ranges from 31 percent below to 31 percent above the 2019 emission estimate of 290.4 MMT CO_2 Eq. The combined uncertainty for indirect soil N_2O emissions ranges from 71 percent below to 154 percent above the 2019 estimate of 54.2 MMT CO_2 Eq.

Table 5-20: Quantitative Uncertainty Estimates of N$_2$O Emissions from Agricultural Soil Management in 2019 (MMT CO$_2$ 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 N$_2$O emissions from managed croplands and grasslands in Hawaii and Alaska. The Inventory currently includes the N$_2$O 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 N$_2$O emissions and NO$_3^-$ 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 N$_2$O 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 N$_2$O 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.

Planned Improvements

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

Several planned improvements are underway associated with improving the DayCent biogeochemical model. These improvements include a better representation of plant phenology, particularly senescence events following grain filling in crops. In addition, crop parameters associated with temperature and water stress effects on plant production will be further improved in DayCent with additional model calibration. Model development is underway to represent the influence of nitrification inhibitors and slow-release fertilizers (e.g., polymer-coated fertilizers) on N_2O emissions. Experimental study sites will continue to be added for quantifying model structural uncertainty. Studies that have continuous (daily) measurements of N_2O (e.g., Scheer et al. 2013) will be given priority.

Improvements are underway to simulate crop residue burning in the DayCent model based on the amount of crop residues burned according to the data that is used in the Field Burning of Agricultural Residues source category (see Section 5.7). Alaska and Hawaii are not included for all sources in the current Inventory for agricultural soil management, with the exception of N_2O emissions from drained organic soils in croplands and grasslands for Hawaii, synthetic fertilizer and PRP N amendments for grasslands in Alaska and Hawaii. There is also an improvement based on updating the Tier 1 emission factor for N_2O emissions from drained organic soils by using the revised factor in the 2013 Supplement to the *2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands* (IPCC 2013).

In addition, there is a planned improvement associated with implementation of the Tier 1 method. Specifically, soil N_2O emissions will be estimated and reported for N mineralization from soil organic matter decomposition that is accelerated with *Forest Land Converted to Cropland* and *Grassland Converted to Cropland*. A review of available data on biosolids (i.e., treated sewage sludge) application will also be undertaken to improve the distribution of biosolids application on croplands, grasslands and settlements.

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

5.5 Liming (CRF Source Category 3G)

Crushed limestone ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) are added to soils by land managers to increase soil pH (i.e., to reduce acidification). Carbon dioxide emissions occur as these compounds react with hydrogen ions in soils. The rate of degradation of applied limestone and dolomite depends on the soil conditions, soil type, climate regime, and whether limestone or dolomite is applied. Emissions from limestone and dolomite that are used in industrial processes (e.g., cement production, glass production, etc.) are reported in the IPPU chapter. Emissions from liming of soils have fluctuated between 1990 and 2019 in the United States, ranging from 2.2 MMT CO_2 Eq. to 6.0 MMT CO_2 Eq. across the entire time series. In 2019, liming of soils in the United States resulted in emissions of 2.4 MMT CO_2 Eq. (0.7 MMT C), representing a 52 percent decrease in emissions since 1990 (see Table 5-21 and Table 5-22). The trend is driven by variation in the amount of limestone and dolomite applied to soils over the time period.

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.

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.

Methodology

Carbon dioxide emissions from application of limestone and dolomite to soils were estimated using a Tier 2 methodology consistent with IPCC (2006). The annual amounts of limestone and dolomite, which are applied to soils (see Table 5-23), were multiplied by CO_2 emission factors from West and McBride (2005). These country-specific emission factors (0.059 metric ton C/metric ton limestone, 0.064 metric ton C/metric ton dolomite) are lower than the IPCC default emission factors because they account for the portion of carbonates that are transported from soils through hydrological processes and eventually deposited in ocean basins (West and McBride 2005). This analysis of lime dissolution is based on studies in the Mississippi River basin, where the vast majority of lime application occurs in the United States (West 2008). Moreover, much of the remaining lime application is occurring under similar precipitation regimes, and so the emission factors are considered a reasonable approximation for all lime application in the United States (West 2008) (See Box 5-5).

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

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 rives and eventually the ocean where $CaCO_3$ 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 CO_2 Eq. using the country-specific factors. In contrast, emissions would be estimated at 5.0 MMT CO_2 Eq. using the IPCC (2006) default emission factors.

Data on "specified" limestone and dolomite amounts were used directly in the emission calculation because the end use is provided by the manufacturers and can be used to directly determine the amount applied to soils. However, it is not possible to determine directly how much of the limestone and dolomite is applied to soils for manufacturer surveys in the "unspecified" and "estimated" categories. For these categories, the amounts of crushed limestone and dolomite applied to soils were determined by multiplying the percentage of total "specified" limestone and dolomite production that is applied to soils, by the total amounts of "unspecified" and "estimated" limestone and dolomite production. In other words, the proportion of total "unspecified" and "estimated" crushed limestone and dolomite that was applied to soils is proportional to the amount of total "specified" crushed limestone and dolomite that was applied to soils.

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

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

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

Uncertainty and Time-Series Consistency

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

A Monte Carlo (Approach 2) uncertainty analysis was applied to estimate the uncertainty in CO_2 emissions from liming. The results of the Approach 2 quantitative uncertainty analysis are summarized in Table 5-24. Carbon dioxide emissions from carbonate lime application to soils in 2019 were estimated to be between -0.27 and 4.61 MMT CO_2 Eq. at the 95 percent confidence level. This confidence interval represents a range of 111 percent below to 88 percent above the 2019 emission estimate of 2.4 MMT CO_2 Eq. Note that there is a small probability of a negative emissions value leading to a net uptake of CO_2 from the atmosphere. Net uptake occurs due to the dominance of the carbonate lime dissolving in carbonic acid rather than nitric acid (West and McBride 2005).

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

Source	Gas	2019 Emission Estimate (MMT CO₂ Eq.)	Uncertainty Range Relative to Emission Estimate[a] (MMT CO₂ 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 ($CO(NH_2)_2$) as a fertilizer leads to greenhouse gas emissions through the release of CO_2 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 (NH_4^+), hydroxyl ion (OH), and bicarbonate (HCO_3^-). The bicarbonate then evolves into CO_2 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 CO_2. 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 CO_2 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 $CO(NH_2)_2$ applied to soils is emitted as CO_2. 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 CO_2 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 CO_2 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 CO$_2$ Emissions from Urea Fertilization (MMT CO$_2$ 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 CO$_2$ 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 CO$_2$ 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 CO$_2$ 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 CO$_2$ 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 CO$_2$ 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 CO_2 emissions because the C released to the atmosphere as CO_2 during burning is reabsorbed during the next growing season by the crop. However, crop residue burning is a net source of CH_4, N_2O, 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, sugarcane and wheat (McCarty 2009). In 2019, CH_4 and N_2O emissions from field burning of agricultural residues were 0.4 MMT CO_2 Eq. (17 kt) and 0.2 MMT CO_2 Eq. (1 kt), respectively (Table 5-29 and Table 5-30). Annual emissions of CH_4 and N_2O 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: CH_4 and N_2O Emissions from Field Burning of Agricultural Residues (MMT CO_2 Eq.)

Gas/Crop Type	1990	2005	2015	2016	2017	2018	2019
CH₄	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**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**	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**	**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

Crop Type	1990		2005		2015	2016	2017	2018	2019
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 CO_2 Eq.

Note: Totals may not sum due to independent rounding.

Table 5-30: CH_4, N_2O, CO, and NO_x Emissions from Field Burning of Agricultural Residues (kt)

Gas/Crop Type	1990		2005		2015	2016	2017	2018	2019
CH_4	**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	+		+		+	+	+	+	+
N_2O	**1**		**1**		**1**	**1**	**1**	**1**	**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**
NOₓ	**13**	**15**	**14**	**14**	**14**	**14**	**14**

+ Does not exceed 0.5 kt.
Note: Totals may not sum due to independent rounding.

Methodology

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

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

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

where,

Crop Production (CP)	= Annual production of crop, by state, kt crop production
Residue: Crop Ratio (RCR)	= Amount of residue produced per unit of crop production, kt residue/kt crop production
Dry Matter Fraction (DMF)	= Amount of dry matter per unit of residue biomass for a crop, kt residue dry matter/ kt residue biomass
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 matter
Fraction Burned (FB)	= Proportion of residue biomass consumed, unitless

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

Crop production data are available by state and year from USDA (2019) for twenty-one crops that are burned in the conterminous United States, including 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, and sugarbeets.[27] Crop area data are based on the 2015 National Resources Inventory (NRI) (USDA-NRCS 2018). In order to estimate total crop production, the crop yield data from USDA Quick Stats crop yields is multiplied by the NRI crop areas. The production data for the crop types are presented in Table 5-31. Alaska and Hawaii are not included in the current analysis, but there is a planned improvement to estimate residue burning emissions for these two states in a future Inventory.

The amount of elemental C or N released through oxidation of the crop residues is used in the following equation to estimate the amount of CH_4, CO, N_2O, and NO_x emissions (E_g, where g is the specific gas, i.e., CH_4, CO, N_2O, and NO_x) from the Field Burning of Agricultural Residues:

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

where,

Emission ratio (EF_g)	=	emission ratio by gas, g CH_4-C or CO-C/g C released, or g N_2O-N or NO_x-N/g N released
Conversion Factor (CF)	=	conversion by molecular weight ratio of CH_4-C to C (16/12), CO-C to C (28/12), N_2O-N to N (44/28), or NO_x-N to N (30/14)

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)
Mass of Fuel ($M_B \times C_f$)	=	IPCC (2006) default carbon fractions with fuel biomass consumption U.S.-

---

[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 (see Planned Improvements).

	Specific Values using NASS Statistics[28] (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 CH_4 and 53 percent lower emissions of N_2O 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 CH_4, CO, N_2O and NO_x, compared to IPCC (2006) approach that is based on dry matter rather than elemental composition.

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.

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

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

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

Crop	1990		2005		2013	2014
Maize	+%		+%		+%	+%
Rice	8%		8%		4%	6%
Wheat	1%		2%		2%	1%
Barley	1%		+%		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	+%		+%		+%	+%
Legume Hay	+%		+%		+%	+%
Peas	+%		+%		+%	+%
Sunflower	+%		+%		+%	+%
Tobacco	2%		2%		3%	3%
Vegetables	0%		+%		+%	+%
Chickpeas	0%		1%		0%	0%
Dry Beans	1%		1%		+%	+%
Lentils	0%		+%		+%	+%
Peanuts	3%		3%		3%	3%
Soybeans	+%		+%		1%	1%
Potatoes	+%		+%		+%	+%
Sugarbeets	+%		+%		+%	+%

+ Does not exceed 0.5 percent

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

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); Akintoye et al. (2005); values for AGR-N and BGR-N are from Grains.

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_2O: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).

For this Inventory, new activity data on the burned areas have not been analyzed for 2015 to 2019. To complete the emissions time series, a linear extrapolation of the trend is applied to estimate the emissions in the last five years of the inventory. Specifically, a linear regression model with autoregressive moving-average (ARMA) errors is used to estimate the trend in emissions over time from 1990 through 2014, and the trend is used to approximate the CH_4, N_2O, CO and NO_x for the last five years in the time series from 2015 to 2019 (Brockwell and Davis 2016). The Tier 2 method described previously will be applied to recalculate the emissions for the last five years in the time series (2015 to 2019) in a future Inventory.

Uncertainty and Time-Series Consistency

Emissions are estimated using a linear regression model with autoregressive moving-average (ARMA) errors for 2019. The linear regression ARMA model produced estimates of the upper and lower bounds to quantify uncertainty (Table 5-35), and the results are summarized in Table 5-35. Methane emissions from field burning of agricultural residues in 2019 are between 0.35 and 0.50 MMT CO_2 Eq. at a 95 percent confidence level. This indicates a range of 18 percent below and 18 percent above the 2019 emission estimate of 0.43 MMT CO_2 Eq. Nitrous oxide emissions are between 0.16 and 0.22 MMT CO_2 Eq., or approximately 17 percent below and 17 percent above the 2019 emission estimate of 0.19 MMT CO_2 Eq.

Table 5-35: Approach 2 Quantitative Uncertainty Estimates for CH_4 and N_2O Emissions from Field Burning of Agricultural Residues (MMT CO_2 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_2O	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.

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

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 the Introduction and Methodology sections.

QA/QC and Verification

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

Recalculations Discussion

Methodological recalculations are associated with two methodological revisions, a) removing the burning efficiency term and b) adopting the combustion efficiency value in IPCC/UNEP/OECD/IEA (1997) (See QA/QC and Verification Section for more information). As a result of these two revisions, the emissions increased on average across the time series by 10 percent and 9 percent for CH_4 and N_2O, respectively. The absolute increases in emissions are 0.4 MMT CO_2 Eq. and 0.2 MMT CO_2 Eq. for CH_4 and N_2O, respectively.

Planned Improvements

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

As previously noted in this chapter, remote sensing data were used in combination with a resource survey to estimate non-CO_2 emissions and these data did not allow identification of burning of sugarcane (see Annex 5). In addition, during the Public Review period of this current (1990 through 2019) Inventory, EPA received feedback on this category/crop type which provided average estimates of emissions of sugarcane burning found in academic literature. EPA plans to assess the information identified in feedback, as well as other available activity data, as part of future inventory improvements.

EWG
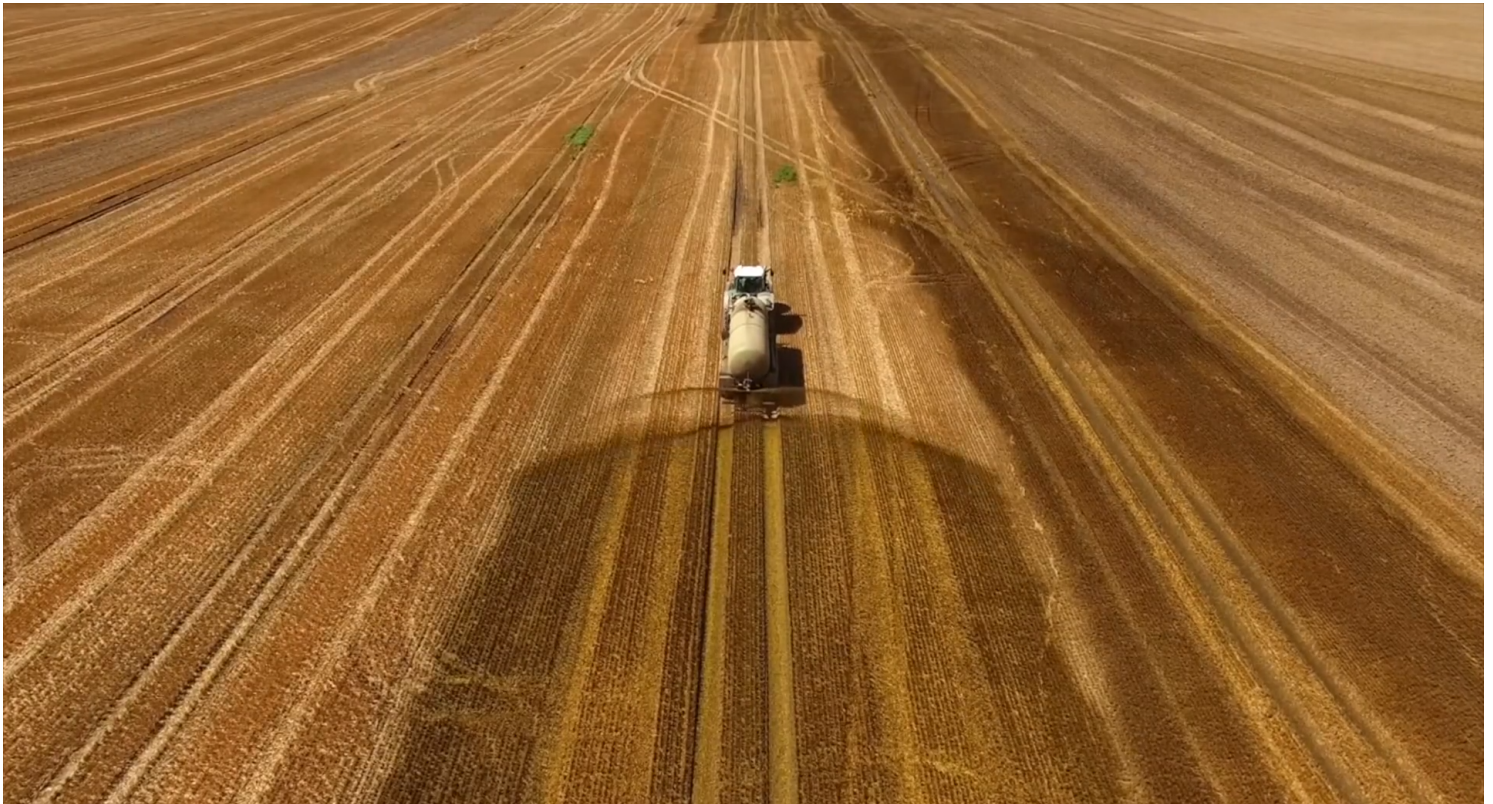
America's Nitrate Habit Is Costly and Dangerous

Prevention Is the Solution, But Voluntary Actions Fall Short

By Anne Weir Schechinger, Senior Analyst, Economics and Craig Cox, Senior Vice President for Agriculture and Natural Resources

TUESDAY, OCTOBER 2, 2018

America has a serious problem with nitrate contamination of drinking water – and it is most severe in the small communities that can least afford to fix it.

Nitrate, primarily from polluted farm runoff, contaminates the public water supplies of almost 1,700 communities nationwide at levels the National Cancer Institute says could increase the risk of cancer.

Federal data show that about two-thirds of those – 1,155 systems serving more than 3 million people – have no treatment systems to lower nitrate concentrations to safer levels. Of the systems without nitrate treatment, more than six out of 10 serve 500 or fewer people, and nearly nine out of 10 serve fewer than 3,300 people.

Removing nitrate from tap water is expensive. A city can spread the cost of treatment over a larger customer base. But in small communities, depending on the technology used, the increased cost of treatment per person could be hundreds or thousands of dollars a year.

It's much cheaper to keep nitrate out of drinking water in the first place than to remove it. But so-called volunteer approaches, like

making tax-funded payments to farmers who change where and how they farm, have not worked. Without aggressive, targeted and enforceable protection efforts, America's nitrate problem will get worse, and more Americans will be at risk of drinking contaminated water.

Nitrogen boosts crop yields, but pollutes water

88% of the communities at risk serve below 3,300 people.

Nitrate pollution comes from a variety of sources, but agriculture is by far the biggest contributor to the problem.

Millions of tons of commercial fertilizer and manure are applied on cropland across the U.S. every year. The nitrogen in fertilizer and manure boosts crop yields. But when it runs off as nitrate, it can pollute both surface water and groundwater, and its harmful impacts are felt far beyond the fields where it is applied.[1]

How does nitrate get into your water?

Water polluted with fertilizer and manure contains nitrate, a chemical that increases the risk of bladder cancer and developmental defects in children born to mothers who drink contaminated water during pregnancy, among many other health risks.

VIEW FULL GRAPHIC

Under the federal Safe Drinking Water Act , the legal limit for nitrate in drinking water is 10 parts per million, or ppm. This limit was set in 1962 to guard against blue baby syndrome, a potentially fatal condition that starves infants of oxygen if they ingest too much nitrate.[2]

But newer studies from the National Cancer Institute have found that drinking water containing 5 ppm of nitrate – half the legal limit – increases the risk of colon, kidney, ovarian and bladder cancers.[3] Other research shows more frequent birth defects in babies whose mothers consumed 5 ppm of nitrate daily from drinking water during pregnancy.[4]

According to data from the Environmental Protection Agency for 2014 and 2015, at least 1,155 communities with average nitrate levels at or above 5 ppm had no treatment system in place to reduce nitrate to safer levels.[5]

Though water systems lacking nitrate treatment are clustered in farming areas of the Midwest and California, they are found in a total of 43 states. The EPA classifies more than 60 percent of the water systems as "very small," serving 500 or fewer residents. Another one-fourth are classified as "small," serving between 501 and 3,300 people.

Treatment options are expensive

68% *of the communities with nitrate above 5ppm and no treatments options are located in 10 states.*

To remove nitrate from drinking water, the EPA recommends either an ion exchange or reverse osmosis treatment system.

Ion exchange systems contain a resin that removes nitrate as water passes through it. In reverse osmosis systems, pressurized water is pushed through a membrane that filters out nitrate and other contaminants. For both systems, the higher the concentration of nitrate, the higher the cost of removing it. Communities usually only use reverse osmosis if they need to remove other contaminants in addition to nitrate.

Treatment costs depend on the size of the system, the concentration of nitrate in the water and the concentration goal for the treatment. These variables make it harder to determine the treatment cost a given community may face. For example:

- In October of 2017, Hiawatha, Kan., began building a new water treatment plant that included an ion exchange system. Nitrate levels in Hiawatha had hit 11 ppm a few months before – it was one of several times the town has warned residents not to drink tap water.[6] The plant will cost the town of about 3,300 an estimated $3.5 million.

- In Chino, Calif., nitrate levels ranged from 9 ppm to 45 ppm before 2005. That year, the city, whose population then was

about 75,000, built an ion exchange system at a cost of $4.6 million.[7]

A direct comparison can't be made because population is not the same as the number of water customers, and contamination levels were different. But it's clear that Chino, the larger system, had a much lower cost per person than Hiawatha.

Researchers at the University of California at Davis came up with a method to make an informed estimate of the range of nitrate treatment costs per 1,000 gallons of water treated.[8] From there, EWG calculated that the cost of building and maintaining an ion exchange treatment system could add as much as $666 a year per person to the cost of providing drinking water in a very small community. A reverse osmosis system could add as much as $2,776 a year. (See the Appendix for details of the UC Davis researchers' and EWG's calculations.)

Water bills are normally calculated per household, not per person. Lacking data on the number of households per community, EWG used per person costs as an indicator of the burden the added cost of treatment could impose on communities.

Based on the UC Davis researchers' base estimates and EWG's analysis, the map below shows high-end cost estimates for each person in almost all communities that have nitrate contamination above 5 ppm, but that lack nitrate treatment systems. Costs could not be calculated for two water systems classified as "very large,"

with 100,000 or more residents. Click on a location to see the cost per person to install and operate an ion exchange system.

Figure 1. Nitrate contamination of tap water is found mostly in small or very small communities.
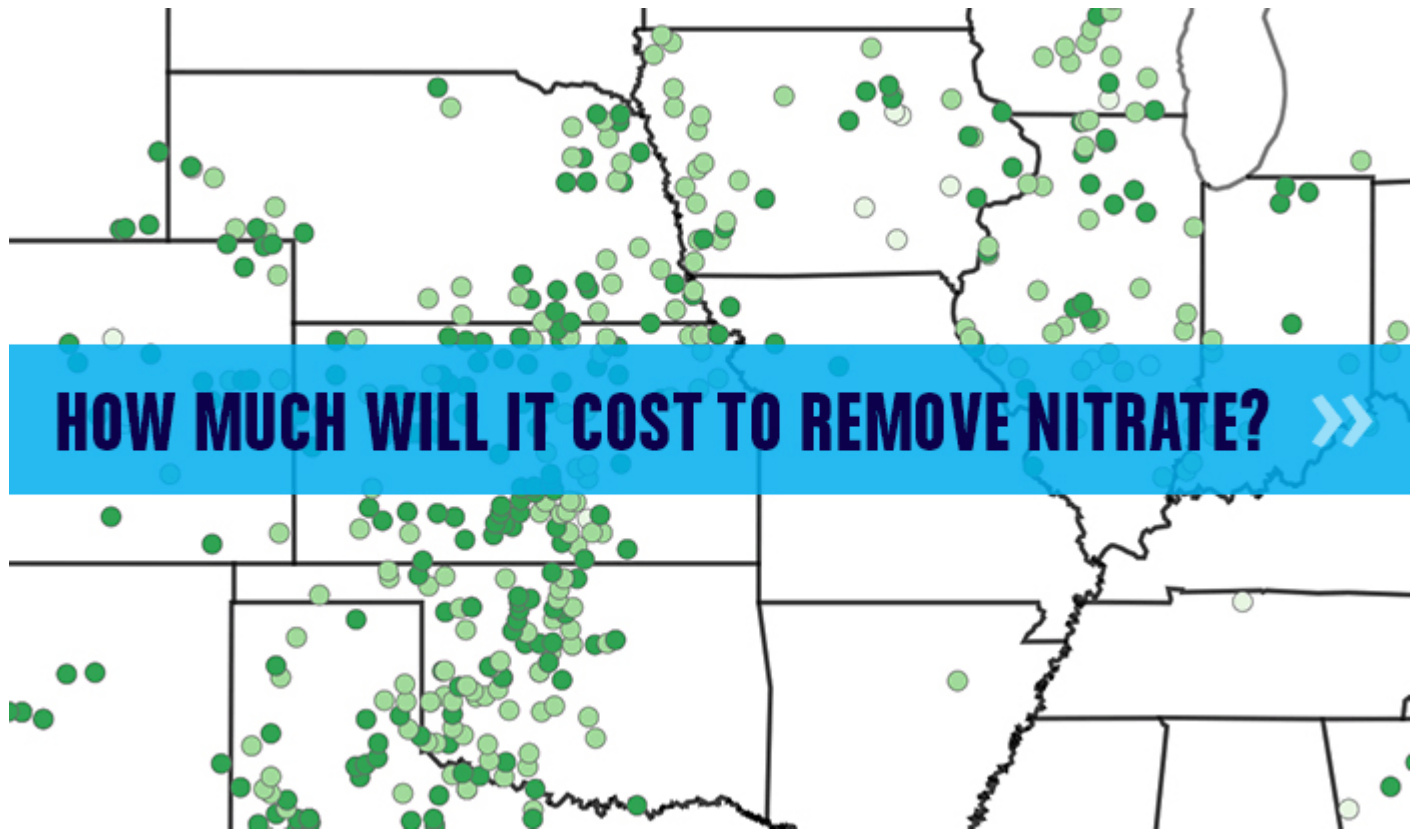
Source: EWG, from Environmental Protection Agency, SDWIS Data, and University of California, Davis, Technical Report 6: Drinking Water Treatment for Nitrate

The table below shows that the additional cost per person for an ion exchange system ranges widely. The low-end estimates range from $28 per person per year for the largest communities, to $90 per person per year for the smallest communities. High-end estimates range from $229 per person per year for large communities to $666 per person per year for very small communities.

Table 1: Cost per person per year to build and operate an ion exchange treatment system.

Community Size	Ion Exchange	
	Low Cost	High Cost
Very Small (<501)	$90	$666
Small (501-3,300)	$47	$378
Medium (3,301-10,000)	$48	$273
Large (10,001-100,000)	$28	$229

Source: EWG, from Environmental Protection Agency, SDWIS Data, and University of California, Davis, Technical Report 6: Drinking Water Treatment for Nitrate

If all U.S. communities with nitrate concentrations at or above 5 ppm in 2014 and 2015, but that lacked nitrate treatment, added ion exchange systems, the total extra cost would range from about $102 million a year to almost $765 million a year. If each of these communities without nitrate treatment chose to add a reverse osmosis system instead, the added cost could be as high as $1.47 billion a year.

Given the expense of a treatment system, some communities have tried other ways to reduce nitrate contamination: blending contaminated water with cleaner water, shutting off wells with seasonally high nitrate levels, digging new or deeper wells, or connecting to another water system that doesn't have a nitrate problem.

But those options are still expensive. To dig and operate a new well, or deepen an old well, a small water system could pay $46,000 to $330,000 a year.[9]

Where the problem is worst

Nitrate contaminates drinking water throughout the nation, but is highly concentrated in a few counties within a few states.

Of the communities with 5 ppm or more of nitrate but no treatment system, almost half are in just five states: California, Texas, Kansas, Pennsylvania and Oklahoma. Adding another five states – Washington, Arizona, Illinois, Nebraska and New York – covers almost 70 percent of such communities.

In 2014 and 2015, California had 142 communities with nitrate levels at or above 5 ppm, but with no nitrate treatment – the most in the nation. More than 40 percent of those communities were in Los Angeles, Kern and Tulare counties. In each of those counties, in 2016, the poverty rate was above the state average and household income levels were below the state average.

According to the UC Davis Center for Watershed Sciences, about 220,000 Californians in the Tulare Lake Basin and Salinas Valley get their drinking water from small water systems that are chronically plagued by nitrate contamination.[10] The estimated cost of long-term solutions to the state's nitrate problem in these areas is $34 million a year, which would add almost $150 per person to the yearly cost of drinking water for some of the poorest people in California.

The pattern of concentrated contamination in a few lower-income counties was similar or even worse in other top states. For

example, in Arizona, 79 percent of the communities without
nitrate treatment were in three counties, and in Pennsylvania,
three counties had almost half the untreated communities.
Looking at the top three counties for nitrate contamination in each
of the top states, more than two-thirds had household income
levels below the state average.

Private wells aren't covered by legal limits

Private wells don't have to comply with the 10 ppm legal limit for
nitrate. More than 43 million Americans get their drinking water
from private wells.[11] A 2009 study from the U.S. Geological Survey
found that 4 percent of all private wells have nitrate levels greater
than the legal limit,[12] but the problem is much worse in farming
areas.

Coates, Minn., is home to about 160 people. More than half of the
land in surrounding Dakota County is used to grow corn and
soybeans. Minnesota Department of Agriculture data show that
more than half of the private wells serving Coates may have
nitrate levels above the federal legal limit.[13]

The national number of private wells with nitrate levels above the
lower cancer threshold is unknown, but would be much higher. An
estimated 17 percent of the groundwater area in Nebraska, and 10
percent of that in California, is contaminated with nitrate above 5
ppm.[14]

Voluntary programs not enough

Keeping nitrate out of drinking water is much cheaper than removing it through treatment. Water treatment costs in communities with elevated nitrate concentrations are four to five times higher than those in areas without a problem.[15]

But except for the largest livestock operations, agriculture is largely exempt from federal Clean Water Act standards. Efforts to reduce polluted farm runoff have relied almost exclusively on so-called voluntary programs, which pay farmers to take steps to prevent pollution.

This approach has inherent weaknesses. The biggest flaw: Landowners who voluntarily start pollution-prevention practices can also stop them.

In the 1990s, Edgerton, Minn., with a population of fewer than 1,200, built a treatment plant to deal with rising nitrate levels, at a cost of about $3,500 per resident.[16] The city also created a voluntary plan to help farmers manage fertilizer and retire cropland. Nitrate levels in the water supply dropped by half and treatment costs were lowered.[17] But when crop prices jumped in 2005, farmers again planted crops on land that had been set aside. Nitrate levels increased and so did the cost of treatment.

Most voluntary conservation efforts do not actually help clean up drinking water. The Department of Agriculture spends billions of dollars a year on programs to encourage farmers to use

conservation practices. Yet EWG's research reveals a stunning underinvestment in the practices needed to protect drinking water in the places where it is most threatened.

For example, the Conservation Stewardship Program and the Environmental Quality Incentives Program pay farmers to plant cover crops, a key practice for preventing contamination of groundwater with nitrate. But almost 40 percent of the communities contaminated with nitrate at levels above 5 ppm are in counties where no farmers receive cover crop payments through those programs.

Clearly, there is a huge opportunity to focus these federal programs more tightly to head off the financial crisis rural communities may face to clean their drinking water.

Still, voluntary practices simply aren't a long-term solution. As conservation practices come and go, taxpayer dollars spent to encourage farmers to implement pollution-prevention practices often fail to achieve lasting progress. In the counties of the communities on our map, almost $30 million was spent in 2015 through the Environmental Quality Incentives Program to encourage better practices, yet serious nitrate pollution remains.[18]

The voluntary approach must be buttressed by setting mandated basic farm standards that target the most damaging practices. The standards should be tailored to different landscapes, watersheds and farming systems – but they must be required.

Voluntary programs could supplement these basic standards, to target the most effective pollution-prevention practices in the right places on the right farms. Voluntary programs are far more effective if practices are targeted to high-pollution areas.[19] New watershed-level tools like the Department of Agriculture's Agricultural Conservation Planning Framework use data to pinpoint locations where conservation practices will do the most good.[20]

There is still time to spare some communities with elevated levels of nitrate from facing the cost of building a treatment plant. Almost three-fourths of the communities without nitrate treatment on our map had average nitrate levels between 5 ppm and 7.5 ppm. Acting now to implement effective farm conservation practices could head off the need for huge capital expenditures down the line.

Combining common-sense standards with the investment of tax dollars to encourage additional steps is a better path to clean drinking water than putting the entire burden on drinking water utilities that serve small communities with limited resources.

Appendix

Share on Facebook   Share on Twitter

References

1. EWG, Interactive Graphic: Farm Pollution from Field to Tap. August 2017. Available at www.ewg.org/tapwater/troubled-water.php#.WyA-9KczqUk

2. EWG, Nitrate. EWG's Tap Water Database, July 2017. Available at www.ewg.org/tapwater/reviewed-nitrate.php#.WsJdr4hzaUk

3. The National Cancer Institute, Nitrate. February 2018. Available at progressreport.cancer.gov/prevention/nitrate

4. Jean D. Brender et al., Prenatal Nitrate Intake from Drinking Water and Selected Birth Defects in Offspring of Participants in the National Birth Defects Prevention Study. Environmental Health Perspectives, September 2013. Available at ehp.niehs.nih.gov/1206249/

5. Environmental Protection Agency, SDWIS Model. Envirofacts. Available at www.epa.gov/enviro/sdwis-model

6. City of Hiawatha Issues High Nitrate Warning. Hiawatha World, July 11, 2017. Available at www.hiawathaworldonline.com/news/city-of-hiawatha-issues-high-nitrate-warning/article_71208a3a-b98e-54a4-bc20-1c4b9bdd7a10.html

7. Vivian B. Jensen et al., Technical Report 6: Drinking Water Treatment for Nitrate. Center for Watershed Sciences, University of California, Davis, July 2012. Available at groundwaternitrate.ucdavis.edu/files/139107.pdf

8. Ibid.

9. Kristin Honeycutt et al., Technical Report 7: Alternative Water Supply Options for Nitrate Contamination. Center for Watershed Sciences, University of California, Davis, July 2012. Available at groundwaternitrate.ucdavis.edu/files/139108.pdf

10. University of California Davis Center for Watershed Sciences, Addressing Nitrate in California's Drinking Water. Available at watermanagement.ucdavis.edu/files/2214/5886/6964/Harter_et_al._2012_Addressing_Nitrate_in_CA_Drinking_Water.pdf

11. Leslie A. DeSimone et al., Quality of Water from Domestic Wells in Principal Aquifers of the United States, 1991–2004. United States Geological Survey,

March 2009. Available at pubs.usgs.gov/circ/circ1332/

12. Ibid.

13. Erin Adler, Tiny Dakota County Town Has a Big Problem: Polluted Water.
    Star Tribune, July 28, 2018. Available at www.startribune.com/tiny-coates-
    has-a-big-problem-polluted-water/489428861/

14. Environmental Protection Agency, Estimated Nitrate Concentrations in
    Groundwater Used for Drinking. Available at www.epa.gov/nutrient-policy-
    data/estimated-nitrate-concentrations-groundwater-used-drinking

15. Minnesota Department of Health and Minnesota Department of Agriculture,
    Nitrate Contamination- What is the Cost? 2004. Available at
    http://www.nesc.wvu.edu/ecommerce/products/DW_PublicEducation/DWFSP
    E347DL.pdf

16. Minnesota Drinking Water Annual Report for 2016.

17. Minnesota Drinking Water Annual Report for 2014.

18. EWG, Environmental Quality Incentives Program (EQIP) in the United States.
    EWG's Conservation Database. Available at conservation.ewg.org/eqip.php?
    fips=00000&regionname=theUnitedStates

19. Iowa Department of Agriculture and Land Stewardship, Iowa Department of
    Natural Resources and Iowa State University College of Agriculture and Life
    Sciences, Iowa Nutrient Reduction Strategy: A Science and Technology-Based
    Framework to Assess and Reduce Nutrients to Iowa Waters and the Gulf of
    Mexico. Updated December 2017. Available at
    www.nutrientstrategy.iastate.edu/documents

20. North Central Region Water Network, The Agricultural Conservation
    Planning Framework (ACPF): A Watershed Planning Tool. United States
    Department of Agriculture, 2018. Available at northcentralwater.org/acpf/

Other EWG Resources

EWG's Tap Water Database

Children's Health Initiative

EWG's Healthy Living App

Guide to Healthy Cleaning

EWG's Food Scores

EWG's Skin Deep

Safe Cell Phone Use

Cancer Prevention Within Reach

EWG.org | EWG's Guide to Sunscreens | EWG's Food Scores | EWG's Guide to Healthy Cleaning | EWG's Shopper's Guide to Pesticides in Produce™

REPORT

CAFOS: WHAT WE DON'T KNOW IS HURTING US

About NRDC

The Natural Resources Defense Council is an international nonprofit environmental organization with more than 3 million members and online activists. Since 1970, our lawyers, scientists, and other environmental specialists have worked to protect the world's natural resources, public health, and the environment. NRDC has offices in New York City, Washington, D.C., Los Angeles, San Francisco, Chicago, Montana, and Beijing. Visit us at nrdc.org.

Table of Contents

Introduction

Corporate livestock facilities, known as concentrated animal feeding operations (CAFOs), threaten the health of communities and pollute the air and water.[1] CAFO-related pollution is more than a nuisance; it is dangerous. Manure from CAFOs contains more than 150 pathogens that have the potential to contaminate water supplies, while fumes and particulate matter elevate rates of asthma, lung disease, and bronchitis among farm workers and people living nearby.[2,3] Nitrates from animal manure poison drinking water sources and contribute to epic dead zones in sensitive aquatic habitats.[4] Confining large numbers of animals in close proximity requires routine antibiotic regimens, and this, in turn, exacerbates the global crisis of antibiotic resistance.[5]

The animal agriculture industry comprises hundreds of thousands of individual animal feeding operations (AFOs)—facilities that raise animals in confinement.[6] CAFOs are a subset of AFOs distinguished by their large size or their designation as significant polluters of surface waters.[7] CAFOs have become more prevalent as part of a decades-long trend of corporate consolidation and vertical integration across the livestock industry. Today the pork, broiler chicken, and beef sectors are all "highly concentrated" in the hands of three or four companies that exercise enormous market power—and control the practices used across these facilities.[8]

This power extends to the regulatory sphere as well. Although the Clean Water Act requires the U.S. Environmental Protection Agency (EPA) to regulate CAFOs, pressure from the corporate livestock industry led to a series of dubious administrative and court decisions in the 2000s that raised serious concerns about the EPA's willingness and capacity to effectively carry out this responsibility.[9]

A decade ago, the nonpartisan Government Accountability Office concluded that the EPA could not fulfill its regulatory duties under the Clean Water Act without accurate and facility-specific information about CAFOs.[10] The EPA, for its part, has admitted that "unlike many other point source industries, the EPA does not have facility-specific information for all CAFOs in the United States."[11]

This report and the accompanying resources are the culmination of NRDC's decadelong effort to understand just how much—or how little—the EPA knows about the

animal agriculture industry that it is supposed to regulate. Fearing that the EPA lacked even basic information about CAFOs—their location, how many animals they confine, how much waste they produce, and how they dispose of that waste—in 2010 NRDC and other environmental organizations reached an agreement with the EPA, which was designed to collect the missing data needed to start cleaning up our waterways and protecting public health from CAFO pollution.[12] When that effort stalled under industry pressure in 2012, NRDC turned to compiling available data using Freedom of Information Act requests and public databases (see "The Search for CAFO Data").

THE SEARCH FOR CAFO DATA

When the EPA's effort to survey CAFOs stalled under industry pressure in 2012, the agency claimed it could get the information it needed from existing sources.[13] Skeptical of this claim, NRDC filed Freedom of Information Act (FOIA) requests to discover how much information regulators actually had.

To comply with NRDC's FOIA request, in early 2013 the EPA released records to NRDC on CAFOs, including information that the EPA had collected from individual states.[14] After the EPA's compliance, industrial livestock interests went on the attack. A lawsuit filed by the American Farm Bureau Federation and the National Pork Producers Council in the summer of 2013 challenged the EPA's decision to release this public information.[15] The corporate livestock industry also sought to shut down the release of any state-gathered information received by the EPA since its initial response, and to prevent the EPA from releasing data in response to future FOIA requests for updated information.

A federal district court dismissed the case in 2015 after ruling that the industry groups did not have standing.[16] On appeal in 2016, however, the Eighth Circuit sided with agribusiness, the court finding that certain information about animal agriculture facility operators fell under a FOIA provision exempting the release of "personnel and medical files and similar files" if disclosure "would constitute a clearly unwarranted invasion of personal privacy."[17] The circuit court sent the case back to the district court to decide whether the EPA had the authority to release this information at its own discretion. The parties settled before the case was resolved, as discussed below.[18]

While the livestock industry was fighting to keep important information under wraps, NRDC undertook the monumental task of compiling and organizing the revised data it had received from the EPA and supplementing that information with additional information gleaned from publicly available databases on state websites.[19] The process of developing the database ran from 2013 until 2016, spanning the same period as the litigation over the FOIA requests. Although the database was completed in 2016, NRDC held out hope that it would be able to update the database using more recent and complete data released by the EPA after the litigation concluded. However, in 2017, under former EPA administrator Scott Pruitt, the EPA signed a settlement agreement with industry groups severely limiting the scope of any future information releases.

As a result, some of the data used in this report date to 2013 or even earlier, depending on when they were gathered. In many cases they are old and incomplete, but they are the best that NRDC is aware of—and that is exactly the problem. Unfortunately, given the Eighth Circuit's decision and the subsequent settlement agreement, the data from the EPA, despite their numerous limitations, are likely more comprehensive than anything the agency will release in the foreseeable future. After years of uncertainty about the EPA's intentions, NRDC decided to move forward in the knowledge that it already had the best data it could get.

Analysis of these records shows pervasive gaps in the data about the size, type, and location of CAFOs and the pollution control measures they are using. NRDC found at least some data on 7,595 CAFOs in 40 states. That leaves more than half of the 17,000-plus CAFOs the EPA estimated to exist in 2012 completely unaccounted for in the agency's own data.[20] Further clouding the situation, NRDC's analysis suggests that the EPA may have significantly underestimated the number of CAFOs and that the true number of undocumented operations is even higher.

Drilling into the existing data reveals stark disparities among the states. NRDC found no information for nine states that the EPA determined housed CAFOs. Where data were available, both the quantity and nature of the data were highly variable from state to state. States are responsible for collecting information and reporting it to the EPA, but there is no standardized collection or reporting requirement. Moreover, many states that appear to have an accurate count of the number of CAFOs nonetheless lack critical data about operations' size, permit status, location, method of storing manure, animal type, and ownership. Regulators and the public need this information to protect communities from CAFO pollution and hold the agriculture industry accountable for its impacts under the law.

States have the opportunity to fill the CAFO data void. Toward that end, NRDC has created a Clean Water Act permit based on some of the better approaches in several states and lessons learned from developing this report. The permit would allow states to collect and share critical data on the CAFOs within their borders. The permit would also improve public accountability by making information more available. It would require all but the smallest AFO facilities to obtain a permit, inform neighbors about their operations, and disclose their location and nutrient management plans to the state, the EPA, and the public. NRDC's permit would clarify that every CAFO's nutrient management plan—which enumerates the permit holder's responsibilities for handling animal waste—is a public record that communities have a right to understand and enforce. Finally, the permit would expand notice requirements so that neighbors are fully informed when a new CAFO wants to open down the road, or when existing operations are planning to expand, thereby increasing their production of waste.

It is time for states to step up where the federal government has failed. This report shows how the EPA's acquiescence to industry demands leaves the agency in the position of knowing little about the facilities it is supposed to hold accountable. Given the dangers posed by CAFO pollution, what the EPA doesn't know can hurt all of us.

THE EXISTING REGULATORY REGIME

The Clean Water Act (CWA) is the primary law regulating the pollution of U.S. waters, including rivers, lakes, and wetlands.[24] The law, passed in 1972, requires that all "point sources" discharging pollutants be subject to permitting requirements under the National Pollutant Discharge Elimination System (NPDES).[25] These permitting requirements set strict discharge limits intended to help eliminate pollution in waterways by 1985.[26] The Act is enforced primarily by the EPA and through cooperative federalism with the states.[27]

Since the original Act, the definition of "point source" has included "concentrated animal feeding operation[s]."[28] Despite this very clear language, actual regulation of CAFOs under the CWA started slowly. In 1974, the EPA issued water-quality protection rules called effluent limitation guidelines (ELGs) for feedlots, which are a specific subset of AFOs used for finishing livestock prior to slaughter.[29] Substantial exemptions excluded smaller facilities, those that discharged only under extreme storm conditions, and chicken operations with dry-manure handling systems.[30] Though all exemptions for point sources were struck down soon after in federal court, the newly formed EPA designated CAFOs as a "low enforcement priority" in light of other pressing environmental challenges facing the new agency; regulation was largely left to the states, with limited federal oversight.[31] This arrangement left certain states with standards that fell well below the requirements imposed by federal law. With little oversight from the EPA, some states with high numbers of CAFOs simply ignored their duty to issue permits well into the 2000s.[32]

Congress exacerbated this passive regulatory approach when, in 1987, it amended the CWA to exempt "agricultural stormwater discharges" from the statutory definition of a point source.[33] A stormwater discharge occurs when rain washes contaminants off a surface and into waterways. Since it is common to apply CAFO manure directly to the surface of agricultural fields, the amendment made it harder to regulate industrial livestock operations. Despite evidence that the amendment was not intended to impact how preexisting regulations applied to CAFOs, these operations began claiming coverage under the exemption.[34]

The EPA addressed this and other issues in 2003 when, in response to a lawsuit brought by NRDC and Public Citizen, the agency finalized its first comprehensive rule governing CAFO regulation under the CWA.[35] This rule—the high-water mark in federal CAFO regulation—obligated all CAFOs to apply for an NPDES permit unless they could demonstrate that they had "no potential to discharge" pollution. However, a federal court disagreed that the CWA supported such an obligation and struck down that element of the rule.[36] (The rule also included standards relating to manure handling, nutrient management plans, and facility design.) In response, the EPA released a new rule in 2008 that limited the permitting obligation to CAFOs that "propose to discharge," but even that circumscribed requirement was struck down.[37] The EPA's 2012 CAFO rule then cemented the loophole by officially exempting from permitting requirements any CAFO that was not actively discharging.[38]

The 2003 rule contained an important additional loophole as well. It specified that discharges from CAFOs qualified as exempt "agricultural stormwater" if they were precipitation-related discharges from the land application area (rather than the production area) and if manure had been "properly" applied.[39] A 2013 district court decision made matters worse. It held that even discharges from the production area were covered by the exemption, contradicting decades of case law.[40] Such discharges include manure, litter, and feathers blown out of poultry houses by a ventilation system. The EPA appealed the decision but later dropped its appeal while maintaining the position that it could require a permit for such discharges in the future.[41]

CAFOs AND THEIR POLLUTION

A relentless tide of consolidation has swept across animal agriculture over the past half century.[42] As technological advances allowed farmers to raise huge numbers of fast-growing animals in close quarters, large industrial operations proliferated while small farms struggled and generally failed to compete on the industry's new terms.[43] This rise in consolidation and the growth of CAFOs go hand in hand.[44]

CAFOs are tailored to produce vast quantities of meat, eggs, and dairy that can be sold to consumers at low prices. For example, Americans pay only about $1.50 for a pound

of fresh chicken, something made possible by facilities that raise tens of thousands of birds in a single building and take advantage of breeding and feeding refinements that allow chickens to reach slaughter weight in about six weeks.[45,46] However, these low consumer prices hide the true costs of production.[47] Supermarket prices do not account for the fact that the public heavily subsidizes CAFOs, including manure management and feed costs, through the United States' vast system of agricultural price supports and subsidies.[48] And supermarket prices do not reflect the environmental and human health costs of CAFOs, which can threaten neighboring communities, pollute waterways, and exacerbate antibiotic resistance. Neither the market nor existing safeguards compel CAFOs or the big businesses that operate them to cover those costs.[49]

MANURE MANAGEMENT

A CAFO raising 82,000 laying hens can produce 2,800 tons of manure in a year, and a single CAFO raising 10,000-head of beef cattle can produce up to 117,000 tons of manure annually.[50] How animal manure is handled, stored, and disposed of affects human and environmental health, but practices vary widely across facilities. "Handling" is the process of getting manure from the facility floor into some kind of storage container, where it sits until it is "disposed." Dry manure handling systems, which are commonly used in poultry operations, release substantial amounts of nitrous oxide and particulate matter into the air.[51] In contrast, wet handling systems, most commonly found on swine and dairy farms, use water to wash manure into storage structures or lagoons, where it releases methane and hydrogen sulfide as it is anaerobically digested.[52] Wet systems can also attract and breed insects, contributing to mosquito and fly problems in communities, and research indicates that leakage from wet storage lagoons can contribute to increased concentrations of nitrate, ammonium, bacteria, and other contaminants in groundwater.[53]

Storing such vast quantities of manure presents one set of problems; disposing of it is another matter entirely. There are no sewage treatment requirements for animal manure, in stark contrast to the requirements that apply to human waste management, and the majority of manure from CAFOs is never treated.[54] Instead, CAFO waste is generally disposed of on land in solid, slurry, or liquid form, depending on the type of animal and the practices of the individual facility.[55] There are several ways to dispose of manure on land, but among the least expensive and most common is applying it directly on top of soil as fertilizer.[56] Manure is an important source of natural fertilizer, but given the immense volumes of manure generated at CAFOs, facilities often apply it in concentrations that far exceed the nutrient requirement of their crops or the holding

capacity of the underlying soil.[57] This nutrient overloading commonly causes contaminants to leach from manure into groundwater, or to run off into streams and rivers after a heavy rain.[58]

Although Clean Water Act regulations place some restrictions on manure application within the boundaries of the CAFOs themselves, these do not extend to the application of manure that is shipped off-site.[59] And in some areas of intensive CAFO farming, like Maryland's Eastern Shore, facilities ship up to 85 percent of their manure off-site where federal rules no longer regulate its application to open fields.[60]

MARYLAND'S MANURE PROBLEM

A 2014 analysis by the Environmental Integrity Project found that poultry operations on Maryland's Eastern Shore spread three times more manure on agricultural land than crops planted there could reasonably use.[61] The study examined records from 62 poultry operations across five counties—a fraction of the 404 poultry operations that raised nearly 218 million chickens in those counties in 2012.[62] These 62 operations alone applied 482,000 pounds of phosphorus, contained in chicken manure, in a single year.[63] The excess nutrients run off into nearby streams and then into the ecologically fragile Chesapeake Bay, where the phosphorus and nitrogen fuel algal outbreaks that can make the water uninhabitable for fish and blue crabs.[64] Outbreaks of cyanobacteria, in particular, have increased in the Chesapeake Bay, and these can damage not only ecosystems but also human health.[65]

The pollution, from CAFOs as well as other point sources, grew so bad that in 2010, after decades of failed efforts to clean up the bay and the surface waters that feed into it, the EPA put the Chesapeake Bay on a "historic and comprehensive 'pollution diet.'"[66] The plan set pollution limits, known as a total maximum daily loads (TMDLs), on nitrogen, phosphorus, and sediment within the Chesapeake Bay Watershed.[67] There are signs that the diet is working: The overall health of the Chesapeake Bay ecosystem is the best it has been in decades, improving from "moderately poor" in 2010 to "moderate" in 2017.[68] However, meeting the EPA's goal of restoring full ecosystem health by 2025 will require further steps to rein in CAFO pollution across the Chesapeake Bay Watershed.

IMPACT ON COMMUNITIES

From what we know, CAFOs tend to be located near populations that lack the political clout to successfully oppose their construction.[69] Thus, low-income and minority populations suffer disproportionately from proximity to CAFO pollution and waste.[70] Localized harms include impaired drinking water, antibiotic resistance, air pollution, and waste spills and associated fish kills.

IMPAIRED DRINKING WATER

CAFOs can pollute local water on a massive scale. According to an analysis of EPA records, "the drinking water of millions of Americans living in or near farming communities across the country is contaminated by dangerous amounts of nitrates and coliform bacteria from fertilizer and manure widely used in agriculture."[71] Tens of thousands of times, nitrate and coliform levels have exceeded the federal limits on contaminants in public water systems set by the Safe Drinking Water Act (SDWA).[72] CAFOs contribute much of this contamination: Both the EPA and the National Cancer Institute trace human health hazards associated with elevated nitrate levels in drinking water to animal waste.[73]

SDWA regulations limit nitrate concentrations to 10 parts per million in municipal drinking water.[74] Regulators chose this limit to protect infants from blue baby syndrome, a potentially fatal condition that prevents blood from carrying sufficient oxygen.[75] However, nitrate levels at even half the legal limit increase risk for colon, kidney, ovarian, and bladder cancers in otherwise healthy adults.[76] Rural residents located near CAFOs are particularly vulnerable because many rural residents draw water from private wells, which are not covered by the SDWA.[77] About 4 percent of private wells in the United States have nitrate levels above the SDWA standards, and that number is much higher in farming communities.[78]

In addition to nitrates, animal manure is a significant source of pathogens that flow or leach into drinking water sources. More than 150 pathogens are found in animal manure.[79] Six of these, *Campylobacter*, *Salmonella*, *Listeria*, *E. coli 0157:H7*, *Cryptosporidium*, and *Giardia*, account for 90 percent of all human foodborne and waterborne diseases.[80] Though healthy people who are exposed to these pathogens often recover after a bout of diarrhea, more vulnerable groups like infants, pregnant women, the elderly, and those with weak immune systems are at risk of severe illness and death.[81]

ANTIBIOTIC RESISTANCE

As industrial animal facilities increasingly rely on antibiotic drugs to suppress disease in their facilities, microbes begin to develop immunity—meaning that the drugs that we depend on to treat serious human infections become less effective. Experts from the World Health Organization (WHO) and the Centers for Disease Control and Prevention (CDC) identify antibiotic resistance as one of the top public health threats in the world.[90] According to the WHO, "Antimicrobial resistance threatens the effective prevention and treatment of an ever-increasing range of infections caused by bacteria, parasites, viruses and fungi."[91]

In 2017, producers fed their cattle, pigs, and poultry over 12.3 million pounds of antibiotics that are "medically important," meaning they are extremely similar or identical to antibiotics that humans depend on to combat infection.[92] CAFOs routinely feed low doses of antibiotics to animals that are not yet sick in order to prevent disease, a need that arises from the dirty and stressful conditions of close confinement.[93] While the antibiotics will kill some bacteria, resistant bacteria remain in the gut, multiply, and are excreted in manure—along with substantial amounts of the antibiotics themselves—and enter the surrounding community's air, soil, and water.[94] When antibiotics-laden runoff from CAFOs spills into the surrounding environment, these drugs further promote antibiotic resistance and can also be toxic to soil microflora and fauna.[95] The problem stretches to meat products as well: A 2015 analysis found that antibiotic-resistant bacteria were present in approximately 35 to 80 percent of raw meat from the supermarket.[96]

AIR POLLUTION

CAFOs can also threaten local air quality by releasing ammonia, hydrogen sulfide, and other gases and particulate material into the atmosphere.[97] For instance,

---

GROUNDWATER IN KEWAUNEE COUNTY, WISCONSIN

In Kewaunee County, Wisconsin, which has experienced a substantial increase in the number of CAFOs over the past two decades, a 2017 study found fecal microbes in 60 percent of sampled wells.[82] Researchers estimated that one of those microbes, *cryptosporidium*, was infecting 140 county residents each year.[83] *Cryptosporidium* symptoms include diarrhea, dehydration, stomach cramps, and vomiting.[84] In October 2014, local environmental groups petitioned the EPA to use its powers under the Safe Drinking Water Act to investigate the sources of the groundwater contamination—which in some cases was turning people's tap water brown—and to order alternative water supplies be made available to Kewaunee County residents.[85,86] In May 2017, under continuous pressure from residents and advocates, the state began supplying bottled water to residents whose wells tested positive for bacteria from livestock manure.[87] In a ruling on a separate petition filed by local residents opposing removal of certain clean water protections from a state permit issued to a dairy CAFO, Administrative Law Judge Jeffrey Boldt noted that "the proliferation of contaminated wells represents a massive regulatory failure to protect groundwater" by the Wisconsin Department of Natural Resources, the state permitting authority.[88] Since the ruling, the petition has been wending its way through the appeals process; in January 2019 the case reached the Wisconsin Supreme Court, which must now decide whether to consider the state's claim that it lacks authority to implement commonsense requirements such as groundwater monitoring at manure-spreading sites.[89]

it has been estimated that the average broiler chicken emits between 0.27 and 0.54 grams of ammonia each day in its manure.[98] These seemingly small numbers add up: An average broiler facility raising 90,000 birds at a time may release more than 15 tons of ammonia a year, causing respiratory problems and chronic lung disease as well as chemical burns to the respiratory tract, skin, and eyes of nearby residents.[99,100] Ammonia, in particular, also contributes to algal outbreaks in nearby bodies of water.[101] Algae outbreaks disrupt oxygen availability, creating dead zones devoid of aquatic and marine life.[102] In some cases, the algal blooms are themselves toxic.[103]

The release of hydrogen sulfide and toxic particulate matter can be equally damaging, with devastating impacts on workers and neighboring residents, particularly children.[104] As many as 30 percent of industrial livestock farmworkers experience occupation-related asthma and chronic bronchitis.[105] Asthma rates are higher among children living near animal operations: One study found that 46 percent of children living on large swine farms (500-plus swine) had asthma, compared with 26 percent of those living on farms without livestock.[106] Beyond asthma, lengthy exposure to the particulate matter and gases can have other deleterious effects including lung disease and heart attacks.[107] Odors from these emissions can also harm a community's quality of life, preventing people from spending time outside and even, according to some studies, impacting mental health.[108]

SMITHFIELD FOODS AND NORTH CAROLINA

The CAFO construction boom began in North Carolina in the 1970s and took off in the 1980s after a series of state laws eliminated sales tax on swine farm equipment and preempted local officials from using zoning powers to limit swine facility construction.[109] Today, North Carolina's approximately 2,500 permitted swine operations raise about nine million animals at any given time, with some individual facilities capable of housing up to 60,000.[110] A series of cases brought by neighbors against Smithfield Foods, a swine producer with operations in North Carolina, allege that the facilities' excessive odors, flies, and noise from truck traffic interfere with nearby residents' use and enjoyment of their property.[111] The CAFOs at the center of the lawsuit are disproportionately located in low-income black communities that lacked the political power to resist their construction.[112] Thus far, juries in four trials have awarded 26 plaintiffs more than half a billion dollars in combined damages, and 22 suits involving another 500 or so residents are still pending.[113] However, this important moment in holding facilities accountable looks to be short-lived: A judge has already substantially decreased several of the awards based on a state law limiting punitive damages.[114] To further shield Smithfield, the North Carolina legislature recently passed legislation—over the governor's veto—limiting plaintiffs' ability to bring similar suits in the future.[115]

SPILLS AND FISH KILLS

Although CAFOs can cause significant environmental risks even when all goes according to plan, unforeseen events or errors can compound the problems, causing manure to leak or spill into surrounding ecosystems. These leaks and spills can have a number of causes, including accidents during transport of manure, willful discharges made in an effort to avoid regulatory requirements, leaking lagoons, and rainstorms that cause storage containers or lagoons to overflow or burst. The sheer scope of the spillage can be staggering: The Wisconsin Department of Natural Resources determined that more than three million gallons of manure were spilled in the state in 2013 and 2014.[116] With so much waste in one place, a single mistake or a single unethical operator can wreak havoc on neighboring communities.

CAFO spills are a common cause of fish kills. In 2017, three large spills killed close to 67,000 fish in Ohio.[117] Other states have dealt with similar issues: In Minnesota, which experiences an average of 20 manure spills each year, approximately 700,000 fish died in a single incident when hog waste washed into Beaver Creek.[118] Spilled manure from a dairy CAFO in Lewis County, New York, killed 375,000 fish.[119] These events can devastate local wildlife and ecosystems.

FISH KILLS IN ILLINOIS

In July 2012, manure from a hog confinement facility in Illinois leaked from fields into a creek, where it reportedly polluted more than 20 miles of the waterway and killed nearly 150,000 fish and 18,000 freshwater mussels.[120] Two years later, biologists could not locate any examples of nine fish species that had been in the creek prior to the discharge.[121] According to one man whose family fished in the creek, "It looked like ink, the water. It had fish all over the place, dead. It wasn't fit for nothing. Not even a wild animal could drink out of it."[122]

MONITORING

As described in more detail below, CAFOs are formally regulated for their environmental impact by the EPA, which generally delegates its responsibilities to state and local environmental agencies.[123] The EPA and state regulators conduct facility inspections, which serve as the primary monitoring mechanism to uncover violations. If a facility is out of compliance, regulators can initiate an enforcement action. However, inspections and enforcement actions occur remarkably infrequently. In the 2017 fiscal year, the EPA conducted 125 inspections and concluded 18 enforcement actions.[124] For context, the EPA estimated in 2017 that there were 19,961 CAFOs in the United States, which means that the EPA inspected 0.6 percent of all facilities.[125] Between 2008 and 2013,

the EPA averaged about 260 inspections annually with an average of 60 enforcement actions across a similar number of CAFOs; remarkably, this still-low figure reflects a period of time when CAFOs were identified as a national enforcement priority.[126]

These figures do not account for inspections conducted by state agencies, which run their own inspection programs.[127] However, the lack of consistent and complete data at the state level raises serious questions about how comprehensively states are keeping tabs on the CAFOs within their borders. The nonpartisan Government Accountability Office (GAO) attributes these state failures, at least in part, to poor oversight by the EPA.[128] Although states must submit annual updates to the EPA on their CAFO permitting programs, reporting standards are insufficient to ensure that the EPA has site-specific data to hold facilities or their state regulators accountable. The GAO concluded that the EPA cannot fulfill its regulatory duties under the Clean Water Act without accurate and facility-specific information about CAFOs.[129] The EPA, for its part, admits that in contrast to its oversight of other major emitting industries, it does not have comprehensive, facility-level information for CAFOs.[130]

THE CAFO DATA VOID

Despite the ongoing harms caused by CAFO pollution, no federal agency collects and maintains the kind of comprehensive data about CAFO size, location, and operations that would be prerequisite to an effective environmental enforcement strategy.[131] Instead, regulators must rely on the patchwork of state-level information that forms the basis for this report. Moreover, regulators rely heavily on self-reporting: Although some proactive states impose a stricter duty on operators to apply for permits, under federal law CAFOs are obligated to apply for a permit only if they discharge.[132] This means the government must demonstrate that a CAFO is discharging into waters if it wants to compel the facility to apply for a permit—a difficult task, especially if the EPA does not even know that the CAFO exists. Given the costs of permitting and the relatively low likelihood of an enforcement action, it is not surprising that many CAFOs operate without a NPDES permit.[133] And even where permits are formally requested, permit standards in some states lag well behind those of the EPA.[134]

NRDC and other groups have repeatedly pressed the EPA to take steps to fill gaps in their information. In 2008, federal officials acknowledged that:

> ...[the] EPA does not have data on the number and location of CAFOs nationwide and the amount of discharges from these operations. Without this information and data on how pollutant concentrations vary by type of operation, it is difficult to estimate the actual discharges occurring and to assess the extent to which CAFOs may be contributing to water pollution.[135]

The EPA cannot currently execute its congressionally imposed duty to protect human health and the environment. Specifically, the EPA's blind spots make it difficult or impossible to evaluate the effectiveness of the NPDES program, identify and permit CAFOs that discharge, promote best management practices, locate and address sources of water quality impairment, estimate the amount of pollution entering water bodies, and efficiently target resources at areas of concern.[136]

The EPA initially responded to these findings, along with pressure from NRDC and other groups, by agreeing to circumvent the states and directly survey the industry. In 2011 the agency proposed the "CAFO Reporting Rule" or "308 Rule."[137] The proposed rule would have leveraged the EPA's authority under Section 308 of the Clean Water Act to require all CAFOs to report certain data.[138] Specifically, the EPA proposed to collect the following basic information about animal operations:

- Name and address of the owner/operator
- Facility location
- NPDES permit status
- Animal type and number
- Location and total number of acres under control of the owner available for land application of manure

The proposed inventory would have replaced the EPA's inconsistent and patchy information with a much-needed national database of animal facilities. At the same time, the proposed information collection was extremely modest; the EPA decided not to collect numerous pieces of data urged by environmental and public health advocates—including the type and capacity of available manure storage, the presence of a nutrient management plan (or lack thereof), and information on off-site transfer of manure.[139] The agency also proposed to limit data collection to once every 10 years. However, under industry pressure, the EPA withdrew the proposed rule in 2012.[140] Instead of collecting the information directly, the EPA stated it would pursue "an approach that relies on a range of existing sources of information, other regulations, and other programs at the federal, state, and local level to gather basic information about CAFOs."[141]

In scuttling the survey, the EPA claimed that it could obtain the missing data by working more closely with states, erroneously asserting that the states and other government entities already have that data.[142] Unfortunately, this approach failed to produce the information that the EPA needs to assess and regulate CAFOs nationwide. Rather, the quantity and quality of data on industrial livestock operations has historically varied widely across states, with some states leading and other lagging.[143] Concerned that the EPA's new approach would fail to address data gaps and inconsistencies, NRDC set out to assess what information was in fact available about these operations across the country.

RESEARCH METHODOLOGY

Following the EPA's failure to act, NRDC attempted to collect publicly available data to better understand just how much (or how little) information the EPA had about CAFOs in the United States. By recreating the EPA's database and then comparing it with the EPA and USDA estimates of the total number of industrial animal facilities, NRDC sought to determine whether existing facility-specific data were sufficient for informing and protecting communities or holding the agriculture industry accountable for its pollution impacts.

NRDC created a database using facility-specific CAFO data obtained from four sources between 2012 and 2015. First, we obtained data from the EPA after filing FOIA requests in 2012. In response to these requests, the EPA provided the CAFO data it had collected directly from the states.[144] Second, NRDC searched the EPA's Permit Compliance System (PCS) and Integrated Compliance Information System (ICIS) for all animal facilities.[145] Data were collected for all animal agricultural facilities in every state with active NPDES permits. Facilities that did not meet the regulatory definition of a CAFO (such as aquaculture operations) were excluded. Third, we conducted an independent search of publicly available data on state permitting websites.[146] These searches took place in the spring of 2015. NRDC extracted all readily available information from these websites on AFOs, regardless of whether they were NPDES permitted, and added this information to the database.[147] Finally, we added data for all NPDES-permitted CAFOs from the EPA's Enforcement and Compliance History Online (ECHO) database in the spring of 2015.[148]

Once we added all the information we could find to the database, we took additional steps to improve the data quality. First, facilities with inactive or expired permits were removed, along with facilities whose permit status indicated they were no longer operational. Second, duplicates were identified and consolidated into a single entry. Where duplicate entries contained different types of data, the information in both fields was retained, and where duplicate entries contained information that conflicted across the same field, NRDC prioritized the most recently collected data.

WHAT WE FOUND (AND DIDN'T FIND)

Overall, NRDC identified 7,595 CAFOs in 40 states. By comparison, the EPA estimated in 2012 that there were 17,329 CAFOs in the United States.[149] This means that we—and, by extension, the public regulators from which NRDC obtained the data—have information for fewer than half of the CAFOs that the EPA estimated to exist. Moreover, the EPA's estimate should be viewed cautiously; in nine states, NRDC found *more* facilities than the EPA estimated. In Maryland, for example, the agency estimated that there were 150 CAFOs, but NRDC's data indicate that there were

at least 789.[150] The problems and inconsistencies in the EPA's data make it impossible to know when the agency is overestimating or, more likely, underestimating the number of CAFOs in a state.

The most defining characteristic of the information we gathered is perhaps the inconsistency among the states in terms of data availability. NRDC found no data for nine states that the EPA determined housed CAFOs. These include Idaho, South Carolina, and Washington, each of which, according to EPA estimates, has more than 100 CAFOs within its borders. Even more striking, California has an EPA-estimated 1,028 CAFOs, but NRDC found data on only a single CAFO.

When data on CAFOs were available, both the quantity and nature of the data were highly variable from state to state. For some states, like Texas, Florida, and Iowa, a wide range of data was available about a limited number of CAFOs, including operations' permit status, location, method of storing manure, size, animal type, and ownership, but there were no data at all for many of the CAFOs that the EPA estimated are located in these states. In other states, such as Arizona, Montana, New York, Pennsylvania, South Dakota, and Utah, it appears likely that all or almost all CAFOs were accounted for, but only a very limited amount of data was available about each.

On balance, the state for which data appeared to be the most complete was, by a fairly wide margin, Tennessee. No other state had data available on permit status, location, method of storing manure, size, animal type, and ownership of more than 5 percent of EPA-estimated operations. For Tennessee, NRDC was able to obtain each of these data points for more than 70 percent of all estimated operations. Other states with relatively high numbers of CAFOs that did relatively well in terms of both the breadth and depth of available data included Alabama, Colorado, Indiana, and Maryland. The section below describes how the states lined up in terms of the particular types of data that were collected.

PERMIT STATUS

Overall, NRDC could identify the permit status of 4,234 CAFOs. This makes up about 24 percent of the EPA's estimate of total CAFOs in the country (and 56 percent of the CAFOs NRDC could identify). The data are skewed heavily toward several states. In 13 states—Colorado, Florida, Kansas, Maryland, Montana, New York, North Dakota, Oregon, Pennsylvania, Tennessee, Texas, Wisconsin, and Wyoming—NRDC found data on permitting for 50 percent or more of estimated CAFOs. These states contain 3,180 of those 4,234 CAFOs for which NRDC has data—more than 75 percent. On the other hand, for 18 states NRDC could find permitting data on less than 1 percent of estimated CAFOs. Several of these states— Arkansas, Arizona, California, Idaho, Louisiana, Nebraska, South Dakota, and Washington—have more than 100 CAFOs, according to EPA estimates.

OWNER/OPERATOR

NRDC found information identifying the owner or operator of 4,248 CAFOs, or roughly 25 percent of the EPA's overall estimate (and 56 percent of the CAFOs NRDC could identify). Though the total number of CAFOs for which NRDC has ownership data is nearly identical to that for permitting status, the list of states for which substantial ownership data are available is quite different. In 16 states—Alabama, Arizona, Delaware, Florida, Georgia, Indiana, Maryland, Michigan, Montana, Nebraska, New Jersey, North Dakota, Pennsylvania, Tennessee, Utah, and Wyoming—NRDC found data for 50 percent or more of estimated CAFOs. For 25 states, NRDC could find no data on ownership.

TYPE AND NUMBER OF ANIMALS

The data point that was most commonly available was information on the type of animal confined at a facility. Since, for example, a single hog produces far more manure than a single chicken, knowing the type of animal in confinement is necessary in order to assess the pollution potential of a CAFO. NRDC was able to determine this for 37 percent of the EPA's estimated number of CAFOs (and 84 percent of the CAFOs NRDC could identify). Knowing the number of animals confined is equally important, but NRDC could determine this for only 29 percent of the EPA's estimated number of CAFOs. This data void is particularly troubling; without knowing a CAFO's size, it is impossible to assess its impact on the community.

States that provided the most complete data on animal type and animal numbers included Alabama, Delaware, Georgia, Indiana, Maryland, North Dakota, South Dakota, Tennessee, and Utah. NRDC found data about the type and number of animals for more than 70 percent of estimated CAFOs in each of these states. Perhaps the most notable

laggard in animal type/number reporting is Pennsylvania. Although there are some data available for 356 CAFOs in the state, the type of animal is reported for only 51 of these, and there is no information about facility size. This is particularly concerning given that Pennsylvania is known to be a diverse CAFO state, housing cattle, dairy, hog, broiler, and egg layer operations.[151]

LOCATION

NRDC's database includes the location of 34 percent of CAFOs the EPA estimates exist (and 77 percent of the CAFOs NRDC could identify). Location is a crucial data point as it informs regulators and the public where concentrations of animals have become sufficiently high to warrant increased scrutiny, greater monitoring, or development of individualized permit conditions. Geographic data serves other important purposes, such as advancing our understanding of how CAFOs create disparate impacts on communities of color. For example, facility location data informed a complaint to the EPA under Title VI of the Civil Rights Act alleging that North Carolina's Department of Environmental Quality's permitting process had a racially discriminatory impact.[152] (see "Smithfield Foods and North Carolina.")

The states where NRDC could find location data for 80 percent or more of estimated CAFOs were New Jersey, Indiana, Colorado, Delaware, Montana, Michigan, Oregon, New York, and Maryland. While the southeastern United States generally performed well in terms of data availability, it lagged in facility location information. Across this region, NRDC found location data for more than half of all estimated CAFOs only in Georgia, Tennessee, and Alabama.

WASTE MANAGEMENT PRACTICES

Despite the crucial importance of waste management for the impact of CAFOs on neighboring communities, NRDC could find less information about waste management practices than for any other data category included in this report. Our database contains waste management information for 5.6 percent of the EPA's estimated number of CAFOs (and 13 percent of the operations NRDC could identify). NRDC could find this data for more than 2 percent of CAFOs in only seven states: Alabama, Florida, Iowa, North Carolina, Tennessee, Texas, and Wyoming. Some of the states with the most troubling records on waste management are notably absent from this list.

CONCLUSION

The fight for clean water and breathable air in communities neighboring industrial animal facilities stands at a crossroads. Efforts to improve how CAFOs are regulated have stagnated over the past decade, and the current administration has attacked federal environmental protections across the board, including the CWA. Yet in the absence of federal leadership, some states are seizing the opportunity to step up.

Recent developments suggest two directions in which regulators could move in the years to come. On the one hand, the current administration has pursued a path that weakens clean water protections and could allow more CAFO pollution to enter streams and other bodies of water. In this connection, the EPA recently reversed its longstanding interpretation that some subsurface discharges might trigger permitting under the CWA.[153] For decades, the EPA has maintained that point source discharges to subsurface waters with a direct connection to surface waters are subject to CWA permitting requirements.[154] In these circumstances, contamination from manure lagoons, especially unlined lagoons, can leak into subsurface water and then migrate to surface waters. The EPA's interest in reopening this issue signals its intent to exempt even more facilities from scrutiny and enforcement under the law. Weakening the interpretation discounts one of the primary routes by which CAFOs pollute bodies of water. Ultimately, this move will leave the EPA and the public with even less information about polluting facilities.

On the other hand, some states are taking a different path toward better regulation, more data collection, better transparency, and healthier communities. For example, the New York Supreme Court recently ruled that the state's dairy CAFO permit must be revised to reflect the CWA's rule that pollution-handling practices at permitted facilities are a matter of public record.[155] In the CAFO context, this means nutrient management plans must be available for public review, which is critical to holding facilities accountable and ensuring that the best management practices are used when and where they are required.

The CWA provides minimum regulatory standards for New York and the 45 other states that currently run their own CAFO permitting program.[156] States are free, however, to go above CWA standards to protect their local waters and the health of their local communities, and NRDC is working to empower states to do so.[157] To facilitate the process, NRDC scoured state permitting programs from across the nation to identify best practices and synthesize them into a permit that states can adopt under their existing powers.

NRDC's CAFO permit goes above and beyond the current federal minimum NPDES requirements in some ways. For example, many facilities avoid their duty to obtain a CWA permit by claiming not to discharge. The permit expands coverage to all confined animal operations within a state, so that every facility above a certain size threshold has a duty to obtain a permit. It requires that basic information on each facility, like location, size, and waste management practices, be available in a public database, as well as reported directly to the state and the EPA. It is important for regulators and the public to have this information in order to understand how CAFOs are distributed over the landscape and which water bodies are threatened by high concentrations of polluting facilities. The permit addresses the data collection void that has hobbled the EPA and the states in meeting their regulatory duties. The permit also improves transparency and accountability more generally.

Collecting and sharing basic information on facilities is only part of the battle. Communities deserve to know the conditions and standards that apply to nearby facilities, and those same communities need the power to hold facilities accountable when they violate the terms of their permit. In order to obtain a permit, CAFO operators currently must develop a nutrient management plan that specifies how the facility will live up to its obligations under the permit. NRDC's permit clarifies that the nutrient management plan must be public information. It also expands notice requirements so that nearby residents are fully informed when a CAFO wants to expand its facility or open a new one, or when existing operations are making changes that will increase the production of waste. To further empower communities to hold neighboring CAFOs accountable, the permit affirms the right of individual citizens to bring a civil action against any person found to be in violation of CWA standards or limitations.

There is a long road ahead in the fight to clean up industrial animal operations and hold them accountable for their pollution. As a first step, the EPA, states, and communities must understand the true scope and scale of the agriculture industry. That understanding requires comprehensive and reliable CAFO data that are currently not available. To remedy this, federal policymakers should reintroduce the reporting requirements they dropped under industry pressure in 2012. In the absence of federal action, states must use their existing authority to demand transparency and accountability of the CAFOs within their borders. Citizens concerned with the environmental and health threats that CAFOs pose to their communities must demand much more from their representatives at both the state and federal levels.

TRANSPARENCY OF CAFO DATA
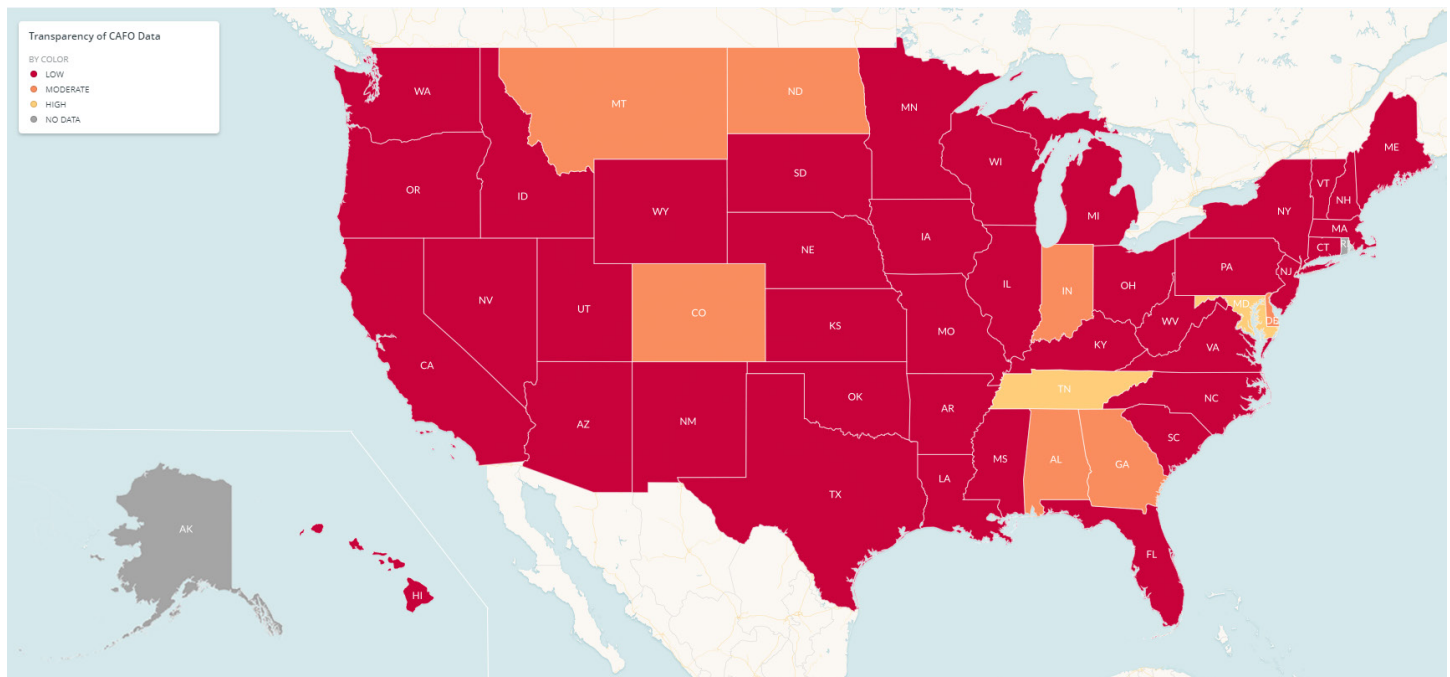
STATE TRANSPARENCY RATINGS						
State	Overall Transparency Rating	Transparency of Permit status	Transparency of Location	Transparency of Manure storage	Transparency of Type of Animal	Transparency of Count of Animal	Transparency of Owner Information
Alabama	Moderate	Low	Moderate	Moderate	Moderate	Moderate	Moderate
Alaska	No Data	No Data	No Data	No Data	No Data	No Data	No Data
Arizona	Low	Low	Low	Low	High	Low	High
Arkansas	Low	Low	Low	Low	Low	Low	Low
California	Low	Low	Low	Low	Low	Low	Low
Colorado	Moderate	High	High	Low	High	Low	Low
Connecticut	Low	Low	Low	Low	Low	Low	Low
Delaware	Moderate	Low	High	Low	High	High	High
Florida	Low	Low	Low	Low	Low	Low	Low
Georgia	Moderate	Low	Moderate	Low	Moderate	Moderate	Moderate
Hawaii	Low	Low	Low	Low	Low	Low	Low
Idaho	Low	Low	Low	Low	Low	Low	Low
Illinois	Low	Low	Low	Low	Low	Low	Low
Indiana	Moderate	Low	High	Low	High	High	High
Iowa	Low	Low	Low	Low	Low	Low	Low
Kansas	Low	Moderate	Low	Low	Moderate	Low	Low
Kentucky	Low	Low	Low	Low	Low	Low	Low
Louisiana	Low	Low	Low	Low	Low	Low	Low
Maine	Low	Low	Low	Low	Low	Low	Low
Maryland	High	High	High	Low	High	High	High
Massachusetts	Low	Low	Low	Low	Low	Low	Low
Michigan	Low	Low	High	Low	Moderate	Low	High
Minnesota	Low	Low	Low	Low	Low	Low	Low
Mississippi	Low	Low	Low	Low	Low	Low	Low
Missouri	Low	Low	Low	Low	Low	Low	Low
Montana	Moderate	High	High	Low	Moderate	Low	Low
Nebraska	Low	Low	Low	Low	Low	Low	Low
Nevada	Low	Low	Low	Low	Low	Low	Low
New Hampshire	Low	Low	Low	Low	Low	Low	Low
New Jersey	Low	Low	High	Low	Low	Low	High
New Mexico	Low	Low	Low	Low	Low	Low	Low
New York	Low	Low	High	Low	Low	Moderate	Low
North Carolina	Low	Low	Low	Low	Low	Low	Low
North Dakota	Moderate	Moderate	Moderate	Low	Moderate	Moderate	Moderate
Ohio	Low	Low	Moderate	Low	Moderate	Moderate	Low
Oklahoma	Low	Low	Low	Low	Low	Low	Low
Oregon	Low	High	High	Low	Low	High	Low
Pennsylvania	Low	High	Low	Low	Low	Low	High
Rhode Island	No Data	No Data	No Data	No Data	No Data	No Data	No Data
South Carolina	Low	Low	Low	Low	Low	Low	Low
South Dakota	Low	Low	Low	Low	High	High	Low
Tennessee	High	High	Moderate	High	High	High	High
Texas	Low	Low	Low	Low	Low	Low	Low
Utah	Low	Low	Low	Low	Moderate	Moderate	Moderate
Vermont	Low	Low	Low	Low	Low	Low	Low
Virginia	Low	Low	Low	Low	Low	Low	Low
Washington	Low	Low	Low	Low	Low	Low	Low
West Virginia	Low	Low	Low	Low	Low	Low	Low
Wisconsin	Low	High	Moderate	Low	High	Low	Low
Wyoming	Low	Moderate	Low	Low	Low	Low	Moderate

Methodology

DATA SOURCES

We obtained the data on the CAFOs that we used for our analysis, map, and Table 1 from four sources:

SPREADSHEETS OF DATA ON CAFOS OBTAINED FROM THE U.S. ENVIRONMENTAL PROTECTION AGENCY (EPA)

Pursuant to a Freedom of Information Act (FOIA) request filed by NRDC in 2012, we obtained spreadsheets containing data on CAFOs that had been supplied by states to the EPA, as well as a few spreadsheets containing information about CAFOs gathered by the EPA from publicly available sources. Although the EPA initially provided us with unredacted data from states, we returned those documents and based our analysis on subsequent, redacted versions of the EPA's data.

We did not use spreadsheets produced by the FOIA request that contained data gathered from the EPA's own PCS/ICIS systems in order to avoid duplication; we gathered updated versions of those data from the EPA's Envirofacts and ECHO databases later. We also did not use most of the data that were compiled by the EPA from publicly available state web sources, again to avoid duplication; we later gathered updated versions of publicly available information from state CAFO permitting websites.

PCS/ICIS SEARCHES ON THE EPA ENVIROFACTS WEBSITE

On the EPA Envirofacts website, we conducted searches of the Permit Compliance System (PCS) and Integrated Compliance Information System (ICIS) databases at http://www3.epa.gov/enviro/facts/pcs-icis/search.html. We searched in every state for animal agricultural facilities with Standard Industrial Classification codes starting with "02," the code for Agricultural Production—Livestock. These searches were conducted in April and May 2014.

From our searches in Envirofacts, we added to our database those facilities that had active National Pollutant Discharge Elimination System (NPDES) permits. We excluded facilities that had NPDES permits but did not appear to be CAFOs, based on their animal types. For example, we excluded aquaculture facilities from our list.

STATE CAFO PERMITTING WEBSITES CONTAINING INFORMATION ABOUT CAFOS WITHIN THE STATE

In April 2015, we conducted searches of state websites that contained publicly available information about CAFOs. We extracted all readily available information from these websites on AFOs regardless of whether they were NPDES-permitted, and we added this information to our database.

We considered information about CAFOs to be "readily available" if it was downloadable in a batch CSV or Excel file, or if we were otherwise able to copy and paste it from the website en masse. Some state websites contained additional pieces of information about CAFOs on separate pages for each facility, or linked to PDFs of permitting documents for facilities. We did not consider this type of information to be readily available due to the substantial amount of time it would have taken to manually extract information about each facility, one by one. We did not include information that was not readily available in our database, except in the cases of Iowa and Texas. The websites for these states had a substantial amount of important information about CAFOs on separate sub-pages for each facility. Our web contractor was able to write code to extract this information about the facilities from the separate sub-pages.

The state websites from which we gathered data for our database were these:

- Alabama Department of Environmental Management, "Animal Feeding Operations," http://www.adem.state.al.us/programs/water/cafo.cnt.

- Arkansas Department of Environmental Quality, "ADEQ Facility and Permit Summary," https://www.adeq.state.ar.us/home/pdssql/pds.aspx.

- California Environmental Protection Agency, State Water Resources Control Board, "Regulated Facility Report," https://ciwqs.waterboards.ca.gov/ciwqs/readOnly/CiwqsReportServlet?inCommand=reset&reportName=RegulatedFacility.

- Florida Department of Environmental Protection, "Wastewater Facility Information," http://www.dep.state.fl.us/water/wastewater/facinfo.htm.

- Indiana Department of Environmental Management, "Confined Feeding Operations," http://www.in.gov/idem/landquality/2349.htm.

- Iowa Department of Natural Resources, "Animal Feeding Operation," https://programs.iowadnr.gov/animalfeedingoperations/FacilitySearch.aspx.

- Maryland Department of the Environment, "Permit Application Database Search," https://mde.maryland.gov/programs/permits/pages/sb47.aspx.

- Mississippi Department of Environmental Quality, OPC Environmental Permits Division, "Active Permit & Coverage Search," http://opc.deq.state.ms.us/search_ai_alt.aspx#grid.

- New Jersey Department of Environmental Protection, "NJPDES Excel Reports," http://datamine2.state.nj.us/dep/DEP_OPRA/NJDEPexcel.htm.

- North Carolina Department of Environmental Quality, "Animal Feeding Operations: Permits," http://portal.ncdenr.org/web/wq/aps/afo/perm.

- Tennessee Department of Environment and Conservation, "Heritage WPC Permits in Tennessee," http://environment-online.tn.gov:8080/pls/enf_reports/f?p=9034:34001:0: .

- Texas Commission on Environmental Quality, "Water Quality General Permits & Registration Search," http://www2.tceq.texas.gov/wq_dpa/index.cfm. .

- State of Washington Department of Ecology, "Facility/Site Search," https://fortress.wa.gov/ecy/facilitysite/SearchData/ShowSearch.aspx?ModuleType=FacilitySite&RecordSearchMode=New.

- Wisconsin Department of Natural Resources, "CAFO Permittees," http://dnr.wi.gov/topic/AgBusiness/data/CAFO/cafo_all.asp?FULL=1.

THE EPA ENFORCEMENT AND COMPLIANCE HISTORY ONLINE WEBSITE

We conducted a search on April 9, 2015, of all NPDES-permitted CAFOs in the EPA Enforcement and Compliance History Online (ECHO) website, http://echo.epa.gov/?redirect=echo. We added this data to our database by state and included all NPDES-permitted facilities, regardless of their permit status. We later filtered out facilities whose permit status indicated that they were no longer operational, as discussed below.

For a complete listing of each source of data collected for each state, please see our Source List [link to source attribution].

TYPES OF DATA COLLECTED

From the sources listed above, we compiled data about AFOs across 24 categories:

- State
- Permit type
- Permit number
- Permit status
- CAFO name
- CAFO address
- Name of the owner and operator
- Address of the owner and operator
- If contract operation, name and address of the integrator
- Longitude of the operation
- Latitude of the operation
- Type of facility
- Type of animals

- Number of animals (or range)

- Number of animal units

- Type and capacity of manure storage

- Available acreage for land application

- Quantity of manure, process wastewater, and litter generated annually by the CAFO

- Whether the CAFO land-applies (Y/N)

- If the CAFO land-applies, whether it implements a nutrient management plan for land application

- If the CAFO land-applies, whether it employs nutrient management practices and keeps records on site consistent with 40 CFR 122.23(e)

- If the CAFO does not land-apply, alternative uses of manure, litter, and/or wastewater

- Whether the CAFO transfers manure off site

- If the CAFO transfers manure off site, quantity transferred to recipient(s)

ELIMINATING DUPLICATION OF DATA

We took steps to limit the amount of duplication in our database. However, given our multiple sources, there were duplicate entries for many facilities in our database when our collection was complete. To accurately analyze "readily available" data about CAFOs, we had to eliminate this duplication. First, we identified duplicate facilities using the following metrics independently:

- Latitude and longitude to four decimal places

- Permit number

- CAFO name within a state

- CAFO address

- CAFO owner within a state

After duplicates were identified, we consolidated information for those CAFOs across duplicate entries instead of eliminating duplicates entirely, because different entries contained different pieces of information. Thus, where duplicate entries contained information in different fields, the information in both fields was retained. Where duplicate entries contained information that conflicted within the same field, we chose which information to retain on the basis of the source. We created a hierarchy of sources according to how recent their data were and whether they were state or federal sources. We prioritized the latter because federal sources contained consistent and reliable information on NPDES permits that we wanted to retain. This hierarchy was as follows:

1. ECHO (more recently gathered than Envirofacts)

2. Envirofacts

3. State public databases (more recently gathered than FOIA)

4. State data from FOIA request

We used this hierarchy as our default and diverged from it only for a few states in order to ensure that we were capturing the most precise information on animal types, which was contained in the state sources.

Finally, we did a manual review of our data to identify any remaining duplicate entries and consolidated their information as described above.

REFINEMENT OF DATA

After ensuring that there was just one entry for each facility, we limited the facilities that appear in the database to CAFOs that either had a NPDES permit or contained enough animals to qualify as a Large CAFO per federal regulations.

We reviewed state permitting schemes to determine which permits were NPDES permits and included all NPDES-permitted facilities for which we had information in the database.

For facilities without NPDES permit information, we reviewed our data on the number of animals (or range) at each facility to determine whether the facility would qualify as a Large CAFO. We possessed information from a few states on the animal units at each facility rather than the number of animals, but we did not use animal unit data in our analysis because current federal CAFO regulations categorize facilities on the basis of animal numbers, not units. Further, we did not attempt to convert animal unit data to animal numbers because of a lack of information on the specific types of animals contained. We included in our database only those facilities that would qualify as Large CAFOs and did not include facilities that might qualify as a Medium or Small CAFO. We made this decision because in order for a facility to be a Medium or Small CAFO per federal regulations, it must also meet other conditions relating to its pollution or be specifically designated as such by an appropriate authority. Because Large CAFOs are the only category of CAFO defined as such on the basis of animal numbers alone, we could confidently categorize as CAFOs only those facilities with enough animals to be considered Large CAFOs. Therefore, we only included facilities in our CAFO database on the basis of animal numbers alone that would qualify as Large CAFOs.

We used the following definitions of Large CAFOs from the federal regulations:

An AFO is defined as a Large CAFO if it stables or confines as many as or more than the numbers of animals specified in any of the following categories:

- 700 mature dairy cows, whether milked or dry

- 1,000 veal calves

- 1,000 cattle other than mature dairy cows or veal calves. Cattle includes but is not limited to heifers, steers, bulls, and cow/calf pairs

- 2,500 swine each weighing 55 pounds or more

- 10,000 swine each weighing less than 55 pounds

- 500 horses

- 10,000 sheep or lambs

- 55,000 turkeys

- 30,000 laying hens or broilers, if the AFO uses a liquid manure handling system

- 125,000 chickens (other than laying hens), if the AFO uses other than a liquid manure handling system

- 82,000 laying hens, if the AFO uses other than a liquid manure handling system

- 30,000 ducks, if the AFO uses other than a liquid manure handling system

- 5,000 ducks, if the AFO uses a liquid manure handling system

We assigned the animal types that were described in our data to the animal categories in the federal CAFO regulations. When we did not have enough information about the type of animal contained at a CAFO to assign it to an animal category under the federal definitions, we were conservative in assigning it to a category. For example, if we knew that a facility contained swine but did not know the size of the swine, we would assign the swine at that facility to the category "swine each weighing less than 55 pounds." Similarly, if we did not have information on a poultry facility's manure handling system, we would conservatively assume that it used something "other than a liquid manure handling system." By doing so, we avoided designating facilities as large CAFOs when they actually were not.

Using this conservative approach, we classified the following animal types that were present in our data as the following animal types under the CAFO regulations for the purpose of determining whether the number of animals the facility contained put it over the threshold for Large CAFOs.

ANIMAL TYPE IN CAFO REGULATIONS	ANIMAL TYPE FROM DATA
Mature dairy cows, whether milked or dry	Cattle–Milk Cow; Dairy; DAIRY; dairy; Dairy Cattle; dairy cattle; Dairy Cattle (Mature); Dairy Cow, Dry; Dairy Cow, Milking; DAIRY FARMS; Dairy Farms; Mature Dairy; Mature Dairy Cattle; milking; milking/dry
Veal calves	Calves; Cattle - veal; Veal Calves
Cattle other than mature dairy cows or veal calves. Cattle includes but is not limited to heifers, steers, bulls, and cow/calf pairs.	Beef; Beef calf; Beef calves; Beef Cattle; beef cattle; Beef cattle; Beef Cattle Feedlots; BEEF CATTLE FEEDLOTS; Beef Cow; Beef Feeder; Calves; calves and heifers; Cattle; CATTLE; cattle; Cattle – Beef Brood Cow; Cattle – Beef Feeder; Cattle – Beef Stocker Calf; Cattle – Dairy Calf; Cattle – Dairy Heifer; Cattle – Dry Cow; Cattle – includes heifers; Cattle (All except Mature Dairy Cattle and Veal Calves); Cattle (includes heifers); Cattle (Not Mature Dairy/Veal); cows; Dairy calf; Dairy Calves; Dairy Heifer; Dairy Heifers; Dairy Heifers/Calves; heifer; heifers; Heifers; Heifers/Calves
Swine each weighing 55 pounds or more	Boars; finisher swine; Finishers; Finishers-pigs; Pigs, Boar; Pigs, Finishing; Pigs, Gestating; Pigs, Lactating; Sows; sows; Swine – Boar/Stud; Swine (Greater than 55 Lbs.); Swine >= 55 lbs; Swine >= 55 pounds; Swine – Large; Swine Lg; SWINE LG; Swine over 55 lbs.
Swine each weighing less than 55 pounds	farrow to finish swine; grow to finish swine; HOGS; Hogs; Nursery Pigs; nursery swine; nursery swine, finisher swine; Piglets; Pigs, Grower; Pigs, Nursery; production swine; production, finisher swine; Swine; swine; Swine – Farrow to Feeder; Swine – Farrow to Finish; Swine – Farrow to Wean; Swine – Feeder to Finish; Swine – Gilts; Swine – Med; Swine – Other; Swine – Wean to Feeder; Swine – Wean to Finish; Swine (Less than 55 Lbs.); Swine < 55 lbs; Swine < 55 pounds; SWINE LG/MD; Swine Md; SWINE MD; Swine Sm; SWINE SM; Swine under 55 lbs
Horses	Horse; Horses; horses; Horses – Horses; Horses – Other; Horses And Other Equines; Horses and other equines
Sheep or lambs	Sheep; Sheep or Lambs
Turkeys	Turkey; turkey; Turkeys; turkeys; Turkeys and Turkey Eggs
Laying hens or broilers, if the AFO uses a liquid manure handling system	Chickens with liquid manure handling; Layer (Wet); Wet Poultry – Layers; Wet Poultry – Non Laying Pullets; Wet Poultry – Other
Chickens (other than laying hens), if the AFO uses other than a liquid manure handling system	Breeder; Broiler; BROILER; broiler; BROILER FRYER ROAST CHICKENS; Broiler, fryer, and roaster chickens; Broiler, Fryer and Roaster Chickens; Broilers; chicken; Chickens; chickens; Chickens (All except Layers); Chickens (other than laying hens) with dry manure handling; Chickens-not laying hens-dry; Poultry; poultry; Poultry and Eggs; POULTRY AND EGGS NEC; Poultry Hatcheries; Poultry – Breeders; Poultry – Broilers; Poultry – Pullets; Poulty; Pullet; PULLET; Pullet Poultry; Pullets
Laying hens, if the AFO uses other than a liquid manure handling system	Chicken Eggs; CHICKEN EGGS; Chicken Eggs, Poultry and Eggs; Chicken, Layer; Chickens (Layers); Dry Poultry – Laying Chickens; Layer; Layers; layers; Laying Hens; Laying hens – dry manure; Laying Hens (dry); Laying hens with dry manure handling; Poultry – Breeder layers; Poultry – Layers; Poultry, layers
Ducks, if the AFO uses other than a liquid manure handling system	Duck; Ducks
Ducks, if the AFO uses a liquid manure handling system	Ducks – liquid manure

Any listed animal types other than those in the table above were logged as either "other" or "unknown," and their animal numbers, to the extent they were provided, were not used to qualify a facility as a Large CAFO.

FILTERING OF PERMIT STATUSES

We further limited the data in our final data set of CAFOs by filtering out facilities whose permit status clearly indicated that it was no longer operational. We filtered out facilities with the following permit statuses:

- Expired
- Closed
- Termination
- Voided
- App Terminated
- CFO Approval Expiration - Date Issued 03/13/2015
- CFO Approval Expiration - Pending
- CFO Approval Voidance - Date Issued 03/27/2015
- CFO Approval Voidance - Pending
- Historical
- Inactive
- Number was skipped
- Permit Terminated
- REVOK

We encountered other permit statuses indicating that a facility was potentially no longer permitted, and we excluded these facilities as well under the following conditions: 1) if the facility was in the database only because it had an NPDES permit, and 2) if we did not have information indicating that the number of animals the facility contained met the threshold to qualify it as a Large CAFO. We filtered out facilities that met those conditions and had the following permit statuses:

- Incomplete
- No
- NOT
- Not given
- Not issued
- not permitted
- sold
- Transferred
- UNPERMITTED
- WITH
- Withdrawn

FILTERING OF ANIMAL TYPES

We also filtered out of the database facilities that were NPDES-permitted but had the following animal types that indicated they were not CAFOs. These animal types were:

- ANIMAL AQUACULTURE
- Animal aquaculture
- BEEF CATTLE EXCEPT FEEDLOTS
- Beef Cattle, Except Feedlots
- Beef cattle, except feedlots
- Beef Cattle, Except Feedlots
- Beef Cattle, Except Feedlots, Livestock
- Beef; Beef cattle, except feedlots
- Fish hatcheries
- FISHFARM
- Fishing preserves
- LOCAL TRUCKING WITHOUT STORAGE
- Tropical fish farm
- Trout farm

CREATION OF CAFO MAP AND TABLE 1

In the map and Table 1, "Level of Transparency" was determined based on the amount of "readily available" data.

SCORING

The following six categories of information were chosen to assess the transparency of CAFO data in each state. NRDC considers this information to be necessary to protect communities from CAFO pollution and hold industry accountable if they fail to meet their responsibilities under the law:

- Location
- Permit status
- Type of manure storage
- Count of animals
- Type of animal
- Owner information

For each of these six categories for each state, percentages were calculated using the number of CAFOs for which we have the relevant information, divided by the EPA's estimate of the total number of CAFOs in the state at the time the information was gathered.[a] Each category received a score of 0 to 5 points, based on the percentages:

- 0% = 0 points
- Less than 20% = 1 point
- 20% to <40% = 2 points
- 40% to <60% = 3 points
- 60% to <80% = 4 points
- 80% and above = 5 points

a   U.S. Environmental Protection Agency, *NPDES CAFO Rule Implementation Status—National Summary, Midyear 2012*, June 30, 2012, https://www3.epa.gov/npdes/pubs/tracksum%20midyear2012_publish.pdf (accessed July 22, 2018).

The points from each of the six categories were added for the final score. The maximum score was 30. The final score determined the final transparency rating, based on the following ranges:

High Transparency = 24–30 points.
80% or more of all CAFO sites in the state are accounted for with readily available data.

Moderate Transparency = 18–23 points.
About 60% to 80% of all CAFO sites in the state are accounted for with readily available data.

Low Transparency = 1–17 points.
60% or fewer of all CAFO sites in the state are accounted for with readily available data.

No Data = No data were readily available for CAFO sites in the state.

DISCREPANCY IN DATA

For the following states, NRDC found data for more sites than the EPA estimated to be present in the state, which we used as our denominator when calculating percentages. Due to the discrepancy in data, we will make a note of this in the map and table for the following states: Delaware, Maryland, Michigan, Montana, New York, Oregon, Pennsylvania, Utah, and Wisconsin.

ENDNOTES

1   *See* "CAFOs and Their Pollution," *infra.*

2   Chris Augustin, Charles Stoltenow, and Ron Wiederholt, "Biosecure Nutrient Management Practices," 1, North Dakota State University, reprinted January 2017, https://www.ag.ndsu.edu/publications/livestock/biosecure-nutrient-management-practices.

3   Carrie Hribar, *Understanding Concentrated Animal Feeding Operations and Their Impact on Communities*, 6, National Association of Local Boards of Health, 2010, https://www.cdc.gov/nceh/ehs/docs/understanding_cafos_nalboh.pdf.

4   Carol M. Browner, et al., *Environmental Assessment of Proposed Revisions to the National Pollutant Discharge Elimination System Regulation and Effluent Limitations Guidelines for Concentrated Animal Feeding Operations*, 3-1, U.S. Environmental Protection Agency (hereinafter EPA), January 2001, https://www3.epa.gov/npdes/pubs/cafo_proposed_env_assess_ch1-3.pdf.

5   Union of Concerned Scientists, "Prescription for Trouble: Using Antibiotics to Fatten Livestock," last revised August 11, 2009, https://www.ucsusa.org/food_and_agriculture/our-failing-food-system/industrial-agriculture/prescription-for-trouble.html#.XEI5SM9KjaY; Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 10.

6   40 CFR § 122.23(b)(1); U.S. Department of Agriculture, Natural Resources Conservation Service, "Animal Feeding Operations," https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/plantsanimals/livestock/afo/.

7   EPA, Office of Water, *NPDES Permit Writers' Manual for Concentrated Animal Feeding Operations*, EPA 833-F-12-001, § 2.2, February 2012, https://www3.epa.gov/npdes/pubs/cafo_permitmanual_chapter2.pdf. 40 CFR § 122.23(b), (c).

8   Angela Huffman, Joe Maxwell, and Andres Salerno, *Consolidation, Globalization, and the American Family Farm*, 1-2, Organization for Competitive Markets, August 2017, https://competitivemarkets.com/wp-content/uploads/2017/08/Consolidation-Globalization-and-the-American-Family-Farm.pdf.

9   *See* "The Existing Regulatory Regime," *infra.*

10   U.S. Government Accountability Office (hereinafter GAO), *Concentrated Animal Feeding Operations: EPA Needs More Information and a Clearly Defined Strategy to Protect Air and Water Quality from Pollutants of Concern*, GAO-08-944, 4-5, September 2008, https://www.gao.gov/products/GAO-08-944.

11   76 Fed. Reg. 65436.

12   See "The CAFO Data Void," *infra.*

13   Withdrawal CAFO Reporting Rule, 77 Fed. Reg. 42,679-81 (withdrawn July 20, 2012).

14   Several months after their provision, EPA asked the Natural Resources Defense Council (hereinafter NRDC) to return some of these data because they contained information that EPA determined was private, such as personal telephone numbers. Because NRDC was interested in information about these facilities' environmental footprints, NRDC returned the requested information and allowed EPA to update the scope of its release.

15   *Am. Farm Bureau Fed'n v. EPA,* 0:13–cv–01751–ADM–TNL (D.Minn.2013).

16   *Am. Farm Bureau Fed'n v. EPA*, No. CIV. 13-1751 ADM/TNL, 2015 WL 364667 (D. Minn. Jan. 27, 2015).

17   *Am. Farm Bureau Fed'n v. EPA*, 836 F.3d 963 (8th Cir. 2016).

18   *Am. Farm Bureau Fed'n v. EPA*, 836 F.3d 963 (8th Cir. 2016).

19   *See* "Research Methodology," *infra.*

20   EPA, *NPDES CAFO Rule Implementation Status—National Summary, Midyear 2012*, June 30, 2012, https://www3.epa.gov/npdes/pubs/tracksum%20midyear2012_publish.pdf.

21   40 CFR § 122.23(b)(1-2).

22   40 CFR § 122.23(b)(4),(6).

23   ·EPA, Office of Water, *NPDES Permit Writers' Manual.*

24   EPA, "History of the Clean Water Act," https://www.epa.gov/laws-regulations/history-clean-water-act (accessed January 18, 2019); 33 U.S.C. §§ 1251 et seq. (2012).

25   EPA, "NPDES Permit Basics," https://www.epa.gov/npdes/npdes-permit-basics (accessed January 18, 2019); 33 U.S.C. §§ 1311(a), 1342(a), 1362(12).

26   EPA, Office of Water, *NPDES Permit Writers' Manual,* § 1-2, September 2010, https://www.epa.gov/sites/production/files/2015-09/documents/pwm_chapt_01.pdf. 33 U.S.C. § 1251(a)(1).

27   EPA, "Clean Water Act (CWA) Compliance Monitoring," https://www.epa.gov/compliance/clean-water-act-cwa-compliance-monitoring (accessed July 22, 2018).

28   EPA, "Clean Water Act (CWA) Compliance Monitoring."

29   EPA, "*Effluent Limitations Guidelines*," 39 Fed. Reg. 5704 (Feb. 14, 1974) (codified at 40 C.F.R. pt. 412).

30   EPA, *Pollutant Discharge Elimination: Form and Guidelines Regarding Agricultural and Silvicultural Activities*, 38 Fed. Reg. 18,000-18,001 (July 5, 1973).

31   *NRDC v. Costle*, 568 F.2d 1369, 1383 (D.C. Cir. 1977). GAO, *Livestock Agriculture: Increased EPA Oversight Will Improve Environmental Program for Concentrated Animal Feeding Operations*, GAO-03-285, 3, January 2003, https://www.gao.gov/new.items/d03285.pdf.

32   GAO, *Livestock Agriculture: Increased EPA Oversight*, 11, January 2003.

33   Water Quality Act of 1987, Pub. L. No. 100-4, § 503, 101 Stat. 7 (1987).

34   Emily Kenyon, "Enough of This Manure: Why the EPA Needs to Define the Agricultural Stormwater Exemption to Limit the 'Runoff' From the *Alt* Court," *New York University Law Review* 92 (October 2017): 1187, 1197.

35   EPA, *National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feed Operations (CAFOs)*, 68 Fed. Reg. 7176 (Feb. 12, 2003). *Natural Res. Def. Council v. Reilly*, Civ. No. 89-2980, 1991 U.S. Dist. LEXIS 5334, at *12, *30 (D.C. Cir. Apr. 21, 1991).

36   *Waterkeeper All., Inc. v. EPA*, 399 F.3d 486, 524 (2d Cir. 2005).

37   *Nat'l Pork Producers Council v. EPA*, 635 F.3d 738, 756 (5th Cir. 2011).

38   EPA, *National Pollutant Discharge Elimination System Permit Regulation for Concentrated Animal Feeding Operations: Removal of Vacated Elements in Response to 2011 Court Decision*, 77 Fed. Reg. 44,494, 44,494 (July 30, 2012) (codified at 40 C.F.R. pt. 122).

39   40 C.F.R. § 122.23(e) (2011).

40   *Alt v. EPA*, 979 F. Supp. 2d 701, 711 (N.D. W. Va. 2013). This finding contradicted decades of case law in which courts found that over-application of animal waste to agricultural fields may be a "solid waste" under the Resource Conservation and Recovery Act. *See, e.g.*, *Water Keeper All., Inc. v. Smithfield Foods, Inc.*, No. 4:01-CV-27-H(3), 2001 WL 1715730, at *5 (E.D.N.C. Sept. 20, 2001) (animal waste may be a "solid waste" under RCRA when applied "in such large quantities that its usefulness as organic fertilizer is eliminated"); *Cmty. Ass'n for Restoration of the Env't, Inc. v. R & M Haak, LLC*, No. 13-CV-3026-TOR, 2013 WL 3188855, at *5 (E.D. Wash. June 21, 2013) (". . . Defendants over-applied and improperly applied manure to their fields, and allowed liquid manure to leak from lagoons, thereby 'discarding' the manure and qualifying it as 'solid waste' under RCRA").

41   Motion to Voluntarily Dismiss Appeal, *Alt v. EPA*, No. 13-2527 (L) (4th Cir. Sept. 22, 2014); Order, *Alt v. EPA*, No. 13-2527 (L) (4th Cir. Sept. 24, 2014) (granting EPA motion to dismiss appeal). Cynthia Giles, "A Commitment to Keep Our Waters Clean and Safe," *EPA Connect*, September 19, 2014, http://blog.epa.gov/blog/2014/09/a-commitment-to-keep-our-watersclean-and-safe/.

42   James M. MacDonald and Robert A. Hoppe, "Examining Consolidation in U.S. Agriculture," *USDA Economic Research Service*, March 14, 2018, https://www.ers.usda.gov/amber-waves/2018/march/examining-consolidation-in-us-agriculture/.

43   Ibid.

44   GAO, *Concentrated Animal Feeding Operations: EPA Needs More Information*, 4.

45   U.S. Department of Labor, Bureau of Labor Statistics, "Consumer Price Index—Average Price Data: Chicken, fresh, whole, per lb." https://data.bls.gov/PDQWeb/ap (accessed September 21, 2018).

46   Christopher Leonard, "The Ugly Economics of Chicken," *The Week*, April 13, 2014, http://theweek.com/articles/447911/ugly-economics-chicken.

47   *See generally*, Michelle B. Nowlin, "Sustainable Production of Swine: Putting Lipstick on a Pig?" *Vt. L. Rev.* 37 (2013): 1079.

48   Ibid. at 1096-1101.

49   *See generally*, Erin M. Tegtmeier and Michael D. Duffy, "External Costs of Agricultural Production in the United States," *Int'l J. Agric. Sustainability* 2, no. 1 (2004).

50   GAO, *Concentrated Animal Feeding Operations: EPA Needs More Information*, 5.

51   EPA, Emission Standards Division, Office of Air Quality Planning and Standards, *Emissions From Animal Feeding Operations*, preliminary draft report 2-15 (Aug. 15, 2001), www.epa.gov/ttn/chief/ap42/ch09/draft/draftanimalfeed.pdf.

52   EPA, *Emissions From Animal Feeding Operations*.

53   Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 8. Melanie Redding, *Manure and Groundwater Quality: Literature Review*, 26-29, State of Washington, Department of Ecology, June 2016, https://fortress.wa.gov/ecy/publications/documents/1603026.pdf.

54   Amira Oun et al., "Effects of Biosolids and Manure Application on Microbial Water Quality in Rural Areas in the US," *Water* 6, no. 12 (2014): 3701-23, www.mdpi.com/2073-4441/6/12/3701/pdf.

55   Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 7.

56   Ibid. at 2.

57   Ibid. at 3.

58   Ibid.

59   Pew Environmental Group, "Animal Agriculture and the Clean Water Act," 3, 2010, http://www.pewtrusts.org/~/media/assets/2010/12/01/animal-agriculture-and-the-clean-water-act.pdf.

60   Environmental Integrity Project, *Manure Overload on Maryland's Eastern Shore*, 1, December 8, 2014, http://www.environmentalintegrity.org/wp-content/uploads/2016/11/Manure-Overload1.pdf.

61   ˙Ibid.

62   Ibid. at 3.

63   Ibid. at 6.

64   Ibid. at 2.

65   University of Maryland Center for Environmental Science, "Harmful Algal Blooms in the Chesapeake Bay Are Becoming More Frequent," *ScienceDaily*, May 11, 2015, www.sciencedaily.com/releases/2015/05/150511125219.htm.

66   EPA, "Chesapeake Bay Total Maximum Daily Load," https://www.epa.gov/sites/production/files/2015-07/documents/bay_tmdl_fact_sheet.pdf (accessed November 7, 2018).

67   Ibid.

68   University of Maryland Center for Environmental Science, "Eco Health Report Cards: Chesapeake Bay," https://ecoreportcard.org/report-cards/chesapeake-bay/health/ (accessed February 5, 2019).

69   *See, e.g.*, Steve Wing and Jill Johnston, "Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians," NC Policy Watch, 2014, http://www.ncpolicywatch.com/wp-content/uploads/2014/09/UNC-Report.pdf; Wendee Nicole, "CAFOs and Environmental Justice: The Case of North Carolina," 121 *Environ. Health Perspect.* 121 (2013): A182–A189, https://www.ncbi.nlm.nih.gov/pubmed/23732659, citing Bob Edwards and Anthony E. Ladd, "Race, Poverty, Political Capacity and the Spatial Distribution of Swine Waste in North Carolina, 1982–1997," *North Carolina Geographer* 9 (2001): 55–77.

70   Wendee Nicole, "CAFOs and Environmental Justice."

71   Jackie Wang, Nicole Tyau, and Chelsea Rae Ybanez, "Farming Activity Contaminates Water Despite Best Practices," *The Californian*, August 15, 2017, https://www.thecalifornian.com/story/news/2017/08/15/water-near-farms-often-contaminated-nitrates-coliform-bacteria/571000001/.

72   Ibid. EPA, "Drinking Water Contaminants—Standards and Regulations," https://www.epa.gov/dwstandardsregulations (accessed September 18, 2018).

73   EPA, Office of Water, *Environmental Assessment of Proposed Revisions to the National Pollutant Discharge Elimination System Regulation and Effluent Limitations Guidelines for Concentrated Animal Feeding Operations*, January 2001, https://www3.epa.gov/npdes/pubs/cafo_proposed_env_assess_ch1-3.pdf. National Cancer Institute, "Nitrate," February 2018, progressreport.cancer.gov/prevention/nitrate.

74   EPA, "National Primary Drinking Water Regulations," https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations#Inorganic.

75   Environmental Working Group, "Nitrate," Tap Water Database, July 2017, www.ewg.org/tapwater/reviewed-nitrate.php#.WsJdr4hzaUk.

76   Anne W. Schechinger and Craig Cox, "America's Nitrate Habit Is Costly and Dangerous," Environmental Working Group, October 2, 2018, https://www.ewg.org/research/nitratecost/.

77   Leslie A. DeSimone et al., "Quality of Water From Domestic Wells in Principal Aquifers of the United States, 1991–2004," U.S. Geological Survey, March 2009, pubs.usgs.gov/circ/circ1332/.

78   Ibid. at 2, 25-27.

79   Chris Augustin, Charles Stoltenow, and Ron Wiederholt, "Biosecure Nutrient Management Practices, 1."

80   Brown, Vence & Associates, *Task 2 Report: Title 27 Effectiveness to Protect Groundwater Quality*, 22, https://www.waterboards.ca.gov/rwqcb5/water_issues/confined_animal_facilities/library/bva_final_task2_rpt_ess_ctns1_6.pdf (accessed July 20, 2018).

81   Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 9.

82   Coburn Dukehart, "Fecal Microbes Found in 60 Percent of Sampled Wells, Raising Concerns About Dairy Manure, Septic Waste," WisconsinWatch, June 8, 2017, https://www.wisconsinwatch.org/2017/06/fecal-microbes-found-in-60-percent-of-sampled-wells-raising-concerns-about-dairy-manure-septic-waste/.

83   Ibid.

84   Centers for Disease Control and Prevention, Parasites: "Cryptosporidium," https://www.cdc.gov/parasites/crypto/illness.html (accessed February 5, 2019).

85   Coburn Dukehart, "Fecal Microbes Found in 60 Percent of Sampled Wells."

86   Midwest Environmental Advocates, "Safe Drinking Water Act Petition to EPA," http://midwestadvocates.org/issues-actions/actions/safe-drinking-water-act-petition-to-epa-1/.

87   Chuck Quirmbach, "DNR Offers Bottled Drinking Water to Residents With Contaminated Wells," Wisconsin Public Radio, May 10, 2017, https://www.wpr.org/dnr-offers-bottled-drinking-water-residents-contaminated-wells.

88   "In the Matter of the Wisconsin Pollutant Discharge Elimination System Permit No. WI-0059536-03-0 (WPDES Permit) Issued to Kinnard Farms, Inc., Town of Lincoln, Kewaunee County," 2014 WL 5797722, at 12.

89   State ex rel. *Dep't. of Nat. Res. v. Wisconsin Court of Appeals*, District IV, 2018 WI 25, 380 Wis. 2d 354, 909 N.W.2d 114.

90   Centers for Disease Control and Prevention, *Antibiotic Resistance Threats in the United States, 2013*, 5, April 23, 2013, https://www.cdc.gov/drugresistance/threat-report-2013/pdf/ar-threats-2013-508.pdf#page=5. World Health Organization, "Antimicrobial Resistance," February 15, 2018, http://www.who.int/mediacentre/factsheets/fs194/en/.

91   World Health Organization, "Antimicrobial Resistance."

92   US Food and Grug Administration, 2017 Summary Report on Antimicrobials Sold or Distributed for Use in Food-Producing Animals (Dec. 2018) 19, https://www.fda.gov/downloads/ForIndustry/UserFees/AnimalDrugUserFeeActADUFA/UCM628538.pdf

93   Carrie Hribar, Understanding Concentrated Animal Feeding Operations, 10, https://www.cdc.gov/nceh/ehs/docs/understanding_cafos_nalboh.pdf.

94   K. Kumar et al., "Antibiotic Uptake by Plants from Soil Fertilized With Animal Manure," *Journal of Environmental Quality* 34, no. 6 (Nov.–Dec.2005): 2082-85. US Environmental Protection Agency, *National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitations Guidelines and Standards for CAFOs*, 43-44, 50, December 2000, https://www3.epa.gov/npdes/pubs/cafo_ch5.pdf.

95   K. Kumar et al., "Antibiotic Uptake by Plants."

96   Dawn Undurraga, "Supermarket Meat Still Superbugged, Federal Data Show," Environmental Working Group, June 28, 2018, https://www.ewg.org/research/superbugs/. EWG calculations based on data from U.S. Food and Drug Administration, "National Antimicrobial Resistance Monitoring System 2015 Integrated Report," October 23, 2017, https://www.fda.gov/AnimalVeterinary/SafetyHealth/AntimicrobialResistance/NationalAntimicrobialResistanceMonitoringSystem/ucm059103.htm.

97   Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 5.

98   Abel Russ and Eric Schaeffer, *Ammonia Emissions From Broiler Operations Higher Than Previously Thought*, 2, Environmental Integrity Project, January 2018, http://www.environmentalintegrity.org/wp-content/uploads/2017/02/Ammonia-Report.pdf.

99   James M. MacDonald, "Financial Risks and Incomes in Contract Broiler Production," U.S. Department of Agriculture Economic Research Service, August 2014), https://www.ers.usda.gov/amber-waves/2014/august/financial-risks-and-incomes-in-contract-broiler-production/.

100 Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 5. Robert T. Burns et al., *Tyson Broiler Ammonia Emission Monitoring Project: Final Report*, 32, May 1, 2007, https://lib.dr.iastate.edu/cgi/viewcontent.cgi?referer=https://www.google.com/&httpsredir=1&article=1014&context=abe_eng_reports.

101 Robert T. Burns et al., *Tyson Broiler Ammonia Emission Monitoring Project*.

102 Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 4-5.

103 Ibid.

104 Ibid. at 5-6.

105 Ibid. at 6-7.

106 James A. Merchant et al. "Asthma and Farm Exposures in a Cohort of Rural Iowa Children," *Environmental Health Perspectives* 113, no. 3 (March 2005): 350, 355 (Figure 1), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1253764/.

107 Carrie Hribar, *Understanding Concentrated Animal Feeding Operations*, 6.

108 Steve Wing and Susanne Wolf, "Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents," *Environmental Health Perspectives* 108, no. 3 (March 2000). Kendall Thu et al., "A Control Study of the Physical and Mental Health of Residents Living Near a Large-Scale Swine Operation," *Journal of Agricultural Safety and Health* 3, no. 1 (1997): 13-26. Susan Schiffman et al., "The Effect of Environmental Odors Emanating From Commercial Swine Operations on the Mood of Nearby Residents," *Brain Research Bulletin* 37, no. 4 (1995): 369-75.

109 Wendee Nicole, "CAFOs and Environmental Justice."

110 The vast majority of industrial swine facilities in North Carolina operate under a state swine permit; approximately one dozen have obtained a federal NPDES permit. North Carolina Department of Environmental Quality, "Animal Facility Map—List of Permitted Animal Facilities," https://files.nc.gov/ncdeq/List_O_ fPermitted_Animal_Facilities%201-26-2018.xls?SdODtHdc46AxmsgSZ0z_o0dLzRFbNgZs (accessed January 18, 2019). According to North Carolina's permit database, at their full permitted capacity, the state's swine operations could house more than 15 million swine. However, farms may not always have swine present or may not operate at their permitted capacity. The latest U.S. Census of Agriculture estimates that there are approximately nine million swine being raised in North Carolina at any given time. *See* USDA, National Agricultural Statistics Service, Quick Stats, 2012, https://quickstats.nass.usda.gov/#BBCF08EB-60FA- 3108-AEEC-8B4B0D143668.

111 Anne Blythe, "Jury Awards More Than $25 Million to Duplin County Couple in Hog-Farm Case," *News & Observer*, June 29, 2018, https://www.newsobserver.com/news/local/article214096384.html.

112 Wendee Nicole, "CAFOs and Environmental Justice."

113 Craig Jarvis, "Hog Farm's Neighbors Not Entitled to Punitive Damages, Judge Rules," *News & Observer*, December 13, 2018, https://www.newsobserver.com/news/politics-government/state-politics/article223044710.html.

114 Ibid.

115 Sara Brown, "N.C. Legislators Override Veto of Hog Farm Nuisance Protection Bill," *AgPro*, June 28, 2018, https://www.agprofessional.com/article/nc-legislators-override-veto-hog-farm-nuisance-protection-bill.

116 Associated Press, "Environmentalists: Manure Spills Endangering Wisconsin Water," *Ag Web*, https://www.agweb.com/article/environmentalists-manure-spills-endangering-wisconsin-water-naa-associated-press/ (accessed July 20, 2018).

117 Associated Press, "Ohio Hands Out Fines Over Fish Kills Caused by Farm Manure," WTOP, October 11, 2017, https://www.apnews.com/a90c9dcdc7e64164939ef465cdd8cf08. "Crews Cleaning Up Manure Spill; Fish Kill Reported in Nearby Creek," WGAL, October 20, 2017, http://www.wgal.com/article/crews-cleaning-up-manure-spill-fish-kill-reported-in-nearby-creek/13059548.

118 Brian DeVore, "Manure Matters," *Minnesota Conservation Volunteer*, Minnesota Department of Natural Resources, July-August 2002, https://webapps8.dnr.state.mn.us/volunteer_index/past_issues/issue_pdf?issue_id=66.

119 EPA, *Literature Review of Contaminants in Livestock and Poultry Manure and Implications for Water Quality*, 49, July 2013, https://nepis.epa.gov/EPA/html/DLwait.htm?url=/Exe/ZyPDF.cgi/P100H2NI.PDF?Dockey=P100H2NI.PDF.

120 David Jackson and Gary Marx, "Spills of Pig Waste Kill Hundreds of Thousands of Fish in Illinois," *Chicago Tribune*, August 5, 2016, http://www.chicagotribune.com/news/watchdog/pork/ct-pig-farms-pollution-met-20160802-story.html.

121 Ibid.

122 Ibid.

123 EPA, "State Compliance Monitoring Expectations," https://echo.epa.gov/trends/comparative-maps-dashboards/state-compliance-monitoring-expectations (accessed July 18, 2018).

124 EPA, "National Enforcement Initiative: Preventing Animal Waste From Contaminating Surface and Ground Water," https://www.epa.gov/enforcement/national-enforcement-initiative-preventing-animal-waste-contaminating-surface-and-ground (accessed July 18, 2018).

125 EPA, "*NPDES CAFO Permitting Status Report—National Summary, Endyear 2017,*" December 31, 2016, https://www.epa.gov/sites/production/files/2018-05/documents/tracksum_endyear_2017.pdf (accessed September 22, 2018).

126 EPA, "National Enforcement Initiative." Christopher Ball, "EPA Announces Triennial Enforcement Priorities," Manko Gold Katcher Fox LLP, January 16, 2008, https://www.mankogold.com/publications-365.html. "EPA Announces 2012 National Enforcement Initiatives and Regional Priorities," Bingham Greenbaum Doll LLP, December 1, 2011, https://www.bgdlegal.com/newsroom-news-epa-announces-2012-national-enforcement-initiatives-and-regional-priorities.

127 EPA, "EPA Targets Clean Water Act Crimes—Illegal Pollution by Animal Feeding Operations Punished by Fines and Incarceration," *Criminal Enforcement Alert* 1, no. 3 (July 2013): 2, https://www.epa.gov/sites/production/files/documents/cr-cafo-06-13.pdf.

128 GAO, *Livestock Agriculture: Increased EPA Oversight Will Improve Environmental Program for Concentrated Animal Feeding Operations*, GAO-03-285, 3, January 2003, https://www.gao.gov/new.items/d03285.pdf. Food & Water Watch et al., *Petition to Revise the Clean Water Act Regulations for Concentrated Animal Feeding Operations*, 12–13, https://www.foodandwaterwatch.org/sites/default/files/citizens_cafo_cwa_petition.pdf (accessed July 22, 2018).

129 GAO, *Concentrated Animal Feeding Operations: EPA Needs More Information*, 4-5.

130 76 Fed. Reg. 65436.

131 GAO, *Concentrated Animal Feeding Operations: EPA Needs More Information*, 2.

132 EPA, *National Pollutant Discharge Elimination System Permit Regulation for Concentrated Animal Feeding Operations: Removal of Vacated Elements in Response to 2011 Court Decision*, 77 Fed. Reg. 44,494, 44,494, July 30, 2012 (codified at 40 C.F.R. pt. 122).

133 William M. McLaren, "The Death of the Duty to Apply: Limitations to CAFO Oversight Following Waterkeeper & National Pork Producers," *Journal of Animal & Natural Resource Law* 11 (2015): 87, 106. Terence J. Centner, "Clarifying NPDES Requirements for Concentrated Animal Feeding Operations," *Penn State Environmental Law Review* (2006): 361, 362. Food & Water Watch, "The EPA's Failure to Track Factory Farms," 1, August 2013, https://www.foodandwaterwatch.org/sites/default/files/EPA%20Factory%20Farms%20IB%20Aug%202013_0.pdf (stating that "as of 2011, the EPA estimated that only 41 percent of NPDES-defined CAFOs actually had NPDES permits").

134 EPA, "NPDES State Program Information," https://www.epa.gov/npdes/npdes-state-program-information (accessed July 22, 2018). Food & Water Watch et al., *Petition to Revise the Clean Water Act Regulations*, 12–13.

135 GAO, *Concentrated Animal Feeding Operations: EPA Needs More Information*, 31.

136 EPA, "Proposed National Pollutant Discharge Elimination System Concentrated Animal Feeding Operation Reporting Rule," Questions and Answers, October 2011, http://www.epa.gov/npdes/pubs/2011_npdes_cafo_qa.pdf.

137 EPA, "*National Pollutant Discharge Elimination System (NPDES) Concentrated Animal Feeding Operation (CAFO) Reporting Rule,*" 76 Fed. Reg. 65,431, 65,435-37 (proposed Oct. 21, 2011) (to be codified at 40 C.F.R. pts. 9, 122).

138 76 Fed. Reg. 65,437 (October 21, 2011).

139 76 Fed. Reg. 65,435 (October 21, 2011).

140 EPA, "*National Pollutant Discharge Elimination System (NPDES) Concentrated Animal Feeding Operation (CAFO) Reporting Rule,*" 77 Fed. Reg. 42,679, 42,679 (withdrawn July 20, 2012).

141 Ibid. at 42,681.

142 Ibid.

143 Jon Devine, "Seeking Industrial Livestock Information Is Not 'Extreme.' It's Needed to Protect U.S. Waterways and Human Health," *NRDC Expert Blog*, February 21, 2013, https://www.nrdc.org/experts/jon-devine/seeking-industrial-livestock-information-not-extreme-its-needed-protect-us.

144 In addition to the data collected from states, EPA provided spreadsheets containing information that it had compiled from two publicly available sources: online state databases and EPA's own PCS/ICIS system. Instead of using EPA's data from these publicly available sources, NRDC relied on its own more recent searches of the same sources.

145 EPA, "PCS-ISIS Search," https://www.epa.gov/enviro/pcs-icis-search (accessed July 22, 2018).

146 Alabama Department of Environmental Management, "Animal Feeding Operations," http://www.adem.state.al.us/programs/water/cafo.cnt (accessed July 22, 2018). Arkansas Department of Environmental Quality, "ADEQ Facility and Permit Summary," https://www.adeq.state.ar.us/home/pdssql/pds.aspx (accessed July 22, 2018). California Environmental Protection Agency, State Water Resources Control Board, "Regulated Facility Report," https://ciwqs.waterboards.ca.gov/ciwqs/readOnly/ CiwqsReportServlet?inCommand=reset&reportName=RegulatedFacility (accessed July 22, 2018). Florida Department of Environmental Protection, "Wastewater Facility Information," http://www.dep.state.fl.us/water/wastewater/facinfo.htm (accessed July 22, 2018). Indiana Department of Environmental Management, "Confined Feeding Operations," https://vfc.idem.in.gov/DocumentSearch.aspx (accessed July 22, 2018). Iowa Department of Natural Resources, "Animal Feeding Operation," https://programs.iowadnr.gov/animalfeedingoperations/FacilitySearch.aspx (accessed July 22, 2018). Maryland Department of the Environment, "Permit Application Database Search," http://www.mde.state.md.us/programs/Permits/Pages/SB47.aspx (accessed July 22, 2018). Mississippi Department of Environmental Quality, OPC Environmental Permits Division, "Active Permit & Coverage Search," http://opc.deq.state.ms.us/search_ai_alt.aspx#grid (accessed July 22, 2018). New Jersey Department of Environmental Protection, "NJPDES Active Permit List," https://www13.state.nj.us/DataMiner (accessed July 22, 2018). North Carolina Department of Environmental Quality, "List of Permitted Animal Facilities," https://deq.nc.gov/about/divisions/water-resources/water-resources-permits/wastewater-branch/animal-feeding-operation-permits/animal-facility-map (accessed July 22, 2018). Tennessee Department of Environment and Conservation, "DWR Permits in Tennessee," http://environment-online.tn.gov:8080/pls/enf_reports/f?p=9034:34001:0 (accessed July 22, 2018). Texas Commission on Environmental Quality, "Water Quality General Permits & Registration Search," http://www2.tceq.texas.gov/wq_dpa/index.cfm (accessed July 22, 2018). State of Washington Department of Ecology, "Facility/ Site Search," https://fortress.wa.gov/ecy/facilitysite/SearchData/ShowSearch.aspx?ModuleType=FacilitySite&RecordSearchMode=New (accessed July 22, 2018). Wisconsin Department of Natural Resources, "CAFO Permittees," http://dnr.wi.gov/topic/AgBusiness/data/CAFO/cafo_all.asp?FULL=1 (accessed July 22, 2018).

147 By "readily available," we mean that a user could download information in a batch (.csv or excel) file or otherwise copy and paste it from the website in batches. Some state websites contained additional pieces of information about CAFOs on separate pages for each facility, or linked to individual PDFs of permitting documents for each facility. This type of information was not considered readily available. In the cases of Iowa and Texas, it was possible to write a program to obtain information from separate pages, and this information was included in the database.

148 EPA, "Enforcement and Compliance History Online (ECHO)," http://echo.epa.gov/?redirect=echo (accessed July 22, 2018).

149 EPA, "*NPDES CAFO Rule Implementation Status—National Summary, Midyear 2012,*" June 30, 2012, https://www3.epa.gov/npdes/pubs/tracksum%20midyear2012_ publish.pdf (accessed July 22, 2018).

150 Ibid.

151 Food & Water Watch, "Factory Farm Map," Pennsylvania, https://www.factoryfarmmap.org/#animal:all;location:PA;year:2012 (accessed July 22, 2018).

152 Earthjustice, *Re: Complaint Under Title VI of the Civil Rights Act of 1964, 42 U.S.C. § 2000d, 40 C.F.R. Part 7*, letter to Gina McCarthy, EPA administrator, and Velveta Golightly-Howell, director, EPA Office of Civil Rights, September 3, 2014, http://blogs.law.unc.edu/documents/civilrights/20140902complaint.pdf.

153 Interpretive Statement on Application of the Clean Water Act National Pollutant Discharge Elimination System Program to Releases of Pollutants From a Point Source to Groundwater, 84 Fed. Reg. 16,810 (Apr. 23, 2019).

154 Ibid. at 7127. EPA, *National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitations Guidelines and Standards for Concentrated Animal Feeding Operations*, 66 Fed. Reg. 2,960, 3,017 (Jan. 12, 2001) (stating, "As a legal and factual matter, EPA has made a determination that, in general, collected or channeled pollutants conveyed to surface waters via groundwater can constitute a discharge subject to the Clean Water Act").

155 *Riverkeeper, Inc. v. Seggos*, No. 902103-17, Decision and Order (N.Y. Sup. Ct. Apr. 23, 2018), https://waterkeeper.org/wp-content/uploads/2018/04/Riverkeeper-v.-Seggos-DO.4.24.2018.pdf.

156 EPA, "NPDES CAFO Permitting Status Report, National Summary, Endyear 2017," (December 31, 2017), https://www.epa.gov/sites/production/files/2018-05/ documents/tracksum_endyear_2017.pdf.

157 Environmental Law Institute, "State Constraints: State-Imposed Limitations on the Authority of Agencies to Regulate Waters Beyond the Scope of the Federal Clean Water Act," 2, May 2013, https://www.eli.org/sites/default/files/eli-pubs/d23-04.pdf. (stating, "The Clean Water Act establishes national minimum standards— essentially, a stringency 'floor'—beneath which states are not allowed to fall in their protection of water quality. States may, however, protect their waters more rigorously").

Cancer Trends Progress Report

Online Summary of Trends in US Cancer Control Measures
Nitrate

Data Up to Date as of:
July 2021

Background
Nitrates and nitrites are nitrogen-oxygen chemical units that naturally occur in soil, water, and some foods. When taken into the body by drinking water and through other dietary sources, nitrate and nitrite can react with amines and amides to form N-nitroso compounds (NOC), which are known to cause cancer in animals and may cause cancer in humans. Excessive nitrate or nitrate exposure can also result in acute acquired methemoglobinemia, a blood abnormality that causes blood to lose its ability to carry oxygen to tissues (anoxia). This is especially dangerous in infants younger than 4 months of age.

The biggest source of nitrate exposure is dietary consumption of certain types of vegetables which are naturally high in nitrate. However, these vegetables also contain compounds that prevent the formation of NOCs. Studies assessing connections between nitrate and cancer in humans have focused on excess exposure from drinking water or food grown in areas where use of nitrogen-based fertilizers is common. Some of the highest levels of nitrate have been measured in shallow wells and surface water supplies that are subject to runoff from nitrogen fertilizers and confined animal feedlot operations and resulting excrement and contamination from leaking septic tanks and sewage. In addition, workers who manufacture these fertilizers can have high exposures to dusts that contain nitrate. Oral tobacco also may contribute to nitrate intake, but is minor compared to diet or contaminated drinking water.

Studies have shown increased risks of colon, kidney, and stomach cancer among people with higher ingestion of water nitrate and higher meat intake compared with low intakes of both, a dietary pattern that results in increased NOC formation. Other studies have shown modest evidence that higher nitrate intake can increase the risk of thyroid cancer and ovarian cancer among women.

Measure
We present exposure data on the 95th percentile of the population, representing people with the greatest exposure. The 95th percentile level means that 95% of the population has concentrations below that level. Public health officials use such reference values to determine whether groups of people are experiencing an exposure that is unusual compared with an exposure experienced by the rest of the population. [Citation]

To calculate whether the differences between 95th percentiles for two different time points is statistically significant, we used a different statistical methodology than that used by the National Center for Environmental Health, who publishes the National Report on Human Exposure to Environmental Chemicals from where our data are derived. Our estimates may differ slightly from those in the original report due to differences in statistical procedures used. [Methodology]

Healthy People 2030 Target
There are no Healthy People 2030 targets regarding nitrate.

Healthy People 2030 is a set of goals set forth by the Department of Health and Human Services.

Data Source
Centers for Disease Control and Prevention, National Center for Health Statistics, National Health and Nutrition Examination Survey.

Trends and Most Recent Estimates

By Sex

95th percentile for urinary (creatinine corrected) concentrations (mg/g of creatinine) of nitrate among persons aged 6 years and older by sex, 2001-2016

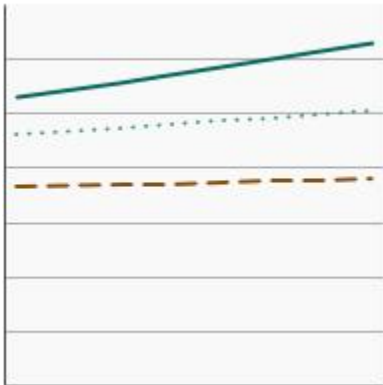| Overview Graph	Detailed Trend Graphs	Most Recent Estimates (2015 to 2016)	
		Dependent Variable	95% Confidence Interval
	Both Sexes	137.6	119.1 - 158.1
	Male	122.1	107.4 - 154.9
	Female	147.1	121.4 - 176.9

By Race/Ethnicity

95th percentile for urinary (creatinine corrected) concentrations (mg/g of creatinine) of nitrate among persons aged 6 years and older by race/ethnicity, 2001-2016

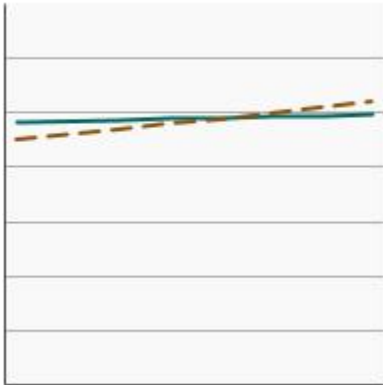| Overview Graph	Detailed Trend Graphs	Most Recent Estimates (2015 to 2016)	
		Dependent Variable	95% Confidence Interval
	All Races	137.6	119.1 - 158.1
	Non-Hispanic White	142.3	112.8 - 164.2
	Non-Hispanic Black	98.6	87.7 - 104.5
	Hispanic	120.7	107.1 - 139.9

By Age

95th percentile for urinary (creatinine corrected) concentrations (mg/g of creatinine) of nitrate among persons aged 6 years and older by age, 2001-2016

Overview Graph	Detailed Trend Graphs	Most Recent Estimates (2015 to 2016)	
		Dependent Variable	95% Confidence Interval
	Ages 6-11	175.4	133.2 - 220.3
	Ages 12-19	103.0	83.5 - 117.7
	Ages 20+	125.7	108.0 - 151.6

By Poverty Income Level

95th percentile for urinary (creatinine corrected) concentrations (mg/g of creatinine) of nitrate among persons aged 6 years and older by poverty income level, 2001-2016

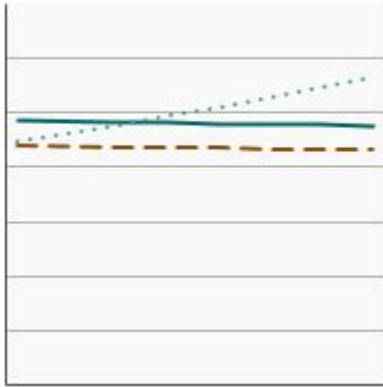| Overview Graph	Detailed Trend Graphs	Most Recent Estimates (2015 to 2016)	
		Dependent Variable	95% Confidence Interval
	< 200% of the federal poverty level	129.5	115.4 - 146.8
	>= 200% of the federal poverty level	143.4	117.5 - 164.3

By Education Level

95th percentile for urinary (creatinine corrected) concentrations (mg/g of creatinine) of nitrate among persons aged 20 years and older by highest level of education obtained, 2001-2016

Overview Graph	Detailed Trend Graphs	Most Recent Estimates (2015 to 2016)	
		Dependent Variable	95% Confidence Interval
	Less than High School	119.4	102.0 - 161.2
	High School	90.4	81.2 - 152.2
	Greater than High School	135.9	113.1 - 162.4

Additional Information on Nitrate