From: sandylakeerie@aol.com <sandylakeerie@aol.com>
Sent: Monday, November 16, 2020 5:33 PM
To: Kyle Dreyfuss-Wells <Dreyfuss-WellsK@neorsd.org>
Subject: EXTERNAL: Followup Inof Lake Erie

Hi Kyle
Thanks for the call.
Sorry I was late.
One thing I did not discuss is that Michigan is proposing to use the GLWQA Annex 4 in place of a TMDL in the Michigan portion of Lake Erie/Lake Erie watershed.

The GLWQA Annex 4 as you know has not legal recourse if the goals/policies do not work where the Clean Water Act TMDL Implementation process does(as you know this was the process to take you guys to court.

And as another matter to discuss. The Chesapeake recover process has been going in the right direction because of a multi faceted TMDL. To keep the public engagement, the University of Maryland developed a Report Card which has helped with media, public engagement and a science data based water quality approach. I was able to get a Western Lake Erie report card - completed this year funded by the City of Toledo, City of Oregon, and Lucas County- the next(two year cylce) report care funding is underway by the same entities but this time, the lead is transitioned to the University of Toledo, with the University of Maryland helping in the transition(the next the University of Maryland will not be needed). Through the years, I have tried to get data on the Central Basin of Lake Erie and in particular, the size and location of the dead zone but it has been increasingly very difficult. Would you entertain helping to fund a central Lake Erie basin report card(it would include the central basin tributaries?

Attached are the 2019 CAFO/Manure/Maumee studies, the Pat Glibert research report, and an Iowa news article. Also attaching information on SERA 17in 2017 which is the backdrop for USDA NRCS 590, also attached, ag funding which also provides the policies states use for manure manage. And attached is the IJC GL Manure Management report that is being considered to enter another phase in 2021.

Please let me know if there are questions etc.

Take care

Sandy Bihn

Sandy Bihn
Executive Director
Lake Erie Waterkeeper
3900 N. Summit Bldg. 2
Toledo, Ohio 43611
www.lakeeriewaterkeeper.org
sandylakeerie@aol.com
Text/Cell 419-367-1691

METHODS AND RESULTS: EWG AND ELPC ANALYSIS OF AFOS IN MAUMEE RIVER BASIN

TUESDAY, APRIL 9, 2019

By Sarah Porter, Senior Geospatial Analyst and Project Manager

Special thanks to Stuart Flack, Lucas Stephens and Madeline Fleisher with the Environmental Law & Policy Center and Sandy Bihn with Lake Erie Waterkeeper

Harmful algae blooms in Lake Erie began showing up in the mid-1990s and have increased in severity over time (D'Anglada et al., 2018). These blooms are caused by excess phosphorus, primarily dissolved phosphorus, which is delivered to the lake from upstream tributary watersheds. Nonpoint agricultural release is recognized to be the single largest source of excess phosphorus to western Lake Erie (IJC, 2018), with the two primary sources from agriculture being the application of commercial fertilizer and manure.

The Maumee River watershed basin has been identified as the largest contributor of phosphorus to Lake Erie, delivering an estimated 30 percent of total phosphorus coming to the lake from the U.S. and Canada (Maccoux, 2018). Commercial fertilizer has been the primary focus of research in the region.

The International Joint Commission (2018) estimates that 80 percent of agricultural phosphorus generated in the Western Lake Erie Basin, or WLEB, derives from commercial fertilizer, whereas approximately 20 percent derives from manure from animal feeding operations (AFOs) (IJC, 2018).

Although trends point to decreasing commercial fertilizer application in the WLEB since the 1990s, dissolved phosphorus loads to the lake continue to rise, and blooms continue to increase in severity. Legacy phosphorus in the soil, tile drainage and tillage practices are leading current hypotheses to explain these increasing dissolved phosphorus loads (EPA, 2010).

Manure application from AFOs is assumed to have remained constant over time (IJC, 2018). This is mainly due to a lack of reliable, publicly available information about where and how many of these facilities exist, and the amount of manure and phosphorus they produce.

Animal operations above a certain size threshold are subject to regulation by government agencies. Many AFOs are below this threshold, however, and therefore do not need to apply for a permit that would provide more detailed information about the location, number of animals and other data. As a result, academics and agency officials have had little detailed information about the scope of livestock production in the watershed.

In addition, regulations vary by state, making consolidation of data across state lines challenging. The purpose of this study is to use remote sensing to map all AFOs in the Maumee River basin between 2005 and 2018. In addition, we estimate the number of animals housed at these facilities and the amount of manure and phosphorus they produce. It is our hope that this information will enhance our understanding of the role that AFOs play in the generation of phosphorus in the Maumee River Basin.

METHODS

Locating Animal Feeding Operations

National Agriculture Imagery Program, or NAIP, aerial photography was used to visually locate AFOs in the Maumee Basin. Consistent imagery was available across the study area beginning in 2005, the base year for this study. Due to alternating years of NAIP image collection after 2005, AFOs were categorized into the following periods for time of construction:

- Present in 2005.
- 2005 to 2010.
- 2010 to 2015.
- 2015 to 2018.

To capture very recent AFO construction (late 2018 to January 2019), Planet satellite imagery was used to supplement aerial photography. Several attributes were recorded for each facility, including the number of barns and their total square footage (as calculated by Environmental Law Policy Center), animal type (poultry, swine, beef cattle or dairy cattle) and the year of expansion, if any.

Animal type was assigned to each facility using the best judgment of the geographic information system, or GIS, analyst, based on a number of attributes unique to each facility, including the size and shape of each barn, the presence and number of feed bins, the location of fans, and the presence of lagoons and of visually identifiable animals.

We assigned animal type using permit data when available for a facility. We also used Google Street View, and separate reviewers performed several rounds of quality control. Despite this intensive process, visual assignment proved challenging in some cases, and there may be instances of misidentification in our analysis. In addition, we removed facilities from analysis if they appeared to be abandoned, as evidenced by dilapidated roofs or removal of infrastructure.

Permit Data

We obtained state permit data for facilities in the Maumee Basin from the following sources:

- Ohio: 2018 data (CAFF and NPDES permits, received March 19, 2019), obtained from the Ohio Department of Agriculture in spreadsheet form, with location information for each facility. Locations were geolocated to the nearest mapped facility.
- Michigan: 2018 data (NPDES permits) obtained from the MIWaters website. Permit data were matched from the interactive website to mapped facilities.
- Indiana: 2018 data (CFO permits, received Oc. 31, 2018) obtained from the Indiana Department of Environmental Management, or IDEM. Data were provided at a township scale. Where possible, permit data were matched to mapped facilities. This occurred when a single facility was permitted in a township and only one facility of the same animal type was mapped in that township. As this could not be performed for all facilities in Indiana, the permit status for all facilities in Indiana was considered “unknown” for the remainder of the analysis.

Assigning Animal Counts

Animal counts were estimated for each facility by dividing the mapped square footage of each barn by a square footage per animal. We obtained recommended square footage per animal from a literature review of standards, and they are listed below, along with their source.

Table 1. Square footage per animal type as derived from industry, academic or government guidelines

	Square footage allotted to animal type	Source
Dairy	80 (based on 1100 - 1300 lb heifer)	Penn State Extensior
Cattle	35 (average of access to yard and no access to yard)	Midwest Planning Serv
Swine	7.4 (average of optimal economic and productivity)	National Pork Board
Poultry	.465 (layers)	United Egg Producers

Source: Penn State Extension, Midwest Planning Service, National Pork Board and United Egg Producers

Challenges With Poultry Animal Counts

Poultry production type (broilers, pullets, turkeys or egg layers) was unknown for each poultry facility Although square footage allotted per bird will vary based on production type, we applied guidelines on square footage for laying hens (67 square inches) to all poultry facilities. This choice was guided by data from the USDA 2012 Agricultural Census.

County-level inventory estimates for "pullets for laying flock replacement," "broilers and other meat-type chickens," "turkeys," and "layers," were added up for each county that touched the Maumee, then multiplied by the percentage of the county that lies within the watershed boundary.

Results showed that laying hens are the dominant poultry type (75 percent), followed by pullets (14 percent), turkeys (10 percent) and broilers (1 percent). Although the use of a single square footage per bird will introduce bias among the various poultry types, the inability to distinguish poultry type from aerial imagery required us to make certain assumptions. These biases include underestimating the number of laying hens, due to the 67 square inches per bird being applied to the building footprint and not accounting for modern high-rise laying houses, in which cage systems consist of enclosures arranged in rows and stacked in multiple tiers (USDA, Poultry Industry Manual).

As a result, the number of egg-laying hens in the Maumee basin and their phosphorus contribution may be seriously underestimated. Animal counts for other poultry types (pullets, broilers and turkeys) may be overestimated for barns housing these animals, as they are allocated more space per bird than the 67 square inches for layers used in this study.

Animal counts were estimated for each barn in the Maumee watershed. If a facility was permitted, animal counts from permit data were used rather than estimates using a square footage approach. This includes facilities in Indiana that could be matched to the township level permit data. Estimated animal counts in 2018 are listed in Table 2. Note that the 4,205,379-acre Maumee watershed lies primarily in Ohio (73 percent of land area), followed by Indiana (20 percent of land area) and Michigan (7 percent of land area).

Table 2. Estimated animal counts in the Maumee River Basin, as of 2018.

Estimated Animal Counts In the Maumee Basin (2018)				
	Indiana	Michigan	Ohio	Total
Dairy	12,949	15,494	69,834	98,277
Cattle	21,527	29,288	18,652	69,467
Swine	239,595	18,560	789,904	1,048,059
Poultry	4,610,857	285,076	14,323,216	19,219,149
Total	4,884,928	348,418	15,201,606	20,434,952

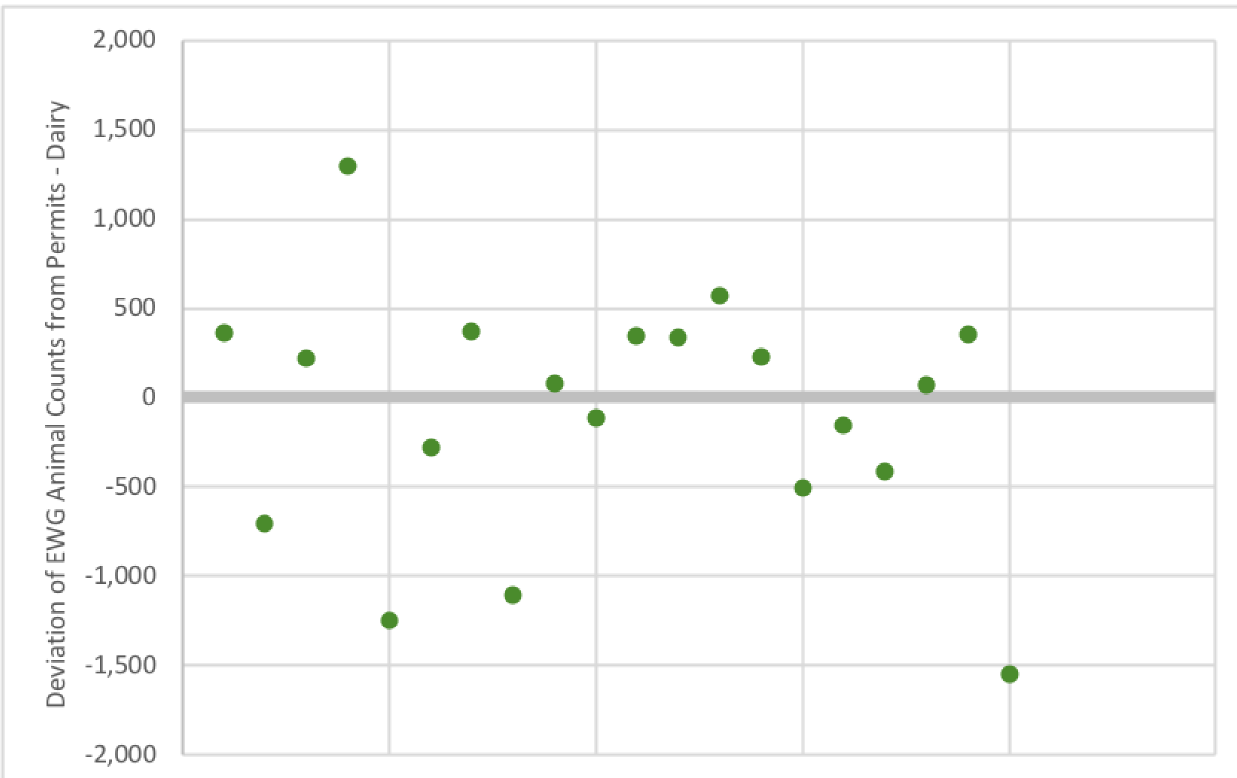
Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management and Michigan Dept. of Environmental Quality

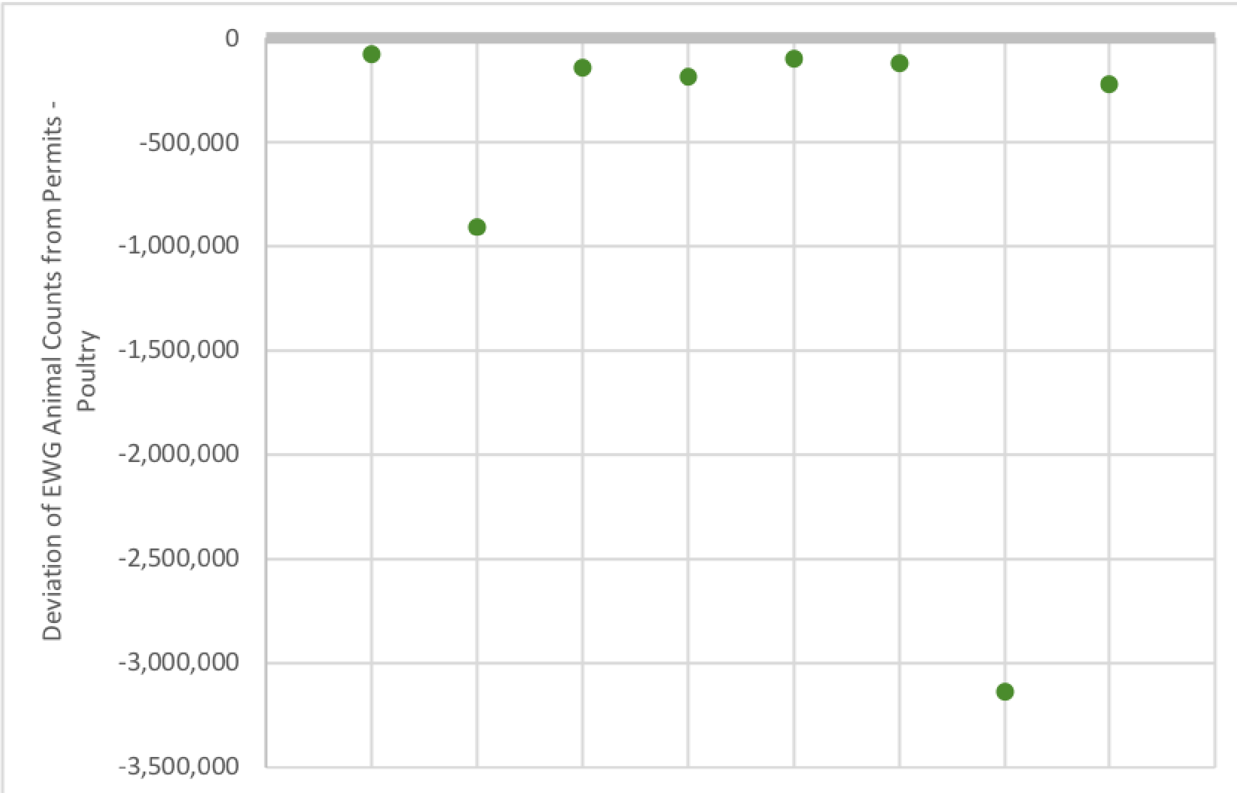
Validation

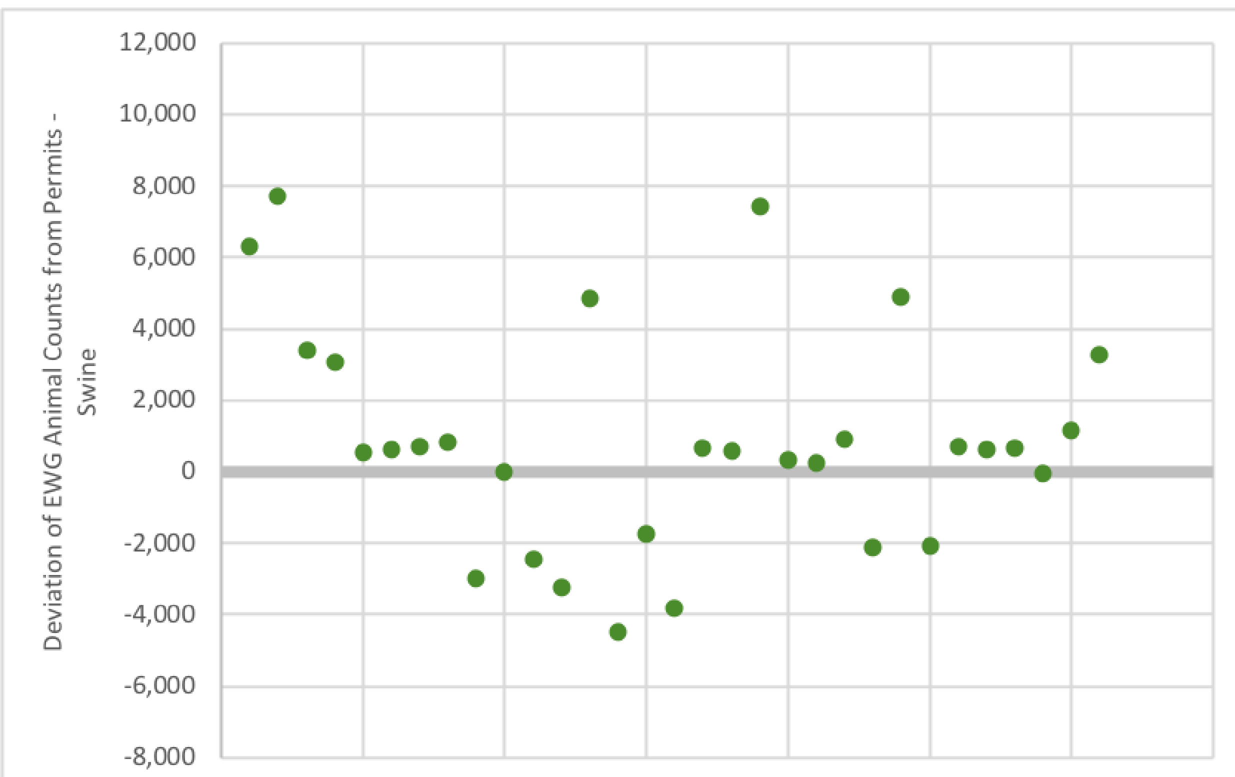
Permitted data provided a means to validate the accuracy of the square footage methodology to assign animal counts. We compared animal counts listed for the 60 permitted facilities in Ohio to what the estimated count would be using a square footage approach. Results are displayed in Figure 1 below, with estimates above the permit count shown above the x-axis and estimates below the permit count shown below the x-axis.

This was performed for dairy, poultry and swine, as there was only one permitted beef cattle facility in the Ohio portion of the Maumee. For dairy and swine, there was an approximately equal amount of over- and under-estimation for each animal type. The average of all permitted swine facilities showed an overestimation of swine by 858 animals when using a square footage approach (standard deviation of 3,108). The average of all dairy facilities showed an underestimation of dairy by 92 cows (standard deviation of 680). However, poultry animal counts were underestimated in every case using the square footage approach (n = 7 permitted poultry facilities, 6 layer, 1 pullet). The average underestimation for layers was over 600,000 birds (standard deviation of > 1 million). Although this may indicate an underestimation of poultry basinwide, it also provides a level of conservatism for estimating overall poultry counts, which will include other poultry types besides layers.

Figure 1. Dairy, Poultry and Swine animal counts using Square Footage Methods Versus Permit Data







Source: EWG and ELPC via Ohio Dept. of Agriculture

Manure Production

The Midwest Planning Service (MWPS-18) “Manure Characteristics” was used to estimate manure production values (Table 3). We used information from permit data, supplemented by the USDA 2012 Ag Census, to inform the selection of a single daily production value (for manure, N and P_20_5) for each animal type.

For beef cattle, daily production values were averaged among all animal sizes listed in Table 3. We were guided in this choice by an overall lack of information on cattle size for facilities in the Maumee, for which only three cattle permits were found. In addition, the Ag Census does not provide a means of determining the distribution of cattle size within a county.

For swine, we used permit data in Ohio to determine that growing pigs are the dominant animal type (> 90 percent of swine are greater than 55 pounds, n = 32 permits). Therefore, we chose to average production values for all sizes of swine listed (boars excluded) to represent an approximately 183 pound growing pig. Ohio permit data for dairy cattle showed that 99 percent of dairy animals (n = 17) are mature cows, which informed our decision to use manure and nutrient production values for a mature 1,400 pound dairy cow.

Manure production for laying hens was applied to all poultry rather than average values for layers and broilers, which was informed both by the dominance of laying hens in the USDA Ag Census (75 percent laying hens) and the overall underestimation of the number of chickens when compared to permit data. This resulted in a single value for manure, N and P_2O_5 production for each animal type in pounds per day (Table 4). P_2O_5 was multiplied by .44 to convert to elemental P in pounds per day (MWPS-18).

It is important to note that this number only reflects manure and nutrient production for each animal type and does not account for the addition of water for the purpose of washing or dilution. This can increase volumes of manure production by up to fourfold for liquid swine facilities (MWPS-18) but does not alter the phosphorus content of the manure. Long et al. also demonstrated that using as-excreted literature values may lead to over- or under-estimation of nutrient availability.

Table 3. MWPS Manure Production and Characteristics as produced, from MWPS-18.

		Manure Production		Nutrient Content (lb/day)	
Animal Type	**Size, lb**	**lb/day**	**gal/day**	**N**	**P205**
Dairy cattle	150	13	1.6	0.064	0.03
	250	22	2.6	0.106	0.04
	500	43	5.2	0.213	0.09
	1000	86	10.4	0.425	0.17
	1400	120	14.5	0.595	0.24
Beef cattle	500	30	3.6	0.17	0.13
	750	45	5.3	0.26	0.19
	1000	60	7.1	0.34	0.25

	1250	75	8.9	0.43	0.31
Swine:					
Nursery Pig	35	2.3	0.3	0.02	0.012
Growing Pig	65	4.2	0.5	0.03	0.022
Finishing Pig	150	9.8	1.2	0.07	0.05
	200	13.1	1.6	0.09	0.067
Gestating Sow	275	9	1.1	0.07	0.05
Sow and Litter	375	22.5	2.7	0.1	0.055
Boar	350	11.5	1.4	0.09	0.064
Poultry:					
Layers	4	0.21	0.026	0.0029	0.0025
Broilers	2	0.14	0.016	0.0017	0.0009

Source: Midwest Planning Service (MWPS-18)

Table 4. Manure and nutrient production per animal per day, adapted from MWPS-18.

	Manure Production	**Nutrient Content (lb/day)**		
Animal Type	**lb/day**	**N**	**P205**	**P**
Dairy	120	0.595	0.24	0.1056
Cattle	52.5	0.3	0.22	0.0968
Swine	10.15	0.0633	0.0426	0.0187
Poultry	0.21	0.0029	0.0025	0.0011

Source: EWG and ELPC via Midwest Planning Service

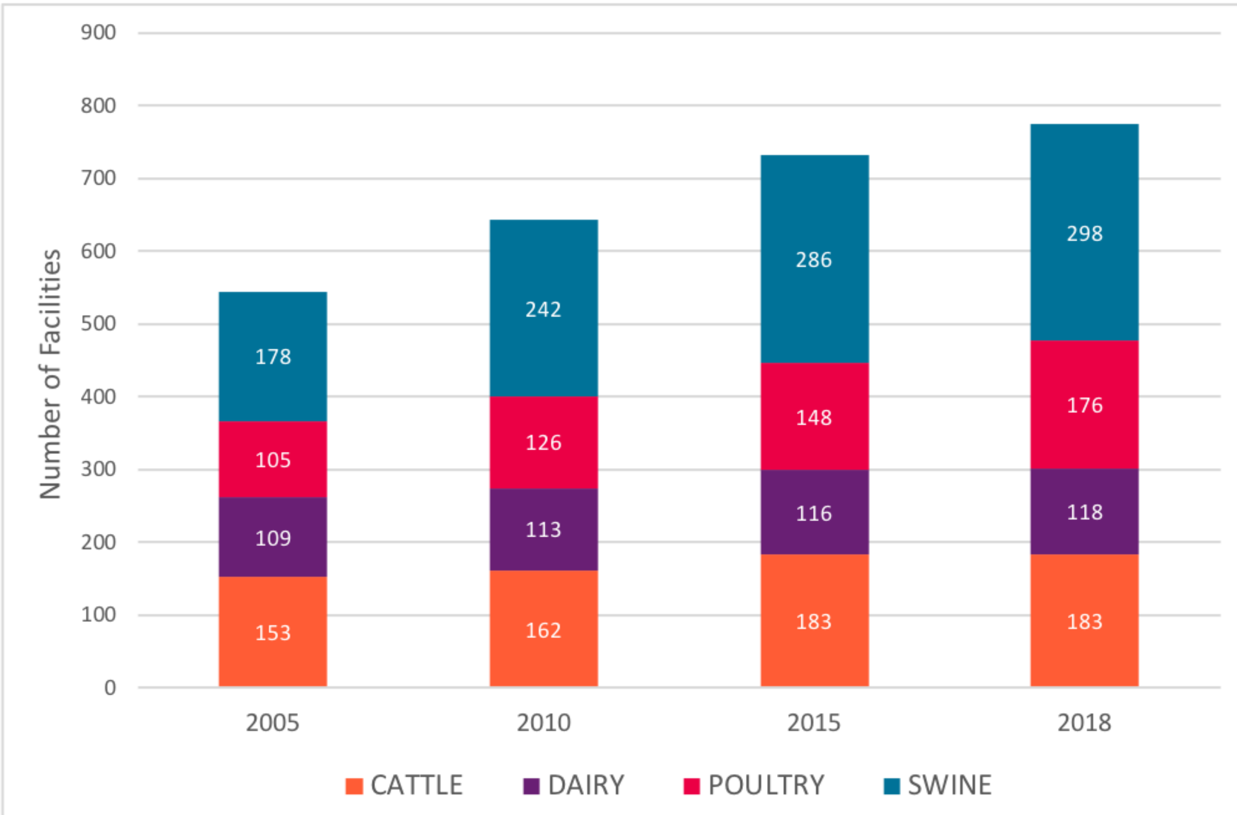
FINDINGS

Growth in Animal Feeding Operations

In 2005, we identified 545 animal feeding operations present in the Maumee River Basin. This included 178 swine, 153 cattle, 109 dairy and 105 poultry facilities. Between 2005 and 2018, 230 AFOs were constructed in the Maumee basin, equating to an average of 18 facilities added each year. The majority of growth was seen in poultry and swine, with 71 poultry facilities (31 percent of all new facilities) and 120 swine facilities (52 percent of all new facilities) constructed during this 13-year period.

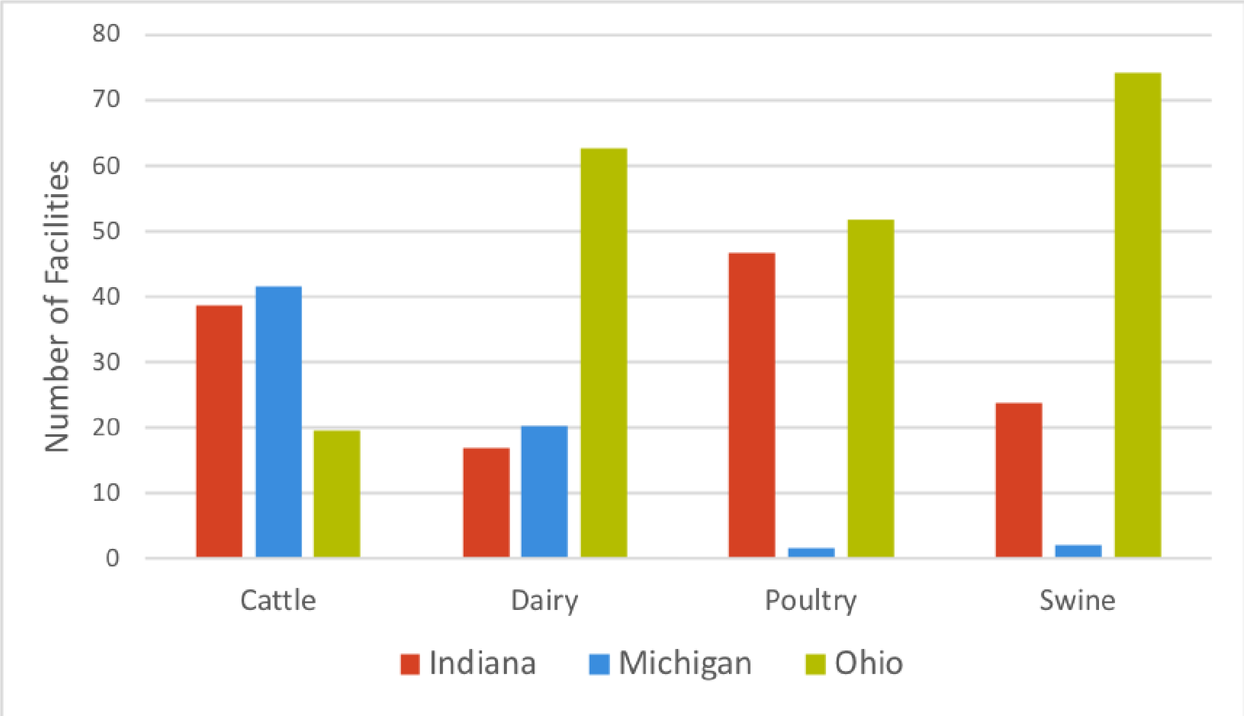
By 2018, 775 AFOs were mapped in the Maumee Basin, which included 298 swine, 183 cattle, 118 dairy and 176 poultry facilities (Figures 2 and 3). Cattle and dairy production exhibited the slowest growth, with 30 cattle facilities and only nine dairy facilities added over the 13-year period.

Figure 2. Growth in AFO facilities in the Maumee River Basin by animal type (2005-2018).



Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management and Michigan Dept. of Environmental Quality

Figure 3. Animal facilities in the Maumee River Basin by state (2018).



Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management and Michigan Dept. of Environmental Quality

Expansion and Facility Characteristics

Of the 775 facilities present in 2018, 213 (27 percent) expanded since their first year of construction, during which either more buildings were added or existing buildings increased in size. To examine the change in facility characteristics over time, mean barn size and mean number of animals at each facility were compared for facilities built before and after 2005 (Table 5).

Both mean barn size and number of animals per facility decreased for cattle, whereas mean barn size and number of animals per facility increased for dairy, poultry and swine, in some cases substantially. For dairy, poultry and swine, mean barn size increased by 61 percent, 57 percent and 75 percent. Mean number of animals at each facility (which may include multiple barns) increased by 13 percent, 22 percent and 33 percent, respectively. These findings suggest that over time, AFO barns are getting larger and more animals are being housed at a single facility. This aligns with IJC 2017 results that show increased consolidation of animal facilities on the U.S. side of the Western Lake Erie Basin.

Table 5. Characteristics of facilities constructed before and after 2005.

	Attribute	Pre-2005	Post 2005
Beef Cattle	Mean barn size (square footage)	5,740	4,171
	Mean no. of animals at each facility (all barns)	389	330
Dairy Cattle	Mean barn size (square footage)	20,971	33,660
	Mean no. of animals at each facility (all barns)	824	934

Poultry	Mean barn size (square footage)	14,141	22,268
	Mean no. of animals at each facility (all barns)	79,259	97,017
Swine	Mean barn size (square footage)	11,140	19,495
	Mean no. of animals at each facility (all barns)	3,115	4,154

Source: EWG and ELPC and Environmental Law Policy Center

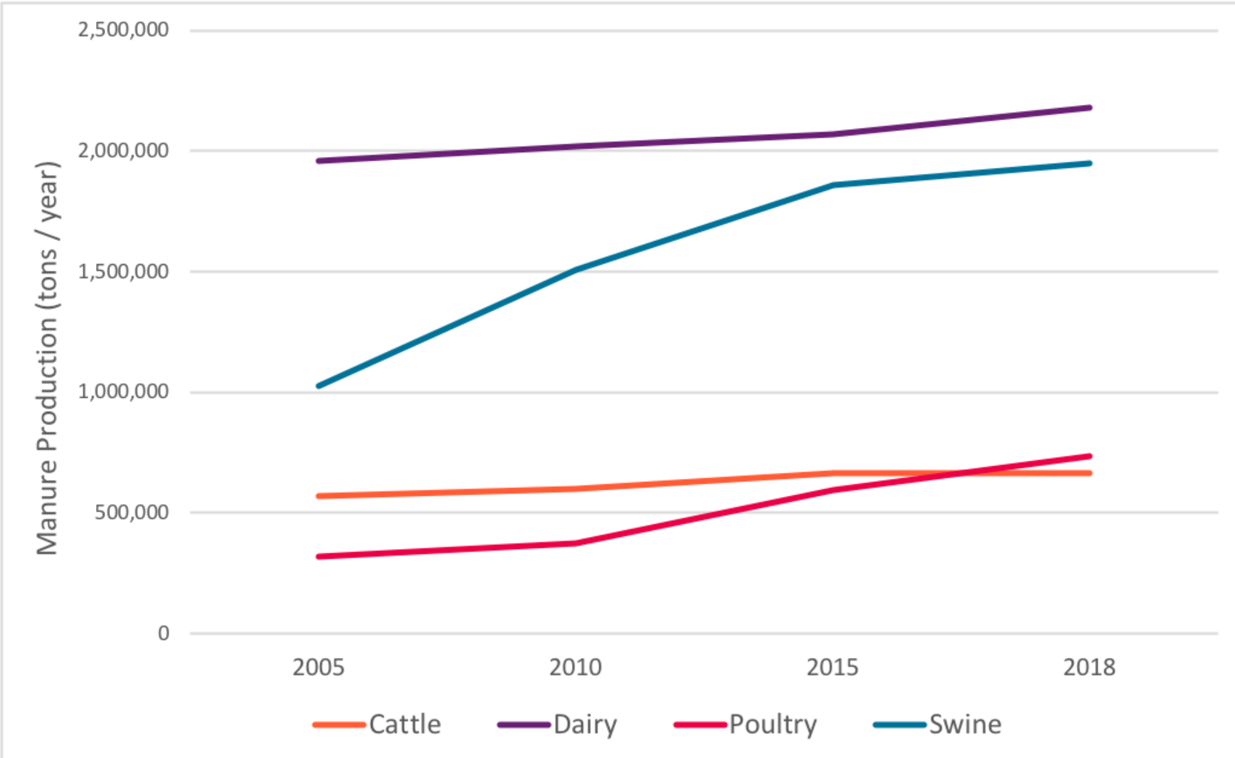
Trends in Manure and Nutrient Production

The amount of manure produced in the Maumee River basin has increased concurrently with the growth of new facilities (Figure 4). Dairy is consistently the largest producer of manure, followed by swine, cattle and poultry. Recent growth in poultry facilities has caused manure from chickens to now equal that of beef cattle in the Maumee basin.

As poultry manure contains more phosphorous than other animal manures, chickens now rival and even exceed swine in phosphorus production in the Maumee basin (Figure 5). Based on numbers from the MWPS-18, poultry manure from egg-laying hens is estimated to have two to three times the amount of phosphorus per pound of manure than beef cattle or hogs, and nearly six times the amount of phosphorus than dairy cattle.

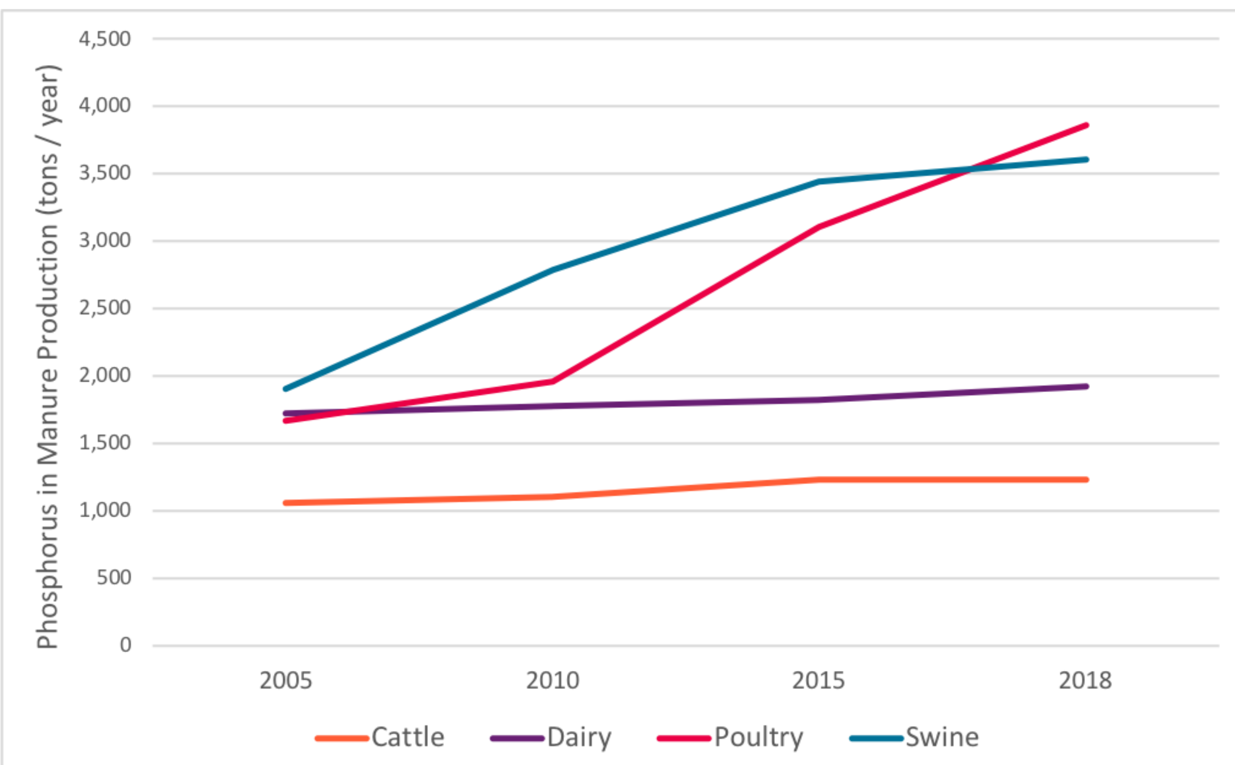
Manure production in the Maumee has increased by 43 percent over the period of study, from 3.9 million tons per year in 2005 to 5.5 million tons per year in 2018. Phosphorus production has increased 67 percent, from 6,348 tons per year in 2005 to 10,610 tons per year in 2018 (Table 6).

Figure 4. Manure Production in the Maumee River Basin, 2005-2018.



Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management, Michigan Dept. of Environmental Quality and Midwest Planning Service

Figure 5. Phosphorus Production in the Maumee River Basin, 2005-2018.



Source: EWG vand ELPC ia Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management, Michigan Dept. of Environmental Quality and Midwest Planning Service

Table 6. Manure and Phosphorus Production in the Maumee River Basin by Animal Type, 2005-2018.

Animal Type	2005	2010	2015	2018
Manure Production (tons/year)				
Cattle	570,813	599,815	665,581	665,581
Dairy	1,960,225	2,021,480	2,069,243	2,178,349
Poultry	318,949	373,174	593,234	736,574
Swine	1,027,116	1,507,520	1,860,483	1,950,584
Total	3,877,103	4,501,989	5,188,541	5,531,088
Phosphorus Production (tons/year)				
Cattle	1,052	1,106	1,227	1,227
Dairy	1,725	1,779	1,821	1,917
Poultry	1,671	1,955	3,107	3,858
Swine	1,900	2,788	3,441	3,608
Total	6,348	7,628	9,597	10,610

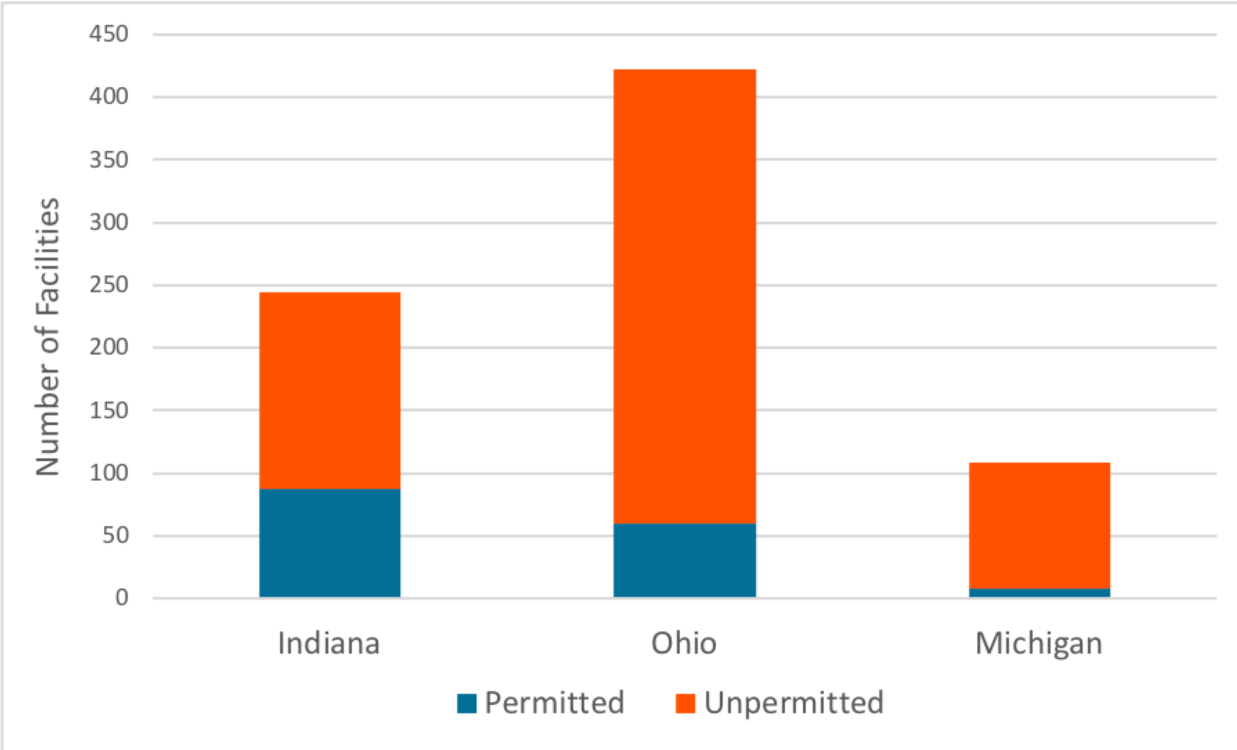
Source: EWG and ELPC via Midwest Planning Service

Permitted Facilities

Each of the three states in the Maumee River basin (Indiana, Ohio and Michigan) has its own regulations about whether an AFO requires a permit. This depends largely on the number of animals housed at each facility. We examined permitted facilities by state to determine the number and type of operations permitted in the Maumee basin as of 2018. Figures 6 and 7 illustrate the percentage of facilities permitted by state and animal type.

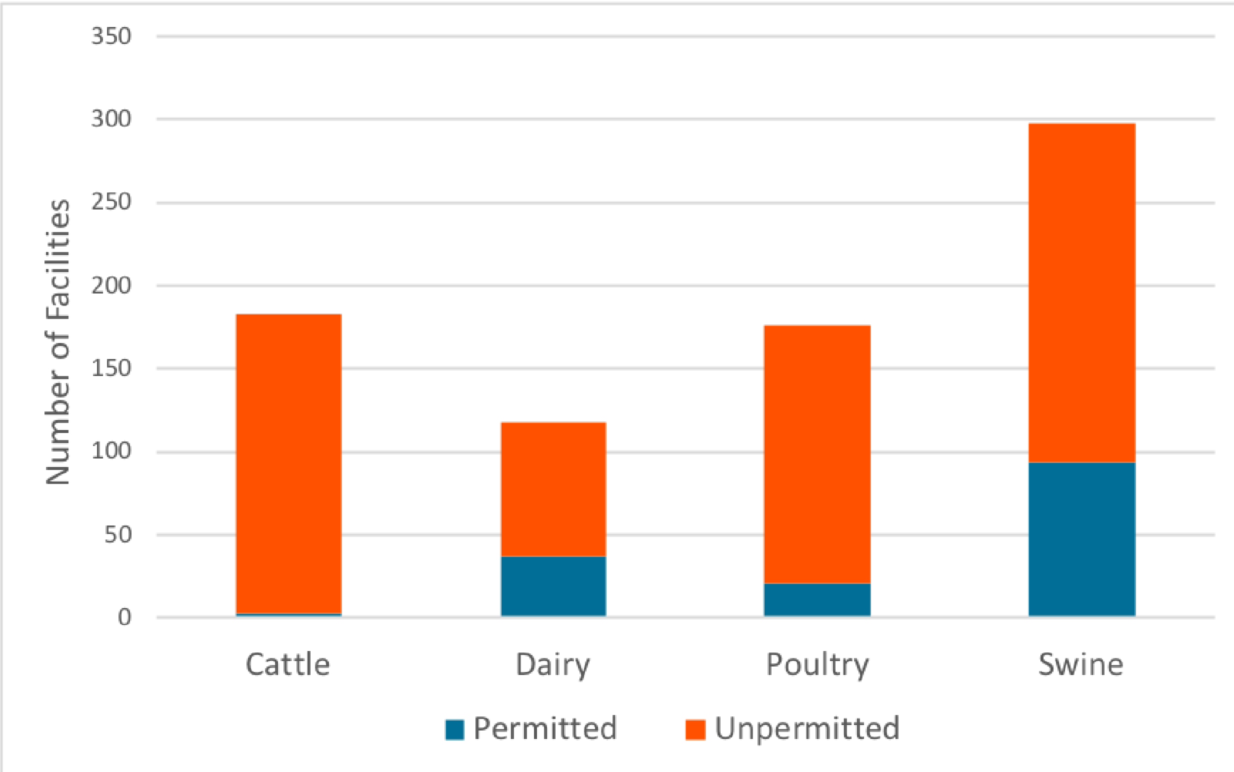
Overall, 155 of the 775 AFOs, or 20 percent, were permitted. The highest percentage of facilities permitted was in Indiana (36 percent), which has the most stringent permitting regulations of the three states. In Ohio, 14 percent of facilities were permitted; in Michigan, only 7 percent. Swine and dairy were the most commonly permitted, with 32 percent and 31 percent of facilities permitted, respectively. Only 12 percent of poultry and 2 percent of cattle facilities had permits.

Figure 6. Percentage of Animal Feeding Operations in the Maumee Permitted by State



Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management and Michigan Dept. of Environmental Quality

Figure 7. Percentage of Animal Feeding Operations in the Maumee Permitted by Animal Type



Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management and Michigan Dept. of Environmental Quality

Permitted Manure in Ohio

Ohio accounts for 74 percent of animals and 68 percent of manure production in the Maumee Basin. To estimate the percentage of manure captured through permitted data in Ohio, manure production from permitted facilities was compared to manure production from all facilities. Results are shown in Table 7.

More than half (56 percent) of the total estimated manure production is not captured by permitted facilities in the Ohio portion of the Maumee. An estimated 79 percent of hog manure, 51 percent of chicken manure, 34 percent of dairy manure and 84 percent of cattle manure is unaccounted for. With only 9 percent of poultry facilities permitted in Ohio but nearly half of the poultry manure accounted for, this would suggest that the few permitted poultry facilities account for the majority of the manure produced.

We also saw this with dairy facilities, in which 66 percent of manure is captured by the 27 percent of facilities permitted. In contrast, permitted facilities for swine and beef cattle make up a much smaller proportion of the manure produced by these animals in the Ohio portion of the Maumee.

Table 7. Permitted Manure in Ohio (2015)

	Number of facilities permitted	Number of facilities mapped	Manure production from all facilities (tons/year)	Manure production from all facilities (tons/year)
Cattle	1	36	28,935	178,709
Dairy	20	74	1,034,249	1,555,448
Poultry	8	91	268,915	548,937
Swine	31	221	302,765	1,472,384
TOTAL	60	422	1,634,865	3,755,479

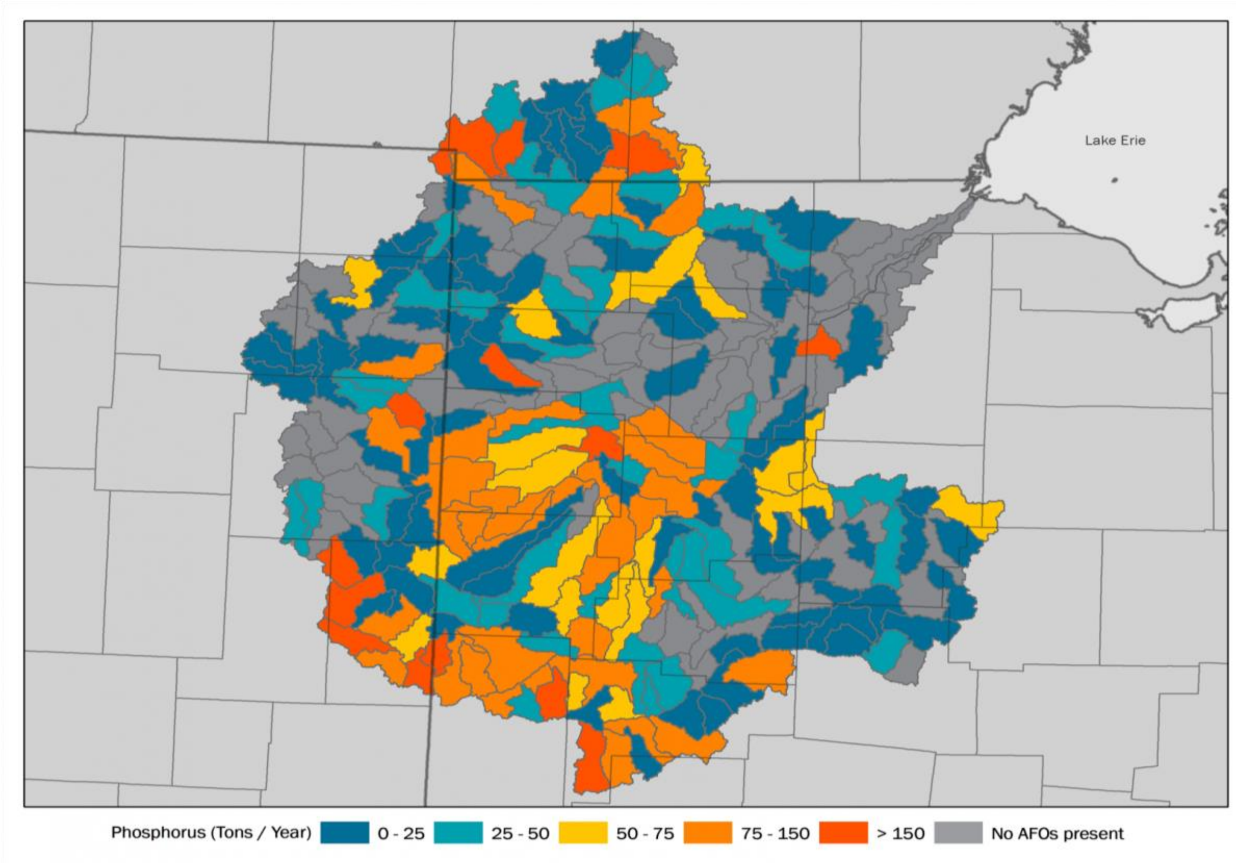
Source: EWG and ELPC via Ohio Dept. of Agriculture

Watershed Analysis

Phosphorus production from animal manure in the Maumee was summed within each HUC12 watershed (Figure 8). There are 252 watersheds in the Maumee Basin, 70 of which do not contain any AFOs. Half of the total phosphorus production from animal operations in the Maumee can be accounted for in just 30 HUC12 watersheds.

Platter Creek produces the most phosphorus of any HUC12 in the Maumee. It contains just four AFOs but accounts for 9 percent of total P production from animal manure in the Maumee. Platter Creek is home to both the largest poultry and largest beef cattle operation in the Maumee. The poultry operation houses more than four million egg-laying hens (more than three times the number of the next largest facility), and the cattle operation houses more than 3000 cattle. Both operations have permits.

Figure 8. Phosphorus Production from Animal Manure by HUC12 Watershed in the Maumee Basin.



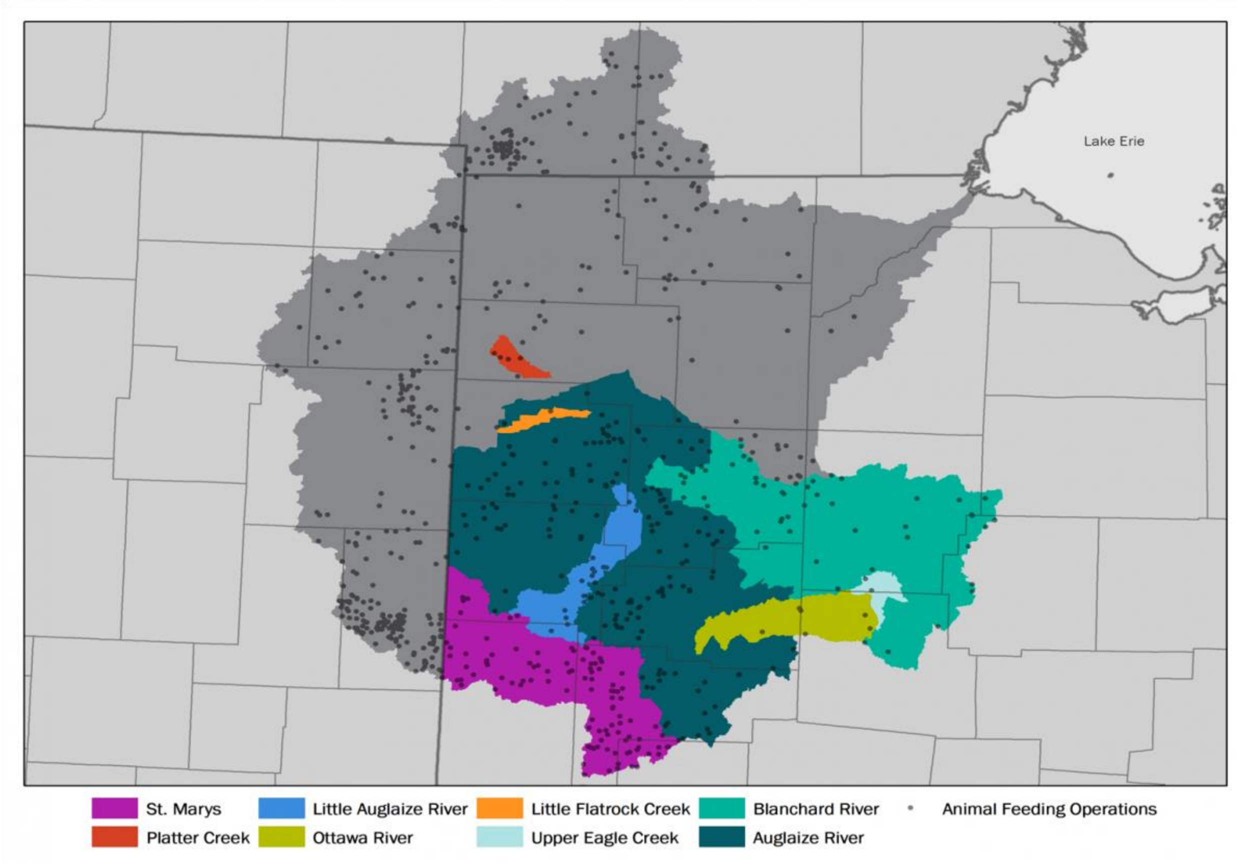
Source: EWG and ELPC via Ohio Dept. of Agriculture, Indiana Dept. of Environmental Management, Michigan Dept. of Environmental Quality and Midwest Planning Service

Distressed Watersheds

Eight watersheds in Ohio were proposed to receive a distressed designation in 2018 by former Gov. John Kasich (Figure 9).

Of the HUC12 watersheds estimated to produce between 75 and 150 tons of phosphorus per year, 24 of the 33 (73 percent) fall within a proposed distressed watershed. Of the HUC12 watersheds estimated to produce more than 150 tons of phosphorus per year, five of the 14 (36 percent) are located within a distressed watershed. More than half (56 percent) of the total phosphorus from animal manure in the Maumee is produced from the 345 animal operations in the eight distressed watersheds. We estimate that Platter Creek generates the most phosphorus from animal manure of any HUC12 watershed in the Maumee Basin. It is also the only stand-alone distressed watershed proposed in 2018.

Figure 9. Ohio Proposed Distressed Watersheds



Source: EWG and ELPC via Ohio Dept. of Agriculture

Source of Phosphorus

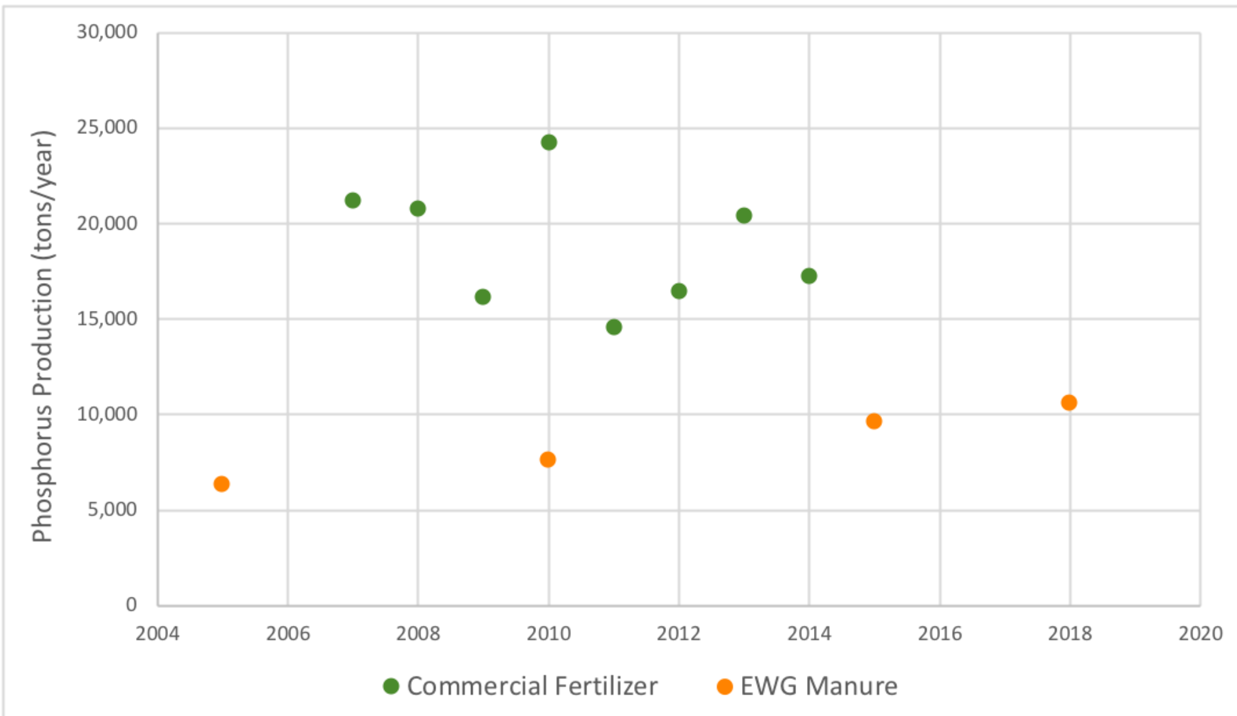
Data on commercial fertilizer in the Maumee basin was obtained from the Nutrient Use Geographic Information System, or NUGIS, of the International Plant Nutrition Institute, or IPNI. IPNI has compiled a nationwide database of county-level fertilizer sales provided by the Association of American Plant Food Control Officials, or AAPFCO. These data are separated into farm and non-farm uses, and additional quality control steps are taken by IPNI to account for errors and reduce spatial bias.

Nutrient data is then aggregated from the county to the watershed scale. Yearly watershed level data on tons of P_20_5excreted from livestock manure were pulled directly from the IPNI database for the Maumee River Basin to estimate trends in commercial fertilizer input over time. Farm fertilizer P_20_5 was multiplied by .44 to convert to elemental P.

IPNI data were provided at five-year intervals corresponding to the USDA Agricultural Census between 1987 and 2007 (1987, 1992, 1997, 2002 and 2007) and yearly between 2008 to 2014. To compare phosphorus production from commercial fertilizer to animal manure estimates from this study, IPNI data were pulled for the years 2007 through 2014. Results suggest that commercial fertilizer rates are gradually declining, which has been documented by numerous other studies (Figure 10; IJC, 2018; Kast, 2018).

Once they are published, it will be valuable to examine more recent IPNI data to estimate the rate of this downward trend in commercial fertilizer use. Over the same time period, phosphorus production rates from animal manure in the Maumee increased by 67 percent. When summing the two nutrient sources, we do not see an overall increase in phosphorus production in the Maumee but rather a shift in the relative contribution of the major agricultural sources.

Figure 10. Phosphorus Production by Agricultural Source in the Maumee Basin.



Source: EWG and ELPC via International Plant Nutrition Institute

REFERENCES

D'Anglada, L., Gould, C., Thur, S., Lape, J., Backer, L., Bricker, S., Clyde, T., Davis, T., Dortch, Q., Duriancik, L., Emery, E., Evans, M.A., Fogarty, L., Friona, T., Garrison, D., Graham, J., Handy, S., Johnson, M.-V., Lee, D., Lewitus, A., Litaker, W., Loeffler, C., Lorenzoni, L., Malloy, E.H., Makuch, J., Martinez, E., Meckley, T., Melnick, R., Myers, D., Ramsdell, J., Rohring, E., Rothlisberger, J., Ruberg, S., Ziegler, T. (2018). "Harmful Algal Blooms and Hypoxia in the United States: A Report on Interagency Progress and Implementation." National Oceanic and Atmospheric Administration.

Environmental Protection Agency, 2010. Ohio Lake Erie Phosphorus Task Force Final Report. Available at: **https://epa.ohio.gov/portals/35/lakeerie/ptaskforce/Task_Force_Final_Report_April_2010.pdf**

Goolsby, D.A., Battaglin, W.A., Lawrence, G.B., Artz R.S., Aulenbach B.T., Hooper R.P., Keeney D.R. and Stensland G.J. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin; Topic 3. Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No. 17. NOAA, Silver Spring, MD

International Joint Commission. Fertilizer Application Patterns and Trends and Their Implications for Water Quality in the Western Lake Erie Basin, 2018.

Kast, J. Manure Management in the Maumee River Watershed and Watershed Modeling to Assess Impacts on Lake Erie's Water Quality. Ohio State University, 2018.

Long, C.M., Muenich, R.L., Kalcic, M.M., and Scavia, D. 2017. Impacts of Concentrated Animal Feeding Operations and manure on nutrient inputs in a watershed. IAGLR 2017 Annual Meeting, Detroit, Michigan, Abstracts, p. 244.

Maccoux MJ, Dove A, Backus SM, and Dolan DM. 2016. Total and soluble reactive phosphorus loadings to Lake Erie: a detailed accounting by year, basin, country, and tributary. J Great Lakes Res 42: 1151–65.

MWPS-11, Agricultural Waste Management Field Handbook. Amend. IA-3, July 2013. MidWest Plan Service (MWPS): Ames, Iowa.

MWPS-18, Livestock Waste Facilities Handbook, 1993. MidWest Plan Service (MWPS): Ames, Iowa.

National Pork Board, Pig Space: Finding the Right Fit. Farm Journal's Pork, January 18, 2011.

Penn State Extension. Recommendations for Calf and Heifer Housing Dimensions for Holsteins. May 2016.

Planet Team (2017). Planet Application Program Interface: In Space for Life on Earth. San Francisco, CA. https://api.planet.com.

Poultry Industry Manual, FAD Prep Foreign Animal Disease Preparedness & Response Plan. March 2013. United States Department of Agriculture, National Animal Health Emergency Management System.

Scavia, D.; Kalcic, M.; Muenich, R.L.; Aloysius, N.; Boles, C.; Confessor, R.; DePinto, J.; Gildow, M.; Martin, J.; Read, J. Informing Lake Erie Agriculture Nutrient Management via Scenario Evaluation; University of Michigan: Ann Arbor, MI, USA, 2016.

United Egg Producers, Compete Guidelines for Cage and Cage-Free Housing. Animal Husbandry Guidelines for U.S. Egg-Laying Flocks, 2017.

USDA NASS, 2012 Census of Agriculture, Available at: **https://www.nass.usda.gov/Publications/AgCensus/2012/**

Biogeochemistry
https://doi.org/10.1007/s10533-020-00691-6

REVIEW ARTICLE

From hogs to HABs: impacts of industrial farming in the US on nitrogen and phosphorus and greenhouse gas pollution

Patricia M. Glibert

Received: 13 February 2020 / Accepted: 22 July 2020

Abstract Nutrient pollution and greenhouse gas emissions related to crop agriculture and confined animal feeding operations (CAFOs) in the US have changed substantially in recent years, in amounts and forms. This review is intended to provide a broad view of how nutrient inputs—from fertilizer and CAFOs—as well as atmospheric NH_3 and greenhouse gas emissions, are changing regionally within the US and how these changes compare with nutrient inputs from human wastewater. Use of commercial nitrogen (N) fertilizer in the US, which now exceeds 12,000,000 metric tonnes (MT) continues to increase, at a rate of 60,000 MT per year, while that of phosphorus (P) has remained nearly constant over the past decade at around 1,800,000 MT. The number of CAFOs in the US has increased nearly 10% since 2012, driven largely by a near 13% increase in hog production. The annualized inventory of cattle, dairy cows, hogs, broiler chickens and turkeys is approximately 8.7 billion, but CAFOs are highly regionally concentrated by animal sector. Country-wide, N applied by fertilizer is about threefold greater than manure N inputs, but for P these inputs are more comparable. Total manure inputs now exceed 4,000,000 MT as N and 1,400,000 MT as P. For both N and P, inputs and proportions vary widely by US region. The waste from hog and dairy operations is mainly held in open lagoons that contribute to NH_3 and greenhouse gas (as CH_4 and N_2O) emissions. Emissions of NH_3 from animal waste in 2019 were estimated at $>$ 4,500,000 MT. Emissions of CH_4 from manure management increased 66% from 1990 to 2017 (that from dairy increased 134%, cattle 9.6%, hogs 29% and poultry 3%), while those of N_2O increased 34% over the same time period (dairy 15%, cattle 46%, hogs 58%, and poultry 14%). Waste from CAFOs contribute substantially to nutrient pollution when spread on fields, often at higher N and P application rates than those of commercial fertilizer. Managing the runoff associated with fertilizer use has improved with best management practices, but reducing the growing waste from CAFO operations is essential if eutrophication and its effects on fresh and marine waters–namely hypoxia and harmful algal blooms (HABs)—are to be reduced.

Keywords N and P fertilizer · Manure · Factory farms · CAFOs · Eutrophication · Algal blooms · Atmospheric deposition · Greenhouse gasses

Responsible Editor: Jacques C. Finlay.

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10533-020-00691-6) contains supplementary material, which is available to authorized users.

P. M. Glibert (✉)
Horn Point Laboratory, University of Maryland Center for Environmental Science, PO Box 775, Cambridge, MD 21613, USA
e-mail: glibert@umces.edu

Introduction

In the 1970s, eutrophication from nitrogen (N) and phosphorus (P) pollution was a problem largely localized to some freshwaters (e.g., Likens 1972, Ketchum 1972), and the major source of nutrient pollution was considered to be sewage wastewater. At that time the US population was about 200 million, but by 2019, population had increased to 328 million (https://www.multpl.com/united-states-population/table/by-year). Eutrophication is the cause of hypoxia zones that have now been documented in most US estuaries and along many coasts (e.g., Cloern 2001; Howarth et al. 2002, Bricker et al. 2007 and references therein) and such zones are increasing worldwide (Diaz and Rosenberg 2008, Kemp et al. 2009; Rabalais et al. 2009, 2010). Freshwater eutrophication is an equally serious US and global problem (e.g., Smith et al. 2006; Du et al. 2019). The corn-belt of the US, the massive 39 million-ha span (primarily encompassing the states of Illinois, Indiana, Iowa, Missouri and Ohio) that uses more than 4.5 million metric tonnes (MT) of chemical N fertilizer and nearly a million MT of N from manure for the growth of corn and soybean (Foley 2013), is considered to be the source of the N fueling the dead zone in the Gulf of Mexico, one of the largest hypoxic zones in the US (e.g., Scavia et al. 2003; Turner et al. 2006; Alexander et al. 2008). Eutrophication is also highly correlated with the increasing frequency and geographic spread of both freshwater and coastal marine harmful algal blooms (HABs; Anderson et al. 2002; Heisler et al. 2008; Glibert et al. 2005, 2014, 2018). These events have now been documented in every state, and recent examples of algal blooms affecting drinking water (Anderson et al. 2008; Steffen et al. 2017), fisheries closures and human health issues are regularly reported throughout the country (e.g., Fleming et al. 2005; Backer et al. 2005; Backer and McGillicuddy 2006; McCabe et al. 2016 among others). Throughout the world, excess N and P have led to a cascade of atmospheric, water and human health problems and managing nutrient pollution has become a grand challenge (e.g., Galloway et al. 2003; Townsend et al. 2003; Howarth 2008; Billen et al. 2013; Sutton et al. 2013; Davidson et al. 2015; Glibert et al. 2014, 2018; Glibert and Burford 2017; Glibert 2020).

In the 1970s, greenhouse gases were only just beginning to be recognized as a threat to future global warming. Since then, global greenhouse gas emissions have increased 75%, with a 25% increase from the 1990s to 2004 alone, primarily due to increases in fossil fuel use globally, but particularly from the rapid industrial development in China and other developing countries (https://www.pbl.nl/en/dossiers/Climatechange/TrendGHGemissions1990-2004). However, agriculture also contributes to this increase, such that by 2017, agricultural sources contributed 10–15% of greenhouse gas emissions in the US (https://www.epa.gov/ghgemissions/sources-greenhouse-gas-emissions; Grossi et al. 2019). Agriculture contributes to such emissions in multiple ways, including direct emissions from livestock (enteric fermentation), and as will be shown below, from handling of animal waste and from fertilizer applications.

Although agriculture-related eutrophication problems have escalated in the past few decades, farming practices actually began to change rapidly after World War II. The so-called Green Revolution, the period during which the manufacture and application of N-based fertilizers expanded at a rapid pace also included other advances in farming technology, such as improved irrigation, mechanized equipment and better seeds (e.g., Smil 2001; Erisman et al. 2008; Pingali 2012). As described by Imhoff (2019, p. 33), "Chemicals were concocted into a slew of pesticides, herbicides and synthetic fertilizers… Plant breeding also evolved, creating high-yielding hybrid grains tailored to meet these shifts in chemical inputs and mechanical growing and harvesting". Thus, compared to pre-industrial times, the US has seen a > fivefold increase in N use on average, but this increase has been up to > 35-fold in some regions of the country (Houlton et al. 2013; Sobota et al. 2015).

Increased fertilizer use led to rising grain yields, but also an oversupply of grains. The US did not become the world's breadbasket by grand or moral intentions, but rather because, as farming became more intensive, there was a surplus and a need to find new markets for products and a desire to raise domestic profits (Walker 2019). The US consequently adopted policies that have promoted the "feeding of the world" in order to sustain profitability (e.g., Imhoff 2019). The US now produces a total weight in corn that is, "remarkably close to the estimated weight of the global population," about 287 million MT (Gunderson et al. 2018). By 2011, about a third of all US crops were exported (Hertel 2018).

Oversaturation of the market at various times has also led to further plowing of the ground for more crops to make up for lost income. The motive is to grow the most high-yielding, high-paying crop.

The US Farm Bill, the major legislation that encompasses agriculture, conservation, and research and food assistance programs, has, over its various iterations and re-authorizations, incentivized monoculture production, primarily corn and soybean. Its major objective is to stabilize prices and incomes, not to protect environmental interests (Ruhl 2000). This massively expensive legislation guides all aspects of the US food and farming systems, but is heavily influenced by special interests, and thus its policies have favored consolidated large-scale farms, and grains over fruits and vegetables, heavy use of chemical fertilizers, among other incentives to maximize profits over environmental stewardship (e.g., Miller 2017; Imhoff 2019).

Because of these shifts and other policy- or economic-related factors, most of the grain grown in US is not used directly for food. It is fed to animals in feedlots (about 36%), used for biofuels (about 40%), exported (about 10%), and used in high-fructose corn syrup and other food products (a few %; Foley 2013; Barton and Clark 2014). Of the total acreage in corn, about 5%, or 2 million ha, is needed just to support the supply of chicken and pork sold at McDonald's and Walmart (von Reusner 2019). Only $\sim$ 1% of all corn grown is directly eaten by people as "sweet corn" (Bittman 2019). The mandate for ethanol production in the US, originally intended to support farmers and reduce foreign dependence on oil, has resulted in 12.5 million ha of corn dedicated to ethanol corn (equivalent to more than all the crop land in Iowa; Imhoff 2019) and likely has contributed to an increase in N fertilizer use in the past 2 decades (e.g., Sabo et al. 2019). In the 1990s, the US produced about 10 million MT of corn for biofuels; in 2018 it was $\sim$ 140 million MT, about 12-fold more than that used for high fructose corn syrup (https://www.worldofcorn.com/#us-corn-at-a-glance). Recent trade tariffs notwithstanding, this demand will continue.

The factory-efficient approach to farming has gone hand-in-hand with changing diets (e.g., Godfrey et al. 2018). People consume more protein—as meat—when wealth increases and as the cost of meat production decreases. Cattle, otherwise adapted to grass, are fed corn because it is a cheap commodity, because "the great pile must be consumed", and because animals can grow to market size much more quickly (Pollan 2006, p. 68). Notable, however, is the fact that the nutritional content of corn-fed beef differs from that of grass-fed beef, with more saturated fat and less omega-3-fatty acids (Pollan 2006). Similarly, corn-fed chickens grow much faster and larger than free-range chickens. Broiler chickens are now about 12% larger than those grown just a decade ago (Pelton et al. 2020).

Concentrated animal feeding operations (CAFOs) began increasing rapidly in the 1990s (e.g., Mallin 2000) as the most economically efficient way to produce the quantity of meat needed. The number of animals per farm and the scale and size of farms increased, while the number of farms decreased; small animal farms were simply no longer economically viable (Fig. 1a). Accordingly, "In one generation, the number of farms producing hogs fell by almost three quarters—while the median number of hogs per farm climbed from 1200 to 40,000" (Walker 2019, p. 35). Furthermore, agribusinesses have concentrated all aspects of animal production by buying companies in the same line of production *and* buying companies that had previously provided them with raw materials or sold finished products, such as meat packing plants. As noted by Walker (2019 p. 134, quoting journalist Barry Lynn), "If antitrust law exists to serve the consumer, and if consumers are best served by getting more for less, and if the best way to get more for less is to encourage business to be 'efficient', and if the best way to be efficient is to build up scale and scope, then ergo, monopoly is the best friend of the consumer".

The proliferation of CAFOs is also a function of the aforementioned growth in corn and soybean production, as the over-production of these commodities depressed the price of livestock feed, which, in turn, created an indirect subsidy for animal production systems (Pollan 2006; Food and Water Watch 2015). Cheap animal feed translates into cheaper meat products. Packing large numbers of animals in confined spaces was also facilitated by the massive use of antibiotics (Walker 2019). In all, US farms, owned increasingly by a comparatively small number of companies, have become "too big to fail" (Walker 2019). Mega-farms owners can also buffer economic downturns far better than family farms.

The dietary change to increased consumption of meat is not just a US phenomenon; Chinese

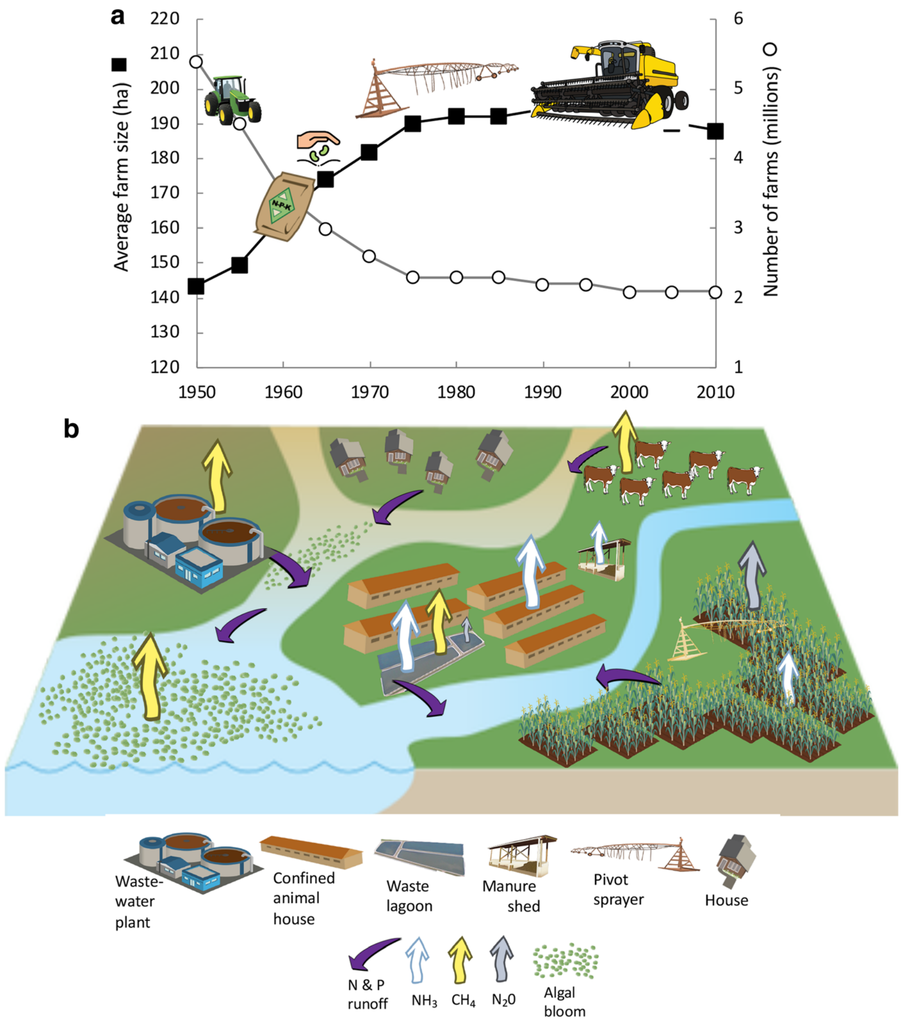


Fig. 1 **a** Change in the average US farm size and number of farms with time. **b** Conceptual schematic of the sources of nitrogen and phosphorus runoff and ammonia and greenhouse gas emissions and effects on algal blooms considered herein. Symbols and icons are from the University of Maryland Center for Environmental Science (UMCES) Integration and Application Network (IAN) image library or from Vectorstock used under an expanded license

consumption of pork, poultry and beef has also increased and meat has become a more consistent dietary component as its economy has grown. China's meat production, in fact, rose 250% from 1986 to 2012, with another 30% rise by 2020, and their need for animal feed is one of the major drivers of their escalation in importation of US and Brazilian soybeans over the past decade (Sheldon 2019). In China, farms with > 1000 head of cattle grew 16% from 2011 to 2014, while those of dairy cows grew 40%. A single Chinese dairy farm with > 100,000 head is currently being developed (DuBois and Gao 2017).

The numbers of animals in CAFOs differs widely, depending on the animal and regional permitting. CAFOs are categorized as small, medium, or large depending on the number and type of animal and the drainage system for their waste (Table 1). Small CAFOs (those with small animal populations just under the definition of medium-sized) are often undercounted or un-permitted and are expanding in many regions where regulations apply only to larger facilities. By keeping animal operations to numbers that do not fall into the category for regulation, operators maintain more options—and more polluting options—for handling waste. Current permitting and

Table 1 Definitions of large and medium CAFOs according to USEPA (https://www3.epa.gov/npdes/pubs/sector_table.pdf)

Animal type	Large	Medium*
Cattle	> 1000	300–999
Dairy	> 700	200–699
Swine (> 55 lbs)	> 2500	750–2499
Swine (< 55 lbs)	> 10,000	3000–9909
Broilers	> 125,000	37,500–124,999
Layers	> 82,000	25,000–81,999

Note that there are many animals in confined conditions in operations with numbers fewer than indicated here and thus are undercounted in this analysis. Small CAFOS have numbers of animals less than those defined for "medium"

*Medium either has animals in range above or has a manmade ditch or pipe that carries manure or wastewater to surface water or the animals come into contact with surface water that passes through the area where they are confined

legal differences between states makes it difficult to obtain an accurate count of the number of CAFOs in the US. Transparency of CAFO data, with respect to permit state, location, manure storage or type, and number of animals is low for almost every state; the US Environmental Protection Agency (US EPA) does not have such data for about half of the CAFOs in its inventory of 2012 (Miller and Muren 2019). New algorithms are being applied to obtain better estimates and these approaches suggest that the number of CAFOs is actually more than 15% higher that which is routinely reported from manual enumerations (Handan-Nader and Ho 2019). Thus, the numbers reported herein are likely similarly underestimated.

Given the density of animals in CAFOs, and the rate at which animals are fed to get them to market as quickly as possible, the amount of animal waste from these operations can be very large (e.g., Cahoon et al. 1999; Mallin 2000, Mallin and Cahoon 2003, Burkholder et al. 2007, Mallin et al. 2015). Although the waste produced by CAFOs across the US is examined in this review, as an example of the scale of this nutrient source, in the Cape Fear River basin of North Carolina, it was estimated that in the early 2000s, there were 5 million hogs, 300 million chickens, and 16 million turkeys produced annually on ~ 2000 CAFOs, yielding 82,700 MT of N and 26,000 MT of P (Mallin et al. 2015 and references therein). Moreover, in the Chesapeake Bay region, where poultry production has increased 6% in the past decade, the manure production from these CAFOs has actually increased 16% because larger, more meaty chickens are being grown (Pelton et al. 2020).

Collectively, farming practices today contribute substantially to N and P pollution of waterways and to NH_3 and greenhouse gas emissions (Fig. 1b). Most CAFOs produce waste at a scale that is more than can be accommodated by the method by which manure was traditionally handled, that is, by spreading it on adjacent land as fertilizer (as dry litter for poultry and as liquid manure for hog and dairy manure; Mallin et al. 2015). There is no wastewater treatment for these animal wastes—other than holding it for periods of time. While much is spread on land, most of the waste from dairy or hog operations is held in large, open-pit lagoons. The breeching of these lagoons during flooding and hurricanes has been a major pollution problem for states such as North Carolina with their large hog population. Many of North Carolina's CAFOs are built on flood plains (www.ecowatch.com/factory-farm-waste-north-carolina-2628852719.htm) where land is comparatively inexpensive (but note that a moratorium has been in place since 1997 disallowing any new lagoons to be constructed in North Carolina). Following Hurricanes Florence in 2018, 33 such lagoons overflowed, spilling over 30 trillion L of wastes, together with thousands of dead hogs, repeating events of years earlier when Hurricane Floyd in 1999 led to spillage of 9 trillion L of hog waste (Buford 2018). In addition to the waste that makes its way into waterways, the volatilization of animal wastes and manures contributes to atmospheric deposition of NH_3/NH_4^+, which has been shown to account for approximately half of the atmospheric N deposition in Mid-Atlantic estuaries such as the Neuse River Estuary and Atlantic coastal waters (Paerl, 1997; Whitall et al., 2003). Each broiler chicken, for example, emits between 0.27 and 0.54 g NH_3 from its manure per day (Russ and Schaeffer 2018). Furthermore, and as will be described herein, liquid manure systems also contribute directly to greenhouse emissions, as CH_4 and N_2O.

The goal of this paper is to highlight inputs of nutrients and greenhouse gas pollution from farms in the US, by source, form, and by region of the country and their rapid changes over the recent years. There have been a number of recent inventories of fertilizer, manure and/or greenhouse gases in the US, built on modeling of a comprehensive suite of sources and fates (e.g., Ruddy et al. 2006; Sobota et al. 2015; Houlton et al. 2013; Swaney et al. 2018a, b; Bouwman et al. 2017; Sabo et al. 2019). Those efforts have focused on defining patterns and trends at fine spatial scales, i.e. at the level of counties or hydrologic units, and quantifying surpluses, not just sources. In contrast, this review is intended to provide the "30,000 ft" view of how nutrient inputs, from fertilizer and CAFOs, as well as atmospheric NH_3 and greenhouse gas emissions, are changing regionally within the US and how these changes compare with nutrient inputs from human wastewater. By highlighting the rapid pace of changes in these important sources of environmental nutrient loads and other pollutants, these data may help to guide broad priorities for management actions for reduction of both water and air pollutants from these industrial operations; regional managers setting local nutrient reduction targets or strategies will want to consult the more detailed nutrient inventories. Although this paper specifically focuses on the US, there are important lessons that are applicable globally.

Methods

Overview

This paper begins with a review of the trends in total farms and their size. The change in use and form of chemical fertilizers (both N and P) in the US over time is then summarized as totals and for the major crops of corn, soybean, wheat, and cotton. The growth in major animal operations (including beef cattle, dairy, hogs, chickens as "broilers", and turkeys) is then considered, as is the total numbers of CAFOs and their change regionally, and the total N and P released by animal type regionally. Emissions of NH_3 and greenhouse gasses are then summarized. The N and P in human wastewater was estimated by state, along with overarching status of wastewater infrastructure by state. Data for these different sources of N and P were compared by aggregated US regions. Every effort was made to capture data from similar time periods for the different parameters; dates encompassed by the different trends are noted throughout.

Data sources and calculations

Publicly-available and/or published data were accessed for all aspects of this analysis, and data sources are identified for each set of data used. Where assumptions or calculations were applied to available data, they are explicitly stated. Rates of change were calculated across various time periods depending on parameter and data availability.

The number and sizes of farms was obtained from https://cropinsuranceinamerica.org/in-the-states/ based on the year 2012. Data for 2017 were obtained from US Farm Data (www.usfarmdata.com/percentage-of-small-medium-and-large-farms-in-the-us).

Annual fertilizer statistics were obtained from the US EPA (https://www.epa.gov/nutrient-policy-data/commercial-fertilizer-purchased). These data are reported by crop and nutrient form. Data reported as P_2O_5 were herein converted to P using the factor 0.436. The US Department of Agriculture (USDA) have made available the total amount of N and P used by state in recent years (https://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx). Fertilizer data are based on available data through 2014; individual years are identified in comparative analyses. Other fertilizer data were obtained from the analyses of Sabo et al. (2019) for N and from comparable US EPA analyses for P (https://doi.org/10.23719/1504278). These latter data, which are reported for 2002, 2007, and 2012, catalogued inputs and fates at the level of hydrologic units, roughly equivalent to medium-river-sized basins (HUC-8). These data were herein sorted and summed by state and then aggregated by US region.

Water use data by crop were from USDA (2008 as reported in Barton and Clark 2014).

Animal inventories were obtained from USDA (for 2012 from www.usda.gov/Publications/AgCensus/2012/Full-Report/Volume_1_Chapter_2_US_State_Level/; for 2016 and 2017 from www.aphis.usda/gov/animal-health/nahms/downloads/Demographics2017.pdf; and for 2019 from www.nass.usda.gov/Statistics_by_State/index.php). Animal inventory comparisons are herein focused on cattle, dairy cows, hogs, broiler chickens and turkeys, and while other animals may be inventoried and reported, these represent the major animals in polluting CAFO operations.

To normalize animal numbers to biomass, equivalent animal units were calculated (equal to a 1000 lb or 453 kg animal). Conversion factors are reported in Online Resources Table S1.

The most recent inventory of CAFOs, as of 2018, as well as the percent of which are permitted, were obtained from the US EPA (https://www.epa.gov/sites/production/files/2019-09/documents/cafo_tracksum_endyear_2018.pdf). As noted by the US EPA in reporting these statistics, these numbers include all CAFOs with numbers of animals above the size thresholds set out for large CAFOs. National maps of CAFOs were obtained from Food and Water Watch (2015, 2020). Changes in CAFOs from 2011 to 2017 were also obtained from Walljasper (2018, https://investigatemidwest.org/2018/06/07/large-animal-operations-on-the-rise/).

Manure inventories were obtained from multiple sources. Data from 1982 to 2001 were obtained from Ruddy et al. (2006; the US Geological Survey, https://water.usgs.gov/pubs/sir/2006/5012/excel/Nutrient_Inputs_1982-2001jan06.xls). The US EPA has reported manure N and P by state for the year 2007 (www.epa.gov/nutrient-policy-data/estimated animal-agriculture-nitrogen-and-phosphorus-manure). Sabo et al. (2019) provided manure N estimates for the years 2002, 2007, and 2012 for N by hydrologic unit, and a similar analysis for P was obtained from the US EPA (https://doi.org/10.23719/1504278). These latter data were not exclusive to cattle, dairy, broilers and turkeys, but were used to convey trends. These data were herein aggregated by state and then by US region. The most recent animal inventories (2019) were used to calculate the current manure inventory. It is recognized that estimates of animal N and P manure content vary widely, and thus 2 different estimates were applied herein. Estimates of N and P content in manure of each animal type as reported by Ruddy et al. (2006; Online Resources Table S2) are applied to be consistent with older estimates, and more recent manure production factors reported by Bouwman et al. (2017; On line Resources Table S2), are also reported.

Emissions of NH_3 from fertilizer use and from livestock were obtained from the US EPA National Emissions Inventory (NEI) data (https://www.epa.gov/air-emissions-inventories/2017-national-emissions-inventory-nei-data). The US EPA and the US Agriculture and Forestry Service have reported summaries of greenhouse gas emission trends due to agriculture (www.epa.gov/sites/production/files/2019-04/documents/us-ghg-inventory-2019-main-text.pdf; USDA 2016). Detailed methodology as well as sources of error in analysis are described in the source data reports. Estimates of NH_3 emissions by animal sector vary widely and represent the composite emissions from animal houses, manure management and land application, and depend on diet, temperature, other environmental conditions and local management practices. To estimate the contribution by animal sector for the most recent animal inventories (2019), emission factors of Bowen and Valiela (2001; Online Resource Table S2) were applied for cattle, dairy, hogs and broilers. It has been suggested (Pelton et al. 2020) that due to the increase in the size of chickens being grown over the past decade, emissions factors for broiler chickens are probably closer to double these earlier estimates. For turkeys, the emission factor reported by the Committee on the Environment and Natural Resources (2000; https://www.esrl.noaa.gov/csl/aqrsd/reports/ammonia.pdf) was applied. Note that the latter source also reports emission factors for other animal sectors, but to be conservative, the former values were applied herein.

Human population was obtained from www.worldpopulationreview.com/states/. Wastewater infrastructure needs by state were obtained from www.infrastructurereportcard.org. Human wastewater N and P were obtained from Sabo et al. (2019) and US EPA (https://doi.org/10.23719/1504278), respectively, based on the years 2002, 2007, and 2012.

Comparisons across regions of the US are based on 10 regions of the US as defined by the Office of Management and Budget (OMB; https://www.gao.gov/assets/120/119653.pdf; Online Resource Fig. S1).

Results

Farm inventories

As of 2012, there were just over 2 million farms in the US. Farms in the northeast and mid-Atlantic (Regions I, II and I II) are the smallest, averaging from 44 to 69 ha per farm with $< 2.7\%$ of them of a size exceeding 400 ha (Fig. 2; Online Resource Fig. S2). Farms were somewhat larger in the southeast and upper Midwest (Regions IV, V), averaging 82–104 ha per farm, with 3.4–6.1% exceeding 400 ha. In all of the other regions of the country, farm sizes averaged > 200 ha per farm with largest farms comprising

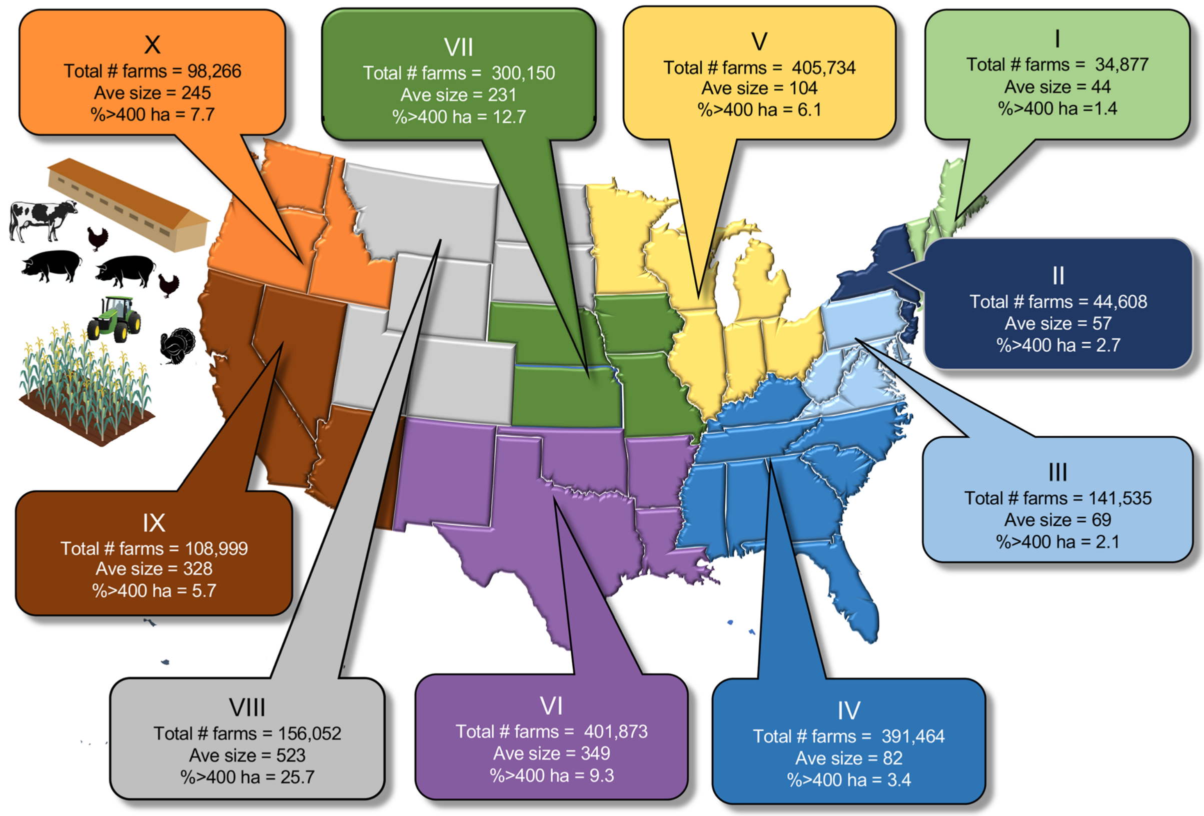


Fig. 2 Farm inventory (as total number of farms, average size (ha), and percent with > 400 ha by region of the country. Data are based on 2012 and are summarized from https://cropinsuranceinamerica.org/in-the-states/. The 10 regions of the US are as designated by the Office of Management and Budget (see also Online Resources Fig. S1). Note that Hawaii is included in Region IX and Alaska in Region X. The farm icons are from the UMCES-IAN image library

5.7–25.7% of farms. While there were still over 2 million farms in 2017, the number was down by 12,000 from the previous year, and the average farm size has increased 0.8 ha farm^{-1} year^{-1} since 2012 (www.usfarmdata.com/percentage-of-small-medium-and-large-farms-in-the-us).

Fertilizer trends with time

From 1960 to 1980, use of N-based fertilizers in the US increased linearly (r^2 = 0.98), with nearly 400,000 MT more used year^{-1} (Fig. 3a). From 1980–1990, there was a slight dip in usage, but after 1990 use of N fertilizers increased again, at a slower rate, with only ~ 60,000 MT added year^{-1} (r^2 = 0.48; Fig. 3a). The current rate of N use is ~ 12 million MT year^{-1} (Figs. 3a).

The formulation of these N fertilizer has changed with time. Use of NH_4NO_3 declined sharply after 1970, and that of anhydrous NH_4 declined after 1980 (Fig. 3b). Use of urea and that of other mixed N solutions (urea-NH_4–NO_3) have both shown steady increases since 1960 (r^2 = 0.98 and 0.96, respectively (Fig. 3c).

For P, as with N, the most rapid rate of increase was from 1960 to 1980, with ~ 60,000 MT of additional P fertilizer used each year (r^2 = 0.90; Fig. 3d). After a decline from 1980 to 1990, the rate of P use year^{-1} has remained essentially unchanged (slope = 0.0). The current rate of P use is ~ 1.8 million MT year^{-1}.

Phosphorous fertilizers also have changed in composition with time. The use of superphosphates, which were common prior to the 1970s, has declined sharply (Fig. 3e). The most recent years have seen a shift to combined N and P forms, of which monoammonium-P use has increased most rapidly; since 1990 its use has increased at the rate of ~ 80,000 MT year^{-1}

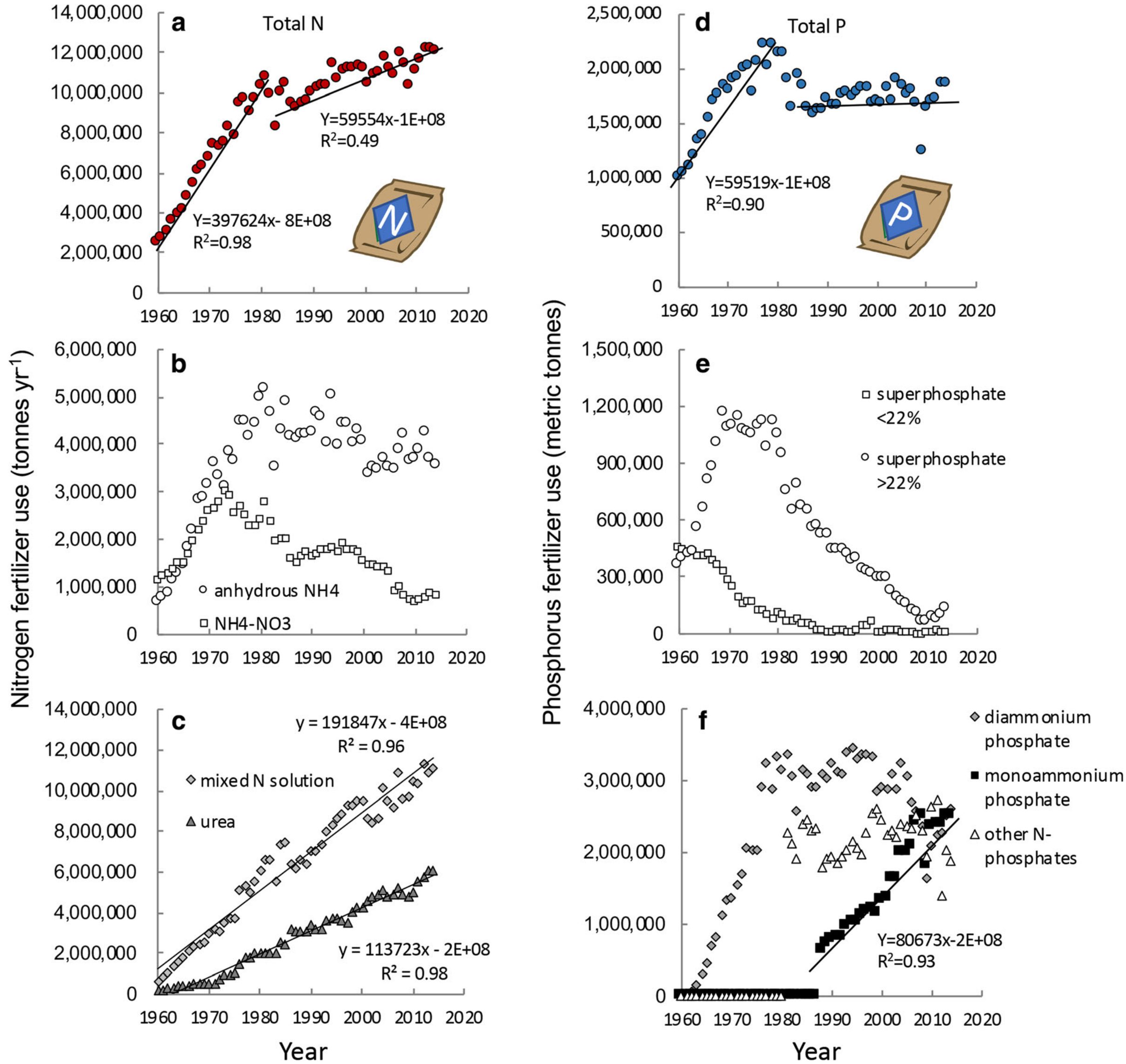


Fig. 3 Change in nitrogen and phosphorus fertilizer use in the US over time as **a** total nitrogen, **b** anhydrous NH_4 and NH_4–NO_3, **c** mixed N solutions (urea-NH_4–NO_3 and urea), **d** total phosphorus (as P), **e** superphosphates, and **f** combined N-phosphorus solutions. Data are from https://www.ers.usda.gov/data-products/fertilizer-use-and-price. Trend lines are shown to highlight specific relationships described in text. Icons are from the UMCES-IAN image library

(r^2 = 0.93), while use of other forms of P have remained essentially flat or have declined (Fig. 3f).

Fertilizer trends by crop

Corn is king, with over 37 million ha planted in this crop as of 2019 (Fig. 4a), yielding 300 million MT (www.nass.usda.gov/Statistics_by_State/index.php, Gunderson et al. 2018). Acreage of corn has increased since the 1970s, and while there was a decline in the early 1980s, there has since been a steady upward trend. Of the three major crops (corn, soybean, and wheat), corn makes up 43–86% of the harvest throughout the country except for the northeast and northwest regions (Regions I and X; Fig. 5a). There are very few states where corn is not grown on an industrial scale (Fig. 6a).

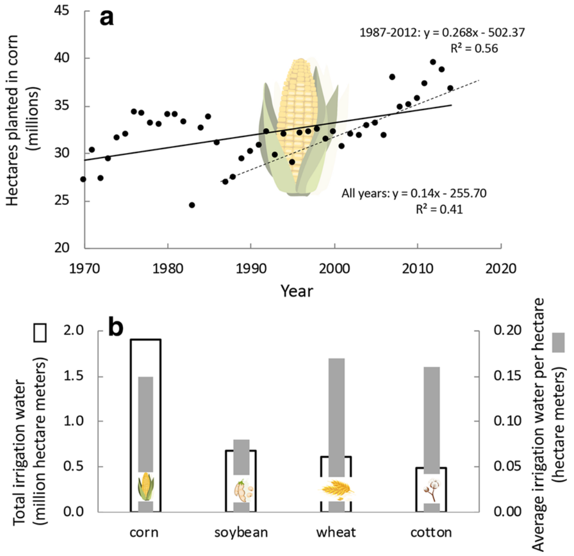


Fig. 4 **a** Hectares planted in corn in the US over time. Trend lines are for time period indicated. Data are from https://www.ers.usda.gov/data-products/fertilizer-use-and-price. **b** Irrigation water applied and per ha water use by crop. Data are from Barton and Clark (2014) based on the USDA 2008 Census of Agriculture. Icons are from Vectorstock used under an expanded license

Corn is also the most intensively fertilized crop. From the 1990s to present, N fertilization rates for corn have hovered in the range of 140–160 kg ha^{-1} or a total of over 5,500,000 MT year^{-1} (Fig. 6b). As is the case with all crops considered here, fertilizer is often used at a rate that exceeds the agronomic demand by more than 25%; this is to ensure the best yield under ideal conditions. From 1996 to 2010 (most recent data available), for more than 50% of crops planted, the rate of N application was greater than 25% above the plant's agronomic need (USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7a). Use of P on corn declined after the 1970s, but has increased about 10% from 2000 to 2014 to 823,000 MT year^{-1} or ~ 30 kg ha^{-1} (Fig. 6c). For 25–50% of crops planted (1996–2010), the rate of P application was greater than 25% above the plant's agronomic need (USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7b). The yield of corn has steadily risen from the mid-1980s, with just over 10 MT ha^{-1} now produced (Fig. 6d). Corn also uses the most water for irrigation, although on a ha^{-1} basis, it is comparatively more efficient than other crops considered herein (Fig. 4b).

Soybean, also grown in the Midwest and eastern states (Fig. 6e), makes up 7–26% of the harvest of the three major grains except in the northeast and west coast (Regions I, IX, X), (Fig. 5a). Over 100 million MT are harvested annually (www.nass.usda.gov/Statistics_by_State/index.php). As a legume, it does not need much N fertilization (except in early growth stages), and the amount of N applied to soybeans declined from a peak in the late 1990s, but has risen again in the most recent years, to 184,000 MT (Fig. 6f). Use of P has remained nearly constant in the range of 20–25 kg ha^{-1} over the recent decades, but a spike in P application to 329,000 MT was observed in the most recent years (Fig. 6g). For 10–15% of crops planted (1996–2010), the rate of P application was greater than 25% above the plant's agronomic need (USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/. Fig. 7b). Yields of soybean, like those of corn have steadily increased over time (Fig. 6h). Soybean are among the most water efficient crops on a ha^{-1} basis (Fig. 4b).

Wheat is grown throughout the US. In the upper northwest, where both winter and spring crops are planted (Fig. 8a), it makes up 84% of the major crops harvested (Fig. 5a). Over 40 million MT are harvested annually (www.nass.usda.gov/Statistics_by_State/index.php). Use of N on wheat has more than doubled over the decades, from ~ 30 kg ha^{-1} in the 1960s to 78 kg ha^{-1} most recently, with a total N application of 1,437,000 MT (Fig. 8b). For 35–50% of crops planted (1996–2010) the rate of N application was greater than 25% above the plant's agronomic need (USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7a). Use of P on wheat reached a peak in the late 1970s, and has declined slightly since then, now at a rate of 242,000 MT (Fig. 8c). For approximately 25% of crops planted (1996–2010), the rate of P application was greater than 25% above the plant's agronomic need (USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7b). Data on yields for the past decade reveal little change (Fig. 8d). Wheat requires about twice the amount of irrigation water on a ha^{-1} basis than does soybean (Fig. 4b).

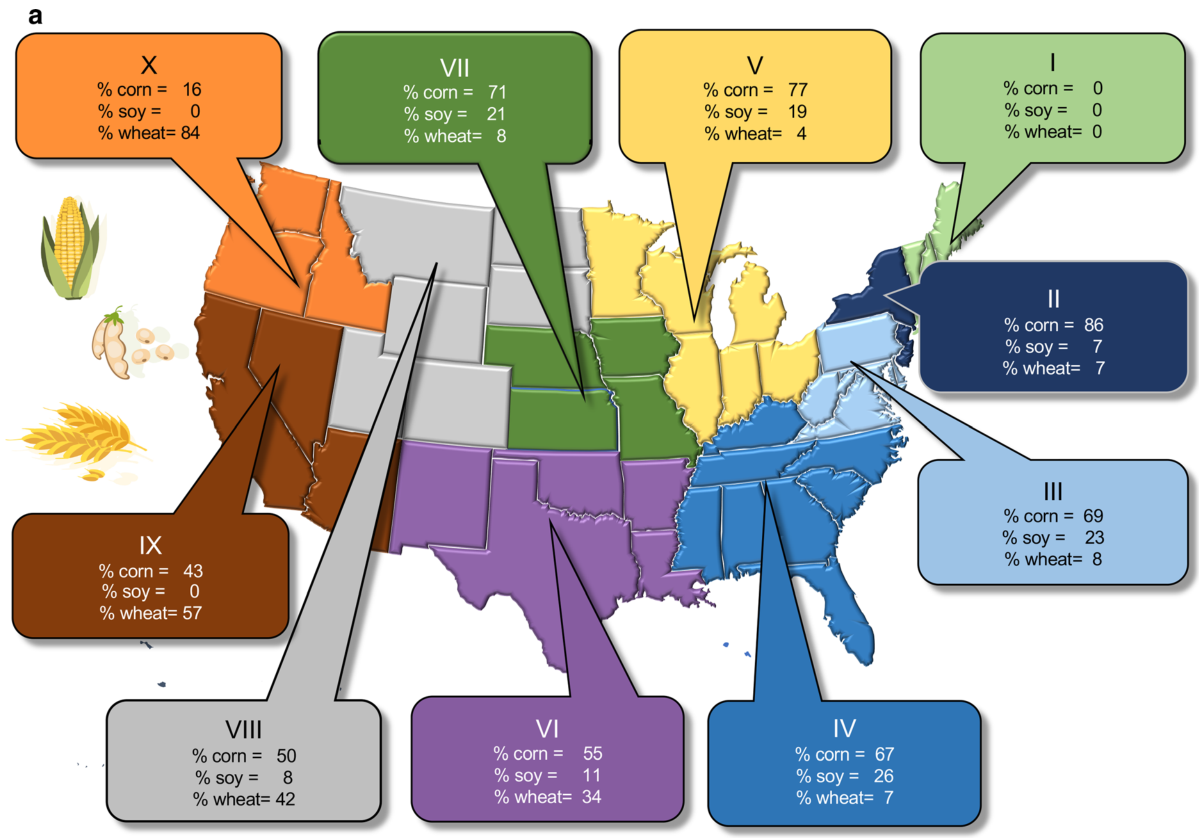


Fig. 5 **a** Percent of corn, soybean and wheat grown in the 10 regions of the US designated by the Office of Management and Budget (see also Online Resources Fig. S1). **b** Percent of cattle, dairy, hogs and poultry production for the same US regions, as based on equivalent animal units (see text for explanation). Data are from 2019 from https://www.nass.usda.gov/Statistics_by_State/index.php. Symbols and icons are from Vectorstock used under an expanded license

Cotton is grown in the southern states (Fig. 8e). Applications of N to cotton have remained at roughly 100 kg ha^{-1} for the past decades (Fig. 8f), a rate of N application that was more than 25% above the plant's agronomic need for more than 65% of crops (through 2007; USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7a). Use of P on cotton has steadily declined from > 60 kg ha^{-1} in the 1960s to 45 kg ha^{-1} most recently, with the most recent application being a total of 39,000 MT (Fig. 8g). Application rates are more than 25% above the plant's agronomic need for more than 50% of crops planted (through 2007; USDA 2019; https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/; Fig. 7b). Yields of cotton have also increased over time (Fig. 8h). Cotton requires comparatively slightly more irrigation water than corn on a ha^{-1} basis, but its overall irrigation demands are far less due to the overall planted acreage (Fig. 4b).

Fertilizer trends by region and state

Regions V and VII are the most heavily fertilized regions, and fertilizer application rates for these regions increased by 32% and 31% for N and by 4.3% and 25% for P from 2002 to 2012 (Fig. 9a,b). Although overall application rates are less in Region VIII, the rate of increase from 2002 to 2012 of both N and P was greater, 64% and 34%, respectively (Fig. 9a,b). Application rates of N and P declined in Regions IV, VI, and IX over this same period. In every region of the US, the N:P of fertilizer application increased from 2002 to 2012 (Fig. 9c).

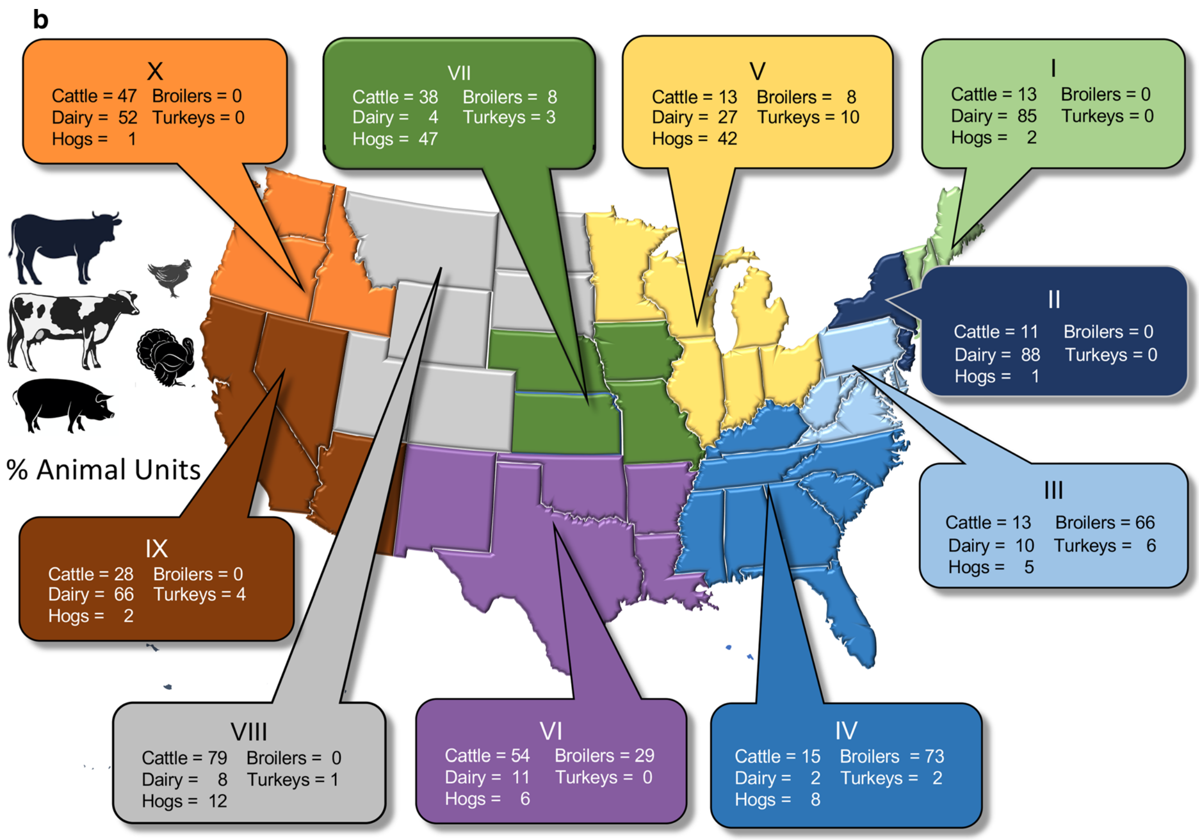


Fig. 5 continued

State-by-state fertilizer use statistics are summarized in the Online Resources material based on 2011 data (Online Resource Fig. S3). Iowa applies N and P more intensively than any other state. As of 2011, its rate of N use was > 1.2 million MT year^{-1}, while its rate of P use was $\sim$ 200,000 MT year^{-1}. In addition to Iowa, the top states in terms of N usage include Illinois, Nebraska, California, and Minnesota, while the top states for P fertilizer use include, in addition to Iowa, Minnesota, Illinois, Nebraska, and South Dakota.

Animal operations

In 2019, the US produced approximately 8.7 billion animals annually in CAFOs, the vast majority being in chickens (Fig. 10a, b). In the 15 years from 1997 to 2012, the number of cattle (on farms with > 500 head) increased 4.3%, dairy cows (on farms with > 500 head) increased 121%, hogs (on farms with > 1,000 head) increased 37%, broiler chickens (on farms producing > 500,000 chickens annually) increased 80% and layers (on farms with > 100,000 hens) increased nearly 25% (Food and Water Watch 2015). This was a net increase of approximately 1 million cattle, 300,000 dairy cows, nearly 14 million hogs and over 250 million broilers, or the equivalent to adding 550 animals every day for 15 years, for hogs adding 3,000 animals every day for 15 years, and for broiler chickens, adding 85,000 chickens every day for 15 years (Food and Water Watch 2015). From 2012 to 2019 cattle increased 13%, dairy cows and broiler chickens $\leq$ 1%, while hog production increased 13%. During this same time, turkey production decreased 30% (Fig. 10). Thus, the increase in hog production proceeded at about the same rate as pre-2012, adding the equivalent of 3,000 animals or more per day from 2012 to 2019.

Based on animal units, dairy production dominates in the northeast (Regions I,II), broiler production in the southeast (Regions III, IV), hog production in the Regions V, VII, cattle in Regions VI,VII and VIII,

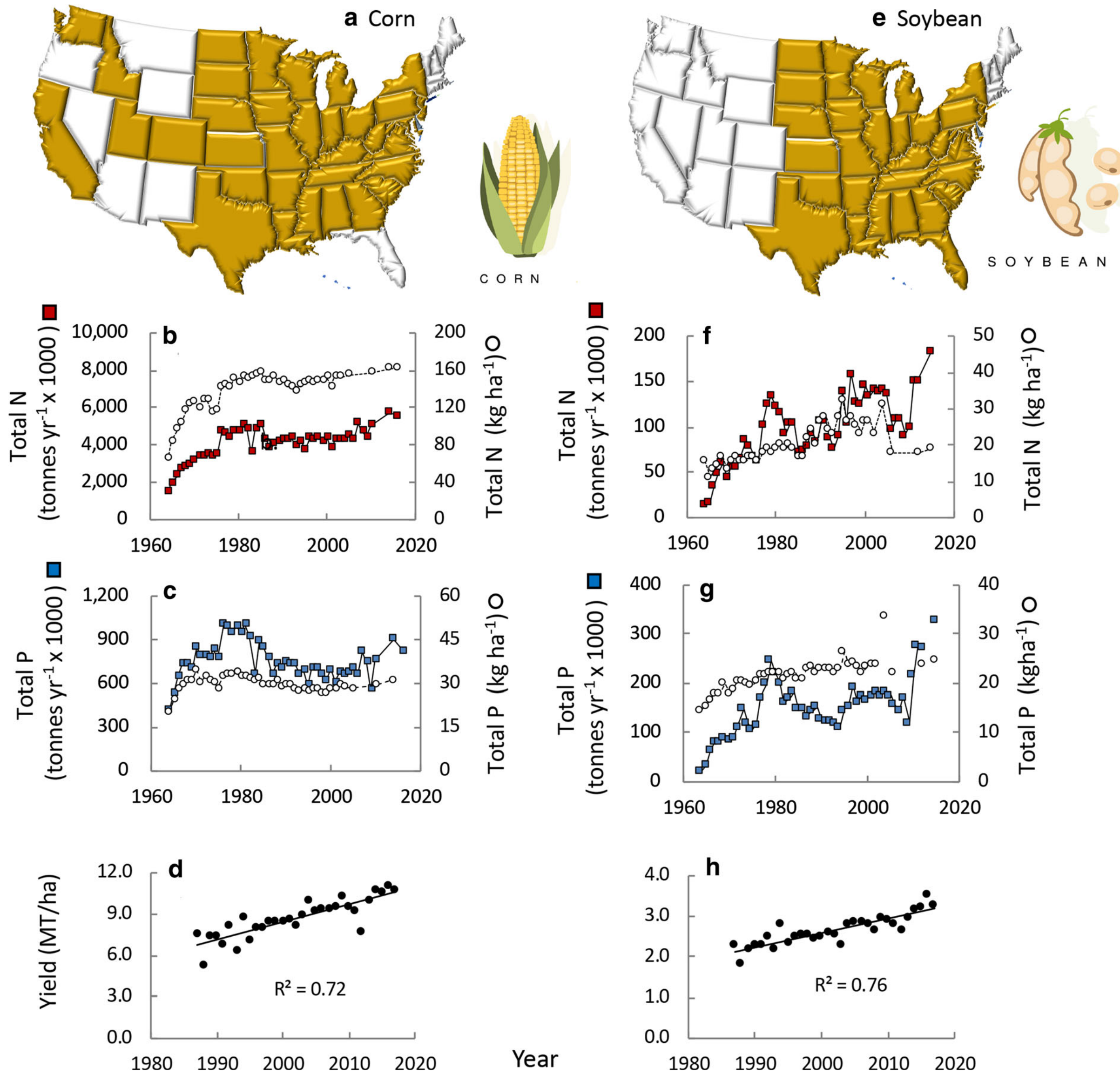


Fig. 6 **a** States growing corn, **b** total N fertilizer used on corn over time (squares) and amount per ha (circles); **c** total P fertilizer used on corn (squares) and amount per ha (circles), **d** yield of corn per hectare; **e–h** comparable relationships for soybean. Data are from https://www.ers.usda.gov/data-products/fertilizer-use-and-price. Symbols used are from Vectorstock used under an expanded license

while dairy production again dominates in the west (Regions IX, X) (Fig. 5b). State-by-state animal population statistics for 2019 are summarized in the Online Resources material (Online Resources Figs. S4 and S5). Note that these statistics are likely underestimates of the total confined animal populations, as described above (and these statistics do not include populations of animals beyond the groups considered here). Georgia, Alabama, and Arkansas produce over 1 billion broilers annually, Texas has the largest number of cattle, over 4.6 million not including calves, and Kansas, Nebraska, and Texas together account for $> 60\%$ of cattle in feedlots (www.aphis.gov/animal_health/nahms/downloads/Demographics2017.pdf). California has the largest number of dairy cows, over 1.7 million (Online Resource Fig. S4), and Iowa has the largest numbers of hogs, with 23 million, outpacing North Carolina, with the next largest populations

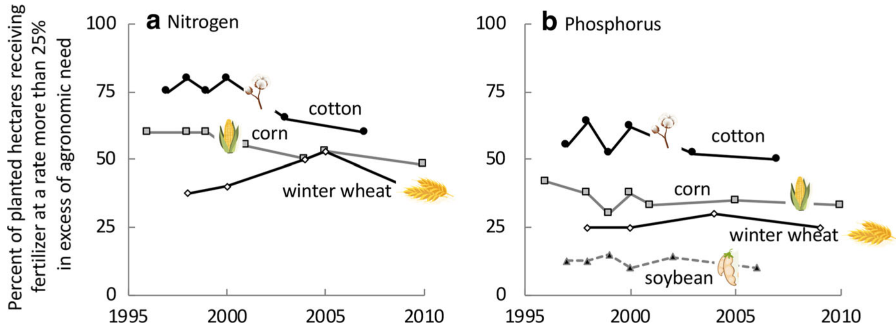


Fig. 7 Percent of hectares planted in crop indicated receiving **a** nitrogen or **b** phosphorus fertilizer more than 25% above the recommended agronomic need of the plant. Replotted from https:www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/. Symbols used are from Vectorstock used under an expanded license

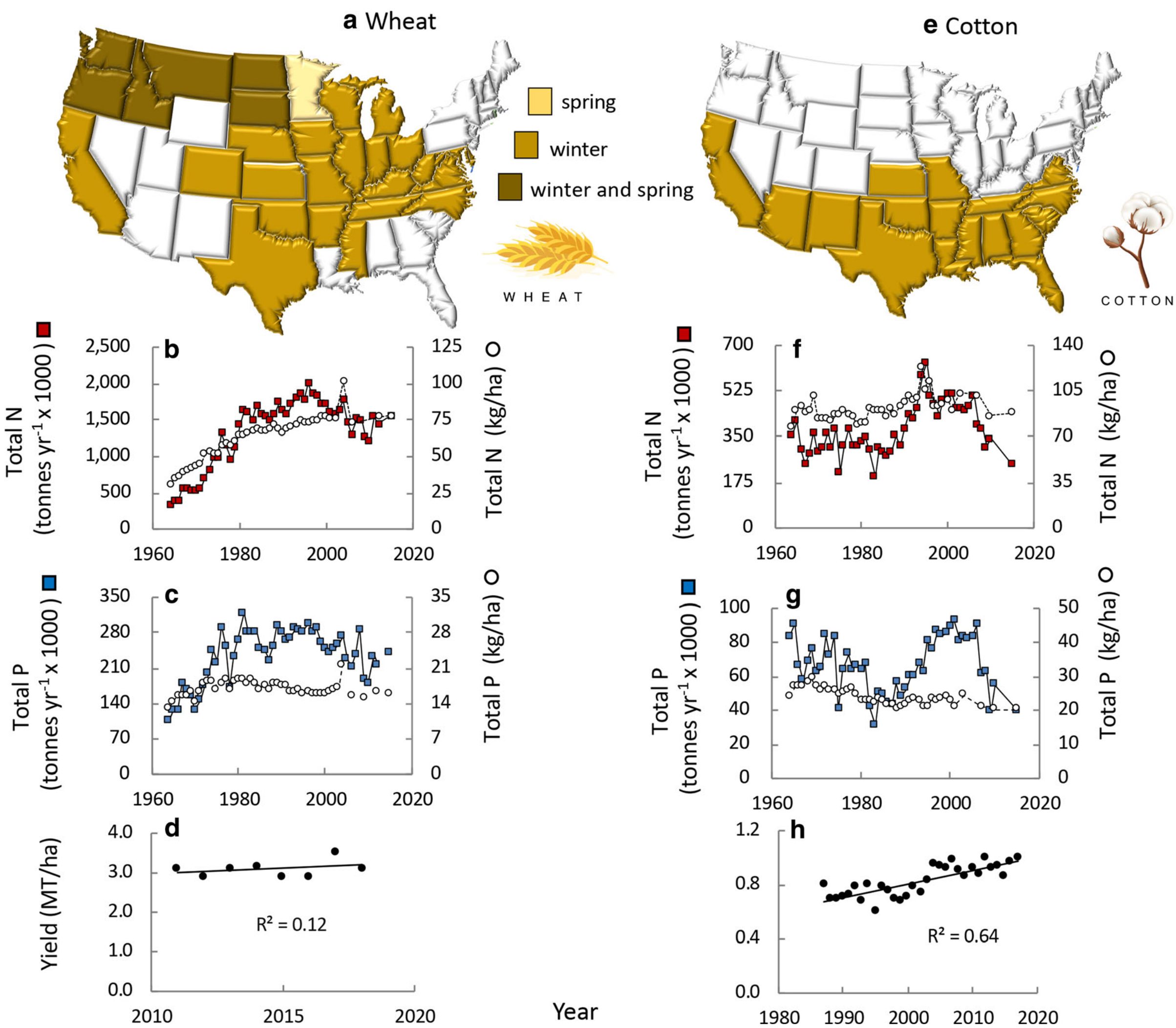


Fig. 8 As for Fig. 6 except for **a**–**d** wheat and for **e**–**h** cotton

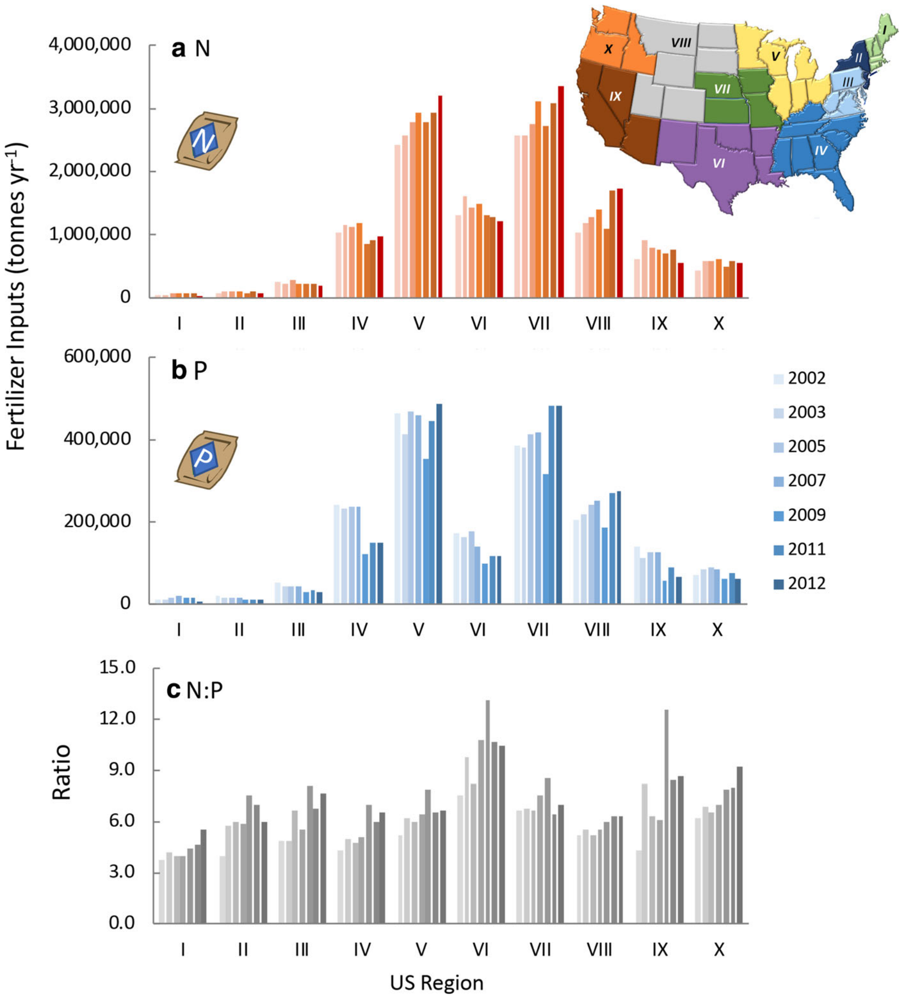


Fig. 9 **a** Nitrogen fertilizer purchased by region of the country from 2003 to 2012 and percent change (**b**) As for (**a**) except for P fertilizer. Data from 2003 to 2011 are from the US EPA (https://www.epa.gov/nutrient-policy-data/commercial-fertilizer-purchased). Data for 2002 and 2012 for N were obtained from Sabo et al. (2019), and data for P for the same years are from US EPA (https://doi.org/10.23719/1504278). **c** The ratio of N:P (by weight) for the same years. The 10 regions of the US are as designated by the Office of Management and Budget (see also Online Resources Fig. S1). Note that Hawaii is included in Region IX and Alaska in Region X

of these animals, by more than a factor of 2 (Online Resource Fig. S4). The largest region for broiler production is the southeast, with Georgia, Alabama, Arkansas, North Carolina and Mississippi the 5 largest producing states (Online Resource Fig. S5). Turkeys are produced in 13 states, with Arkansas, Minnesota, and North Carolina the largest producers, each with $> 20{,}000{,}000$ animals produced year^{-1} (Online Resource Fig. S5).

As of 2018, the US had over 20,000 CAFOs, a number that has increased $\sim$ 8% in the past decade, but a number that likely underestimates the true value (Fig. 11a; Online Resources Fig. S6a). The highest concentration of CAFOs is in Region VII with over 5,800, followed by Regions IV with 3621, and Region V with 3409 (Fig. 11b). The largest expansion in such operations was in Region VII, where 69% more CAFOs, and in Region III, 115% more CAFOs, now operate compared to a decade ago (Fig. 11b). States with over 1000 CAFOs in 2018 include Texas, California, Nebraska, North Carolina, Minnesota, and Iowa, which has the highest number overall, with > 3500 (Online Resource Fig. S6a). States with the largest increases in CAFOs from 2011 to 2018 were

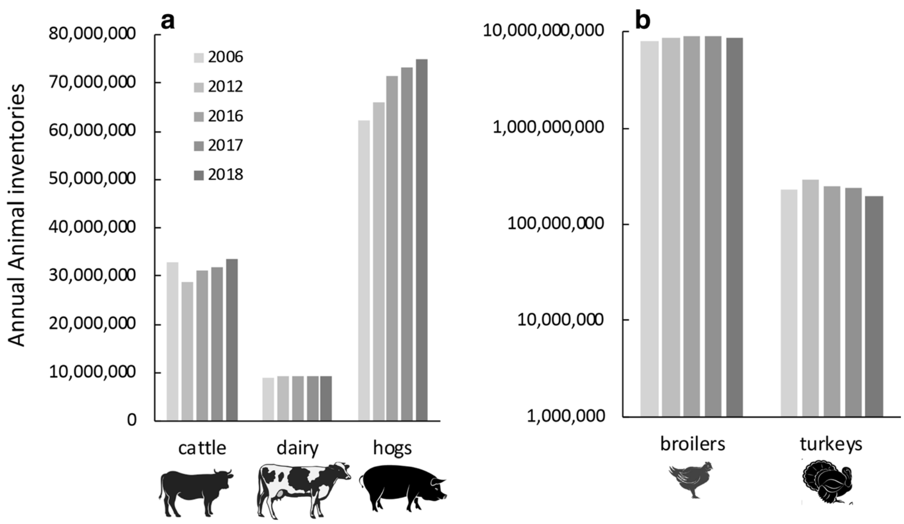


Fig. 10 Change in the number of animals by type in medium and large-sized CAFOs in 2019. **a** Numbers of cattle, dairy cows and hogs, and **b** broiler chickens and turkeys. Note the log scale for panel (**b**). Data from 2019 are from USDA (www.nass.usda.gov/Statictics_by_State/index.php). Symbols used are from Vectorstock used under an expanded license

Maryland and Delaware, in chickens, and Iowa, in hogs (Online Resource Fig. S6b).

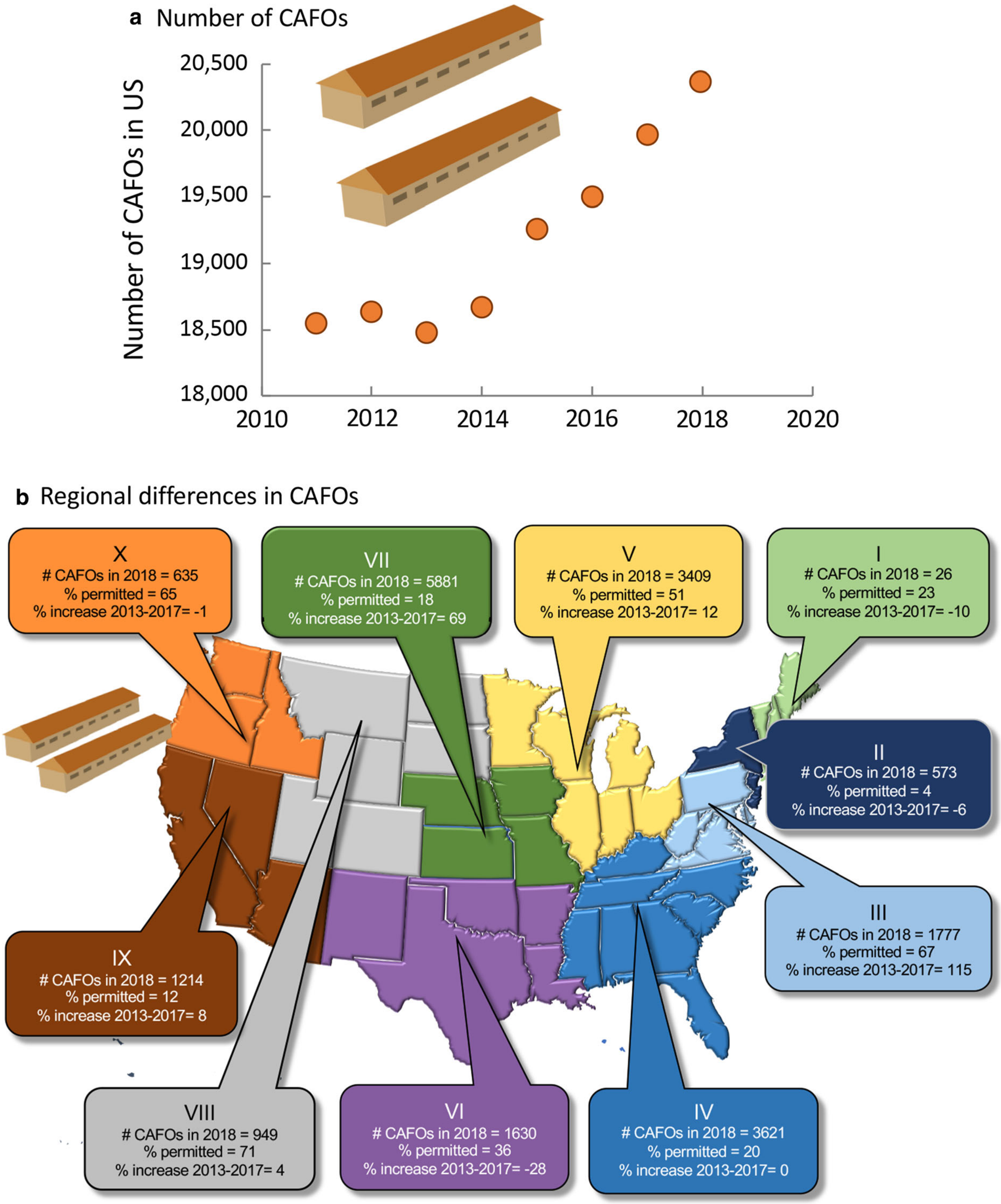
In terms of permitting, the National Pollutant Discharge Elimination System (NPDES, the regulation system authorized by the Clean Water Act) requires that all CAFOs that discharge to a waterbody have NPDES permit coverage (40CFR 122.23(d)(1)). As a consequence, the portion of CAFOs that need NPDES coverage can vary from state to state depending on size, discharge and waste management systems. On average across all states, only 32% of CAFOs are permitted under the NPDES regulations. Regions I, II, IV, VII, and IX had fewer than 20% of operations permitted, while regions III, V, VIII, and X had over 50% of operations permitted (www.epa.gov/sites/production.files/2019-09/documents/cafo_tracksum_endyear_2018.pdf). Iowa, with over 3,700 CAFOs, has permits for just 3%, and North Carolina, with over 1200 CAFOs, has permits for 1%; these are the top 2 states for hog production (Online Resources Fig. 6c). Of the 8 states with the largest CAFOs, 24% have permits. States with higher production of chickens, such as Maryland and Alabama, have much higher permitting percentages.

Cattle operations are concentrated in the Midwest and the largest expansion in cattle CAFOs from 2011 to 2017 were in Missouri and Colorado (Online Resource Fig. S7). Increases in dairy were concentrated in the southwest and upper Midwest, with Texas, Missouri, Colorado, Kansas and South Dakota increasing production by close to, or more than, 20% (Online Resource Fig. S8a–c). Hog production decreased in the southwest but became more concentrated in the upper Midwest from 2011 to 2017 (Online Resource Fig. S8d–f). Virtually every county in Iowa is now in intensive hog production (Online Resource Fig. S8f). Broilers remain concentrated in the southeast, but Ohio increased production by $>50\%$ (Online Resource Fig. S8g-i).

Manure quantities

In most regions of the US, total N and P from manure increased from 2002 to 2012 (Fig. 12a, b). In Regions IV-XIII, $>400{,}000$ MT year^{-1} manure N are released, while in Regions IV–VII, $>200{,}000$ MT year^{-1} manure P are released. The N:P ratio (by weight) of manure is lowest in Regions III and IV (Fig. 12c) and for each region has not changed substantially over this time period.

Based on the animal inventory of 2019, over 4 million MT of manure as N was produced from all animals in confinement considered herein. Applying the conversion factors of Ruddy et al. (2006), $\sim$ 44% was from cattle, $\sim$ 17 18% from dairy cows, hogs, and broilers, and 3.9% from turkeys (Fig. 12d). Applying the conversion factors of Bouwman et al. (2017), the contribution from dairy is nearly twofold



higher, that from cattle and hogs slightly higher, while that from broilers ∼ 30% lower.

For the same time period, over 1.4 million MT year^{-1} of manure as P was produced. Applying the conversion factors of Ruddy et al. (2006), cattle

◀**Fig. 11** **a** Total US changes in CAFOs from 2011 to 2018, **b** numbers of CAFOs by US region in 2018, their percent change from 2013 to 2017 and percent permitted. Data are from EPA (https://www.epa.gov/sites/production/files/2019-09/documents/cafo_tracksum_endyear_2018.pdf) and USDA as summarized by Walljasper (data 2011–2017, https://investigateMidwest.org/2018/06/07/large-animal-feeding-operations-on-the-rise/). Symbols used are from Vectorstock used under an expanded license. The 10 regions of the US are as designated by the Office of Management and Budget (see also Online Resources Fig. 1). Note that Hawaii is included in Region IX and Alaska in Region X

produced 45%, hogs and broilers each 20–23%, dairy cows nearly 8%, while turkeys just 4.3% of this P (Fig. 12e). The Bouwman et al. (2017) conversion factors yield values ∼ 40% lower for cattle, hogs and broilers, but higher values for diary.

Regions IV, VI, VII, and VII produced the most N from cattle, Regions V and IX from dairy cows, Regions IV, V, and VII from hogs, and Regions III and IV from broilers (Regions 12f–i). Regions III, IV and V were the largest turkey production regions (not shown).

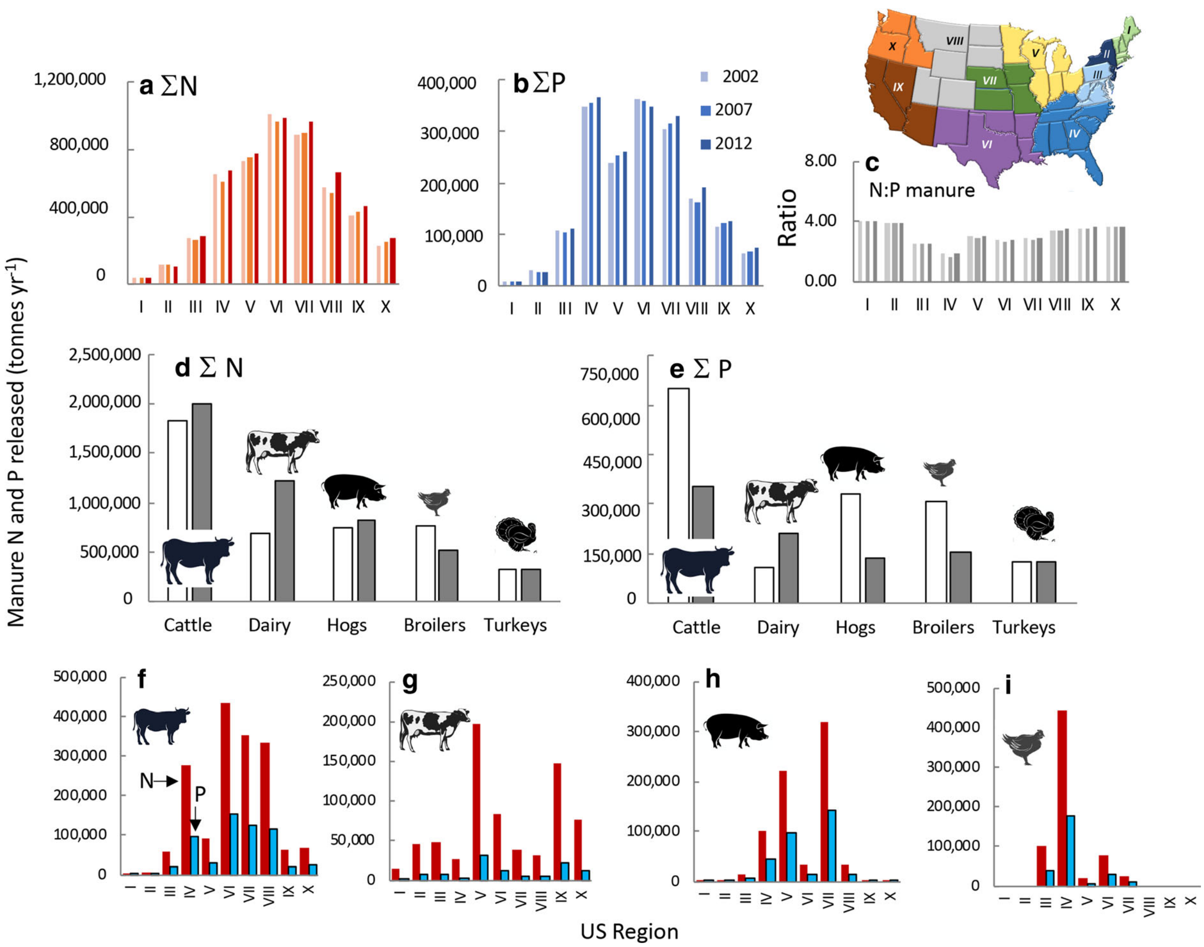


Fig. 12 Daily amount of excretion of manure as **a** N, **b** P, and **c** N:P ratio by weight by US region. Data are for 2002, 2007, and 2012 and were derived from Sabo et al. (2019) for N and US EPA (https://doi.org/10.23719/1504278) for P. The upper inset map shows the US regions. Panels **d**, **e** are calculated data for N and P released as manure by animal type for 2019 (data from USDA www.nass.usda.gov/Statictics_by_State/index.php). Open bars represent values calculated using conversion factors reported by Ruddy et al. (2006); closed bars represent values calculated using conversion factors reported by Bouwman et al. (2017). Panels **f**–**i** show the same 2019 data by US region (applying Ruddy et al. 2006 conversions). The 10 regions of the US are as designated by the Office of Management and Budget (see also Online Resources Fig. S1). Note that Hawaii is included in Region IX and Alaska in Region X. Symbols used are from Vectorstock used under an expanded license

Ammonia emissions

There are two major sources of NH_3 emissions from agricultural operations. It is emitted from fertilizer applications, especially when those applications are NH_4- or urea-based, and from management of manures. Emissions summaries are available by state in the Online Resources (Online Resource Fig. S9; https://www.epa.gov/air-emissions-inventories/2017-national-emissions-inventory-nei-data). Emissions have not only fluctuated with time, generally showing a decline from 2008 to 2014, but the methodology for reporting has changed slightly over time and thus there is high variability in these data from year to year by region (Fig. 13a). Emissions of NH_3 from fertilizer applications ranged from very low in the northeast to a high of over 300,000 MT year^{-1} in Region VIII in 2014 (Fig. 13a). Region VIII also produces the highest NH_3 emissions from livestock waste, with values over threefold higher than those from fertilizer applications (Fig. 13b). Based on data from 2014, states with the largest NH_3 emissions from fertilizer, $>$ 50,000 MT year^{-1}, included California, Texas, Kansas and Illinois (Online Resource Fig. S9a), and those with the largest emissions from livestock waste, $>$ 100,000 MT year^{-1}, include California, Texas, Iowa, and North Carolina (Online Resource Fig. S9b).

Based on the animal inventory of 2019, a total of $>$ 4,500,000 MT year^{-1} of NH_3 were emitted (Fig. 13c). Of this, broilers and turkeys made the largest contribution. This value was derived using a conservative emission factor for broilers, and would be significantly greater if a higher emission factor were applied.

Greenhouse gas emissions

In 2017, the agriculture sector emitted 542 million MT CO_2 Eq (using equivalencies reported by the IPCC Fourth Assessment Report 2007), representing 8.4% of US greenhouse gas total emissions. Direct and indirect emissions, largely as N_2O from soils, contribute substantially to this agriculture component of greenhouse emissions (Fig. 14a,b). Most of this comes from cropland compared to grassland. Although there are interannual variations, the change from 1990 to 2017 in this source was only 6% (Fig. 14b).

Enteric fermentation accounts for the largest fraction of CH_4 emissions from the agriculture sector (Fig. 14c). Of the total production of ruminant animals, cattle were the largest contributors from enteric fermentation (Fig. 14c). From 1990 to 2017, there was an increase in total enteric fermentation emissions of 6.9%, and year-to-year fluctuations in emissions per head per type of animal are attributed to changes in animal diets among other factors. In sharp contrast to the comparatively small percentage change in greenhouse gas emissions over the past decade due to enteric fermentation, there has been a sharp rise in greenhouse gas emissions due to manure management. Emissions of CH_4 from manure management increased 66% from 1990 to 2017 (that from dairy increased 134%, cattle 9.6%, hogs 29% and poultry 3%), while those of N_2O increased 34% over the same time period (dairy 15%, cattle 46%, hogs 58%, and poultry 14%; Fig. 14d,e).

Texas has the highest greenhouse emissions overall (Online Resource Fig. S10a), while California, Idaho, Iowa and North Carolina have the largest CH_4 emissions (Online Resource Fig. S10b), with emissions of the first 2 states largely due to dairy and emissions of the latter two states mostly due to hogs. Kansas, Nebraska and Texas have the largest N_2O emissions due to cattle (Online Resource Fig. S10c).

Human population and wastewater

As of mid-2019, the US human population was 328,557,738 persons (https://worldpopulationreview.com/states/). California is the most populous state, Wyoming the least (Online Resource Fig. S11a). Since 2010, states that have experienced a $>$ 10% increase in population include Texas, Florida, Washington, Arizona, Colorado, Utah, Nevada and Idaho. Only Illinois, Connecticut and West Virginia have undergone population declines over this period. Due to the size of the state and its large population, wastewater from California's urban areas contribute more than any other state.

Based on the human wastewater estimates of Sabo et al. (2019) for N and the US EPA for P, aggregated by region, wastes for both elements are highest from Regions IV, V, and IX (Fig. 15a,b; Online Resource Fig. S11b, c). Wastewater N has increased from 2002 to 2012 in virtually all regions, but wastewater P in some regions has declined (Fig. 15b). Accordingly, wastewater N:P proportions increased from 2002 to 2012 in all but Regions IV, VI, VII, and IX (Fig. 15c).

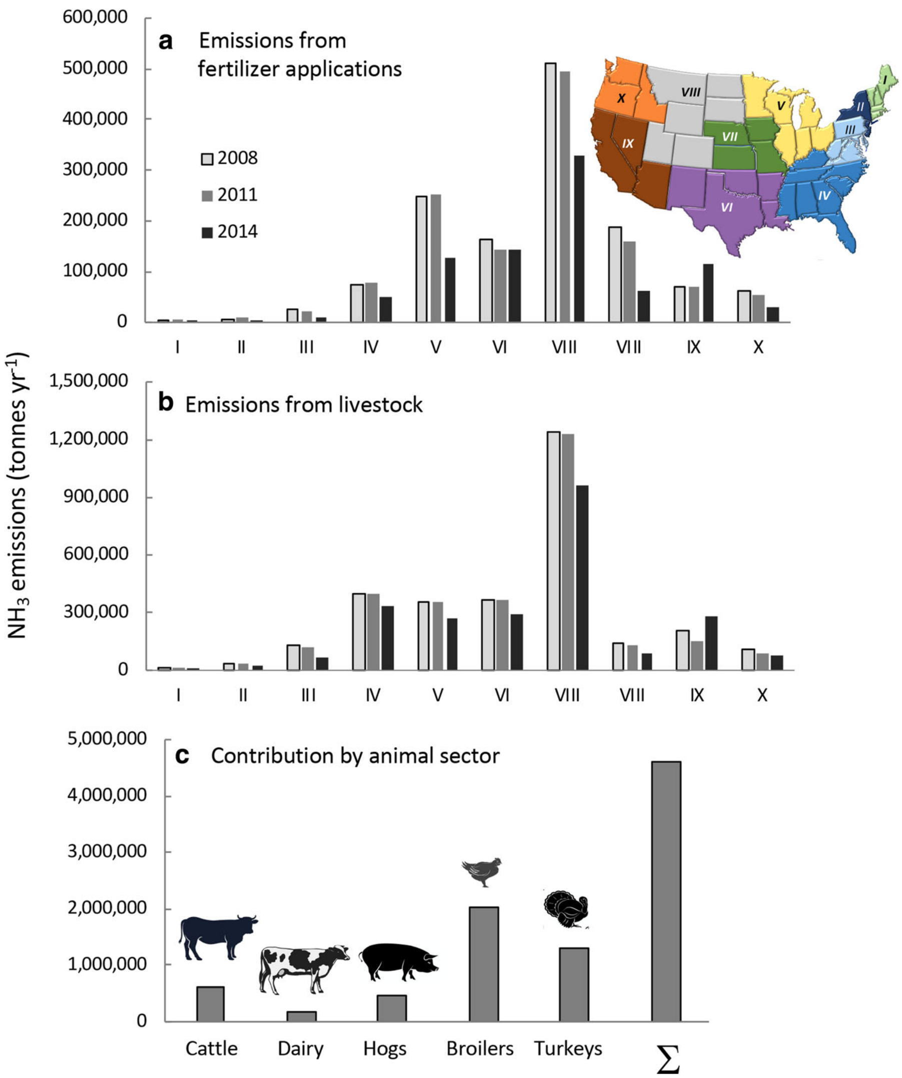


Fig. 13 **a** NH_3 emissions from fertilizer applications and **b** from livestock (as total MT) for different regions of the country and recent changes, and **c** emission for 2014 by animal type. Data were derived from the EPA National Emissions Inventory (NEI) (https://www.epa.gov/air-emissions-inventories/2017-national-emissions-inventory-nei-data). The 10 regions of the US are as designated by the Office of Management and Budget (see also Online Resources Fig. S1). Note that Hawaii is included in Region IX and Alaska in Region X. Symbols used are from Vectorstock used under an expanded license

Statistics are also available on the investment needed in wastewater infrastructure by state anticipated over the next 20 years (infrastructurereportcard.org). These data give some clues as to the level of wastewater treatment. States have widely varying infrastructure needs for wastewater treatment in the next 20 years, but overall, those states with the most rapid growing population have proportionately lower estimated infrastructure costs (Online Resource Fig. S11d). California, Texas, Florida, New York Ohio and New Jersey all have needs exceeding $10 million over the next 20 years, but on a per-person basis, the largest costs, > $1500 per person over 20 years, are estimated for New York, New Jersey, Missouri, Maryland, West Virginia, Hawaii, and Rhode Island (Online Resource Fig. S11d).

Summary comparisons of N and P sources by region

For the country as a whole, fertilizer N inputs have been increasing, and total N inputs from this source are

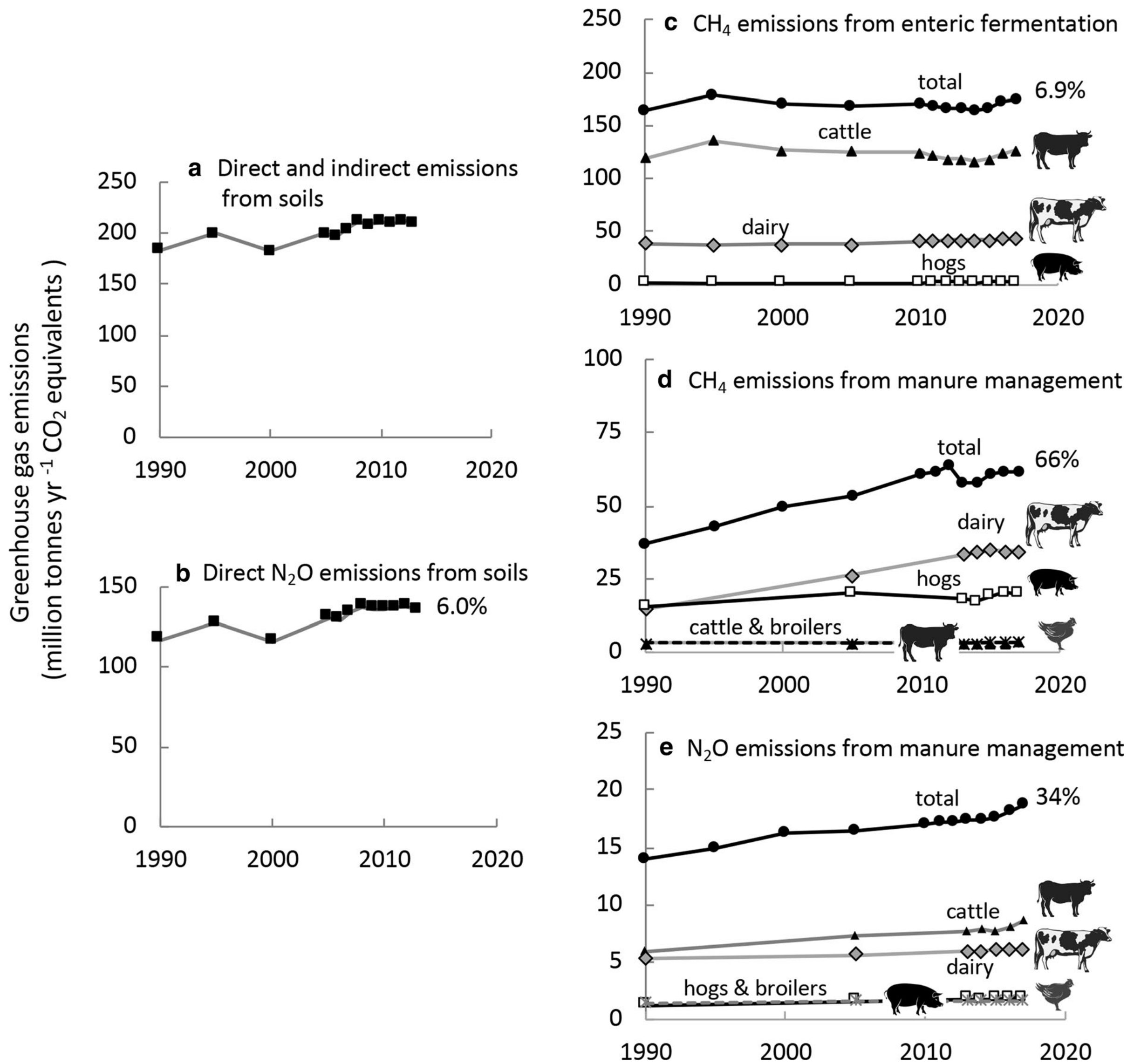


Fig. 14 Greenhouse gas emissions as CO_2 equivalents from **a** direct and indirect sources related to soils: **b** direct N_2O emissions from soils: **c** from CH_4 enteric emissions by animal sector, **d** from CH_4 from manure management by animal sector; and **e** from N_2O emissions from manure management by animal sector. Data are from EPA (www.epa.gov/sites/production/files/2019-04/documents/us-ghg-inventory-2019-main-text.pdf). Symbols used are from Vectorstock used under an expanded license

> twofold those of manure N, > threefold those of atmospheric NH_3, and nearly tenfold higher than those from human wastewater (Fig. 16a). Regionally for 2012, the proportion of fertilizer N inputs relative to human wastewater are very low in the densely populated mid-Atlantic and northeast, Regions I–III, but reach values in excess of 35 in Regions VII and VIII (Fig. 16b). Also, only in Regions I–III are fertilizer inputs less than those of manure N. In all other regions of the country, fertilizer N inputs exceed those of manure by factors ranging from < 2 (Regions IV, VI, and IX) to as high as 4 in Region V (Fig. 16c).

For P, fertilizer and manure P inputs have been roughly on par since the early 2000s, but manure inputs are increasing, while those of P fertilizer have been declining on a relative basis (Fig. 16d). Both of these sources were far in excess of those from human wastewater in 2012. Regionally, fertilizer P inputs

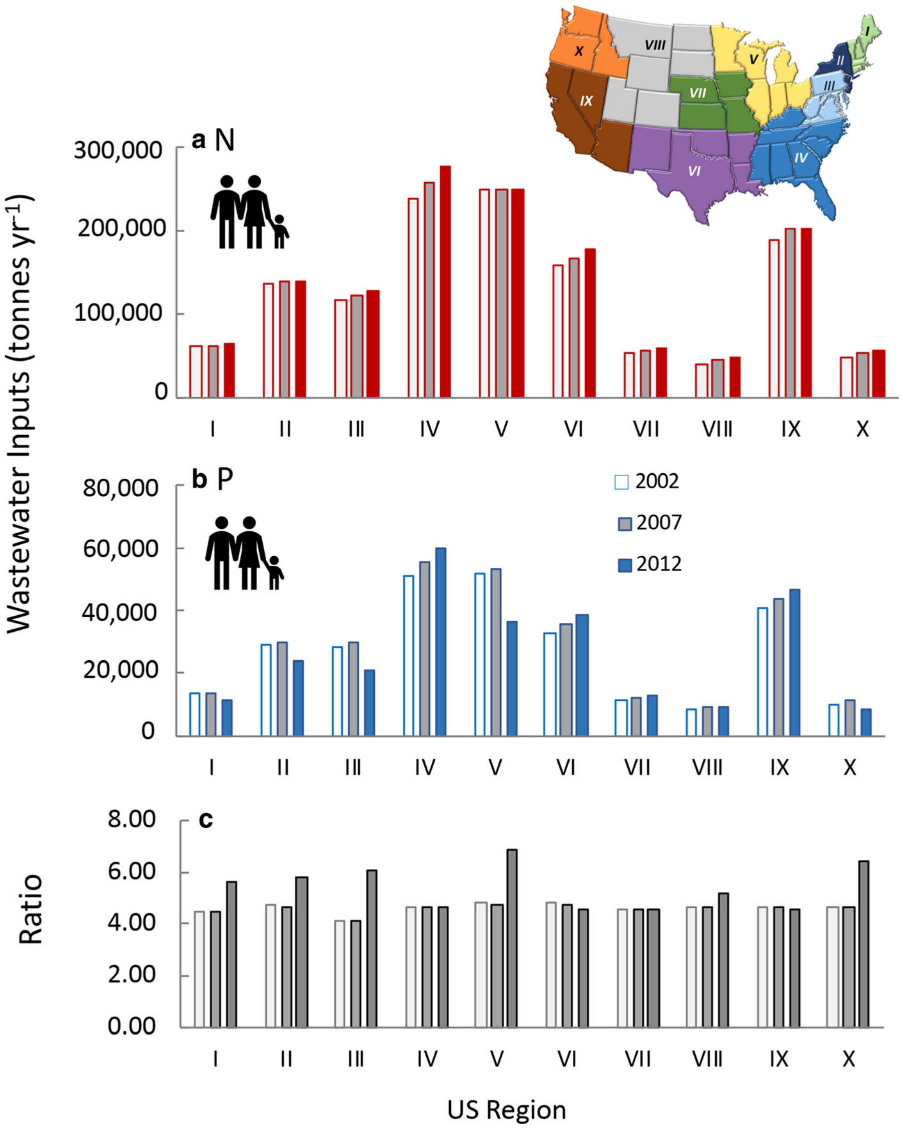


Fig. 15 Human wastewater inputs of **a** nitrogen, **b** phosphorus, and **c** N:P ratio by weight for different regions of the country. Data are for 2002, 2007 and 2012 and were derived from Sabo et al. (2019) for N and from US EPA (https://doi.org/10.23719/1504278) for P. The upper inset map shows the US regions

exceed those of human wastewater by factors > 3 in Regions V, VII, VIII, and X; for all other regions this proportion is < 3 (Fig. 16e). Also, only in Regions V, VII, and VIII did P fertilizer inputs exceed those of manure; for all others, manure inputs of P exceed those of fertilizer (Fig. 16f).

Discussion

Key trends

Farmers have long been considered inherently good stewards of the land. The historical balance that small farmers sustained between animal waste production and crops that fed both animals and people is still the notion that many have with respect to farming (Fig. 17a). This ingrained belief has resulted in agricultural operations having the privilege of

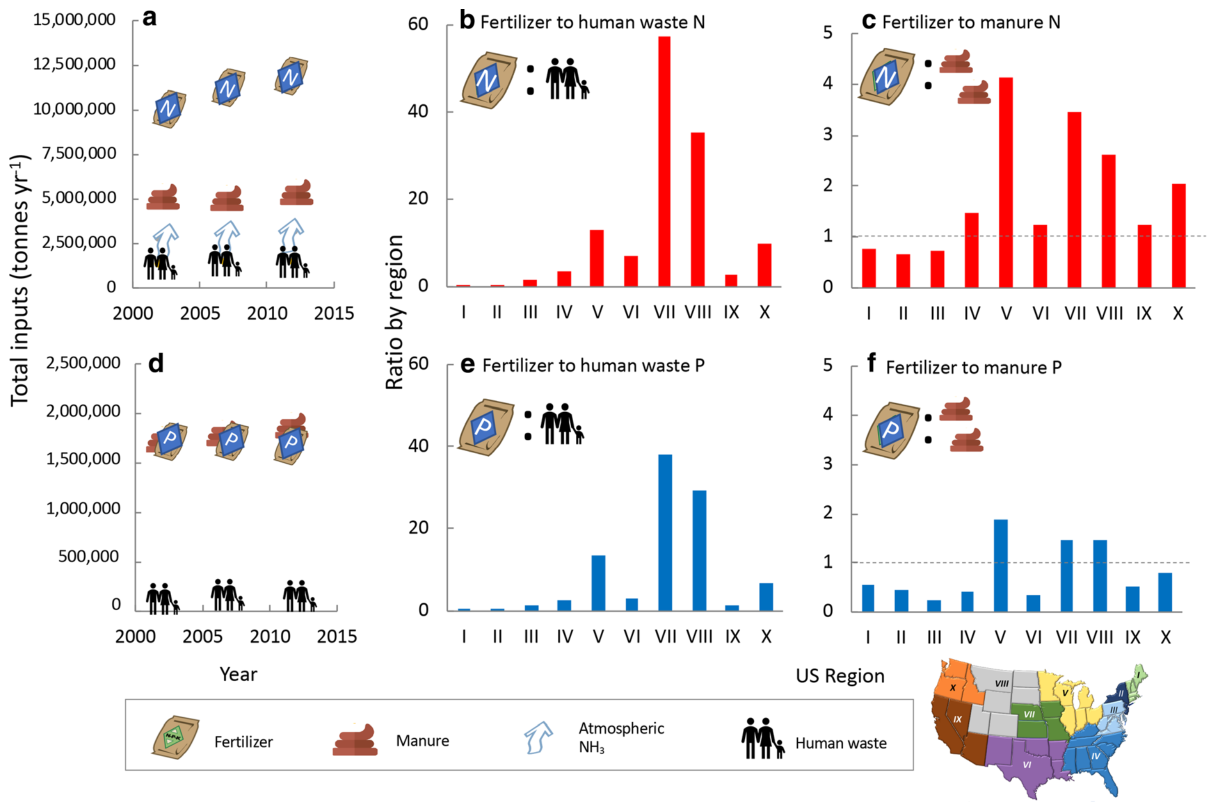


Fig. 16 Comparisons of N and P inputs. **a** Recent changes in N fertilizer, manure N, atmospheric NH_3 and human wastewater for the years 2002, 2007 and 2012 for the entire US. Data were derived from Sabo et al. (2019). Panels **b**, **c** compare fertilizer N to human wastewater N input, and fertilizer N to manure N input, respectively, for different regions of the country. Panel **d** as for panel **a** except for P; data were derived from US EPA (https://doi.org/10.23719/1504278). Panels **e**, **f** are the same as panels **b**, **c** except for P. For panels **c**, **f** a dashed line is shown at a ratio = 1 for reference. The inset map shows the US regions

exemptions of many provisions of environmental laws (Schneider 2010 cited in Tomas 2019). This notion of good stewardship contrasts with current reality and thus, "…rather than reach a middle ground that balanced agriculture and environmental conservation, policy-makers largely yielded to agricultural exceptionalism—nearly every major federal environmental statute passed since 1970 has included carve-outs for farms…" (Ruhl 2000). Now, as the scale of row-crop farms and CAFOs have increased, such good stewardship and environmental nutrient balance within farms can no longer be assumed. Hanson and Hendrickson (2009), citing Stauber et al. (1995) summarized the guiding economic principles of industrialized farming, among which include: "(1) nature is a resource to be exploited and variation is to be suppressed, (2) natural resources are not valued except when a necessary expense in production is incurred, (3) progress is equivalent to the evolution of larger farms and depopulation of farm communities". Farms are now importing fertilizer for crops and feed for animals and the waste production far outpaces that which can be safely recycled back on the land (Fig. 17b). As noted by Pollan (2006), the classically integrated closed ecological loop on traditional farms has been replaced by a disconnected system with a need for increasing chemical fertilizers to support crops and feed for animals, and a resulting manure waste problem from the feedlot.

The effort here is intended to "step back" and to bring attention to recent trends in nutrient sources and that of CAFO proliferation. This paper is hardly the only voice sounding the alarm on the overwhelming nutrient pollution especially from the expansion of CAFOs (e.g., Mallin and Cahoon 2003; Burkholder et al. 2007; Potter et al. 2010; Sakadevan and Nguyan 2016; Rumpler 2016; Martin et al. 2018; Miller and Muren 2019; Pelton et al. 2020 among others). It has

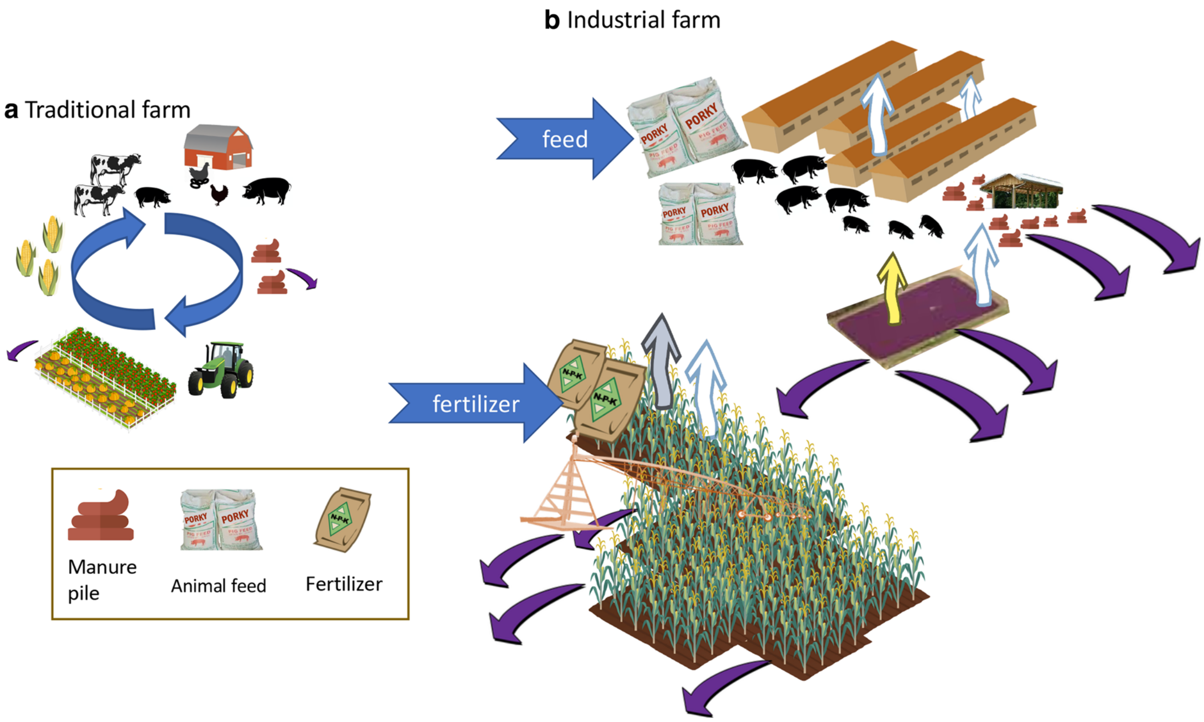


Fig. 17 **a** Classically envisioned nutrient cycle of a traditional farm. Wastes from various animals are used to grow seasonally appropriate crops, and some of this food is used in feeding the animals. **b** On industrial farms, animal populations (typically single species) produce copious manure that is held in waste lagoons and spread on fields of a mono-crop, supplemented with fertilizer. Feed for the animals is tightly controlled and imported on the farm. Wastes from fertilizer runoff and manure N are not recycled but rather lost to the environment. Symbols used are from the UMCES-IAN image library or from Vectorstock used under an expanded license

long been recognized that only a small fraction of agriculturally used or produced N and P (as fertilizer or manure) actually reaches human consumers in the intended food products (e.g., Galloway et al. 2002; Houlton et al. 2013), and roughly half is ultimately lost to the environment in direct runoff and indirect pathways such as atmospheric volatilization and eventual deposition (Galloway et al. 2014). Rather than reporting detailed inventories, the focus here is on total inputs as fertilizer, manure, NH_3 and greenhouse gas production relative to human wastewater. Collectively, this effort—as well as the more comprehensive inventories on which this analysis was based—all underscore that inputs are increasing, nutrient pollution from CAFOs is large and increasing, and highly concentrated in certain regions of the country. Clearly, wastes from the more than annually-produced 8.7 billion animals, mostly confined to nearly 20,000 CAFOs, and 328 million people, combined with roughly 12 million MT of N and 1.8 million MT of P of commercial fertilizer, > 4,000,000 MT of manure as N and > 1,400,000 MT manure as P, along with an estimated > 4,500,000 MT of atmospheric NH_3, spread or deposited annually across nearly 364 million ha of farmland or discharged in local waters, present enormous environmental challenges for the US.

The challenges are amplified when other sources of N and P not considered herein are taken into account. This analysis has conservatively estimated the wastes from CAFOs, as not only the total number of such operations is likely underestimated as noted above, but the waste from many small CAFOs remain un- or under-counted or un-permitted. Several other major pathways of nutrient inputs from the food system were also not addressed here. Meat packing plants, often located near CAFOs and owned by the same companies, contribute substantially to nutrient pollution derived from the blood, urine, feces, fat and meat tissues that are flushed in wastewater streams, yielding high levels of nitrates and other N forms (e.g., Kundu et al. 2013). Moreover, greenhouse gas emissions from the fertilizer industry itself were not included in the analysis herein. Most of the fertilizer produced in the US is either NH_3 or urea, both of which require natural gas and both of which emit CH_4 (Zhou et al. 2019). Although small relative to other sources, CH_4 emissions from this source are estimated to be many-fold higher than the values formally reported from this source (Zhou et al. 2019).

The estimates reported here also have large inherent variability. Many of the conversion factors applied herein have large associated errors. Sales data for fertilizer may not be an accurate reflection of use on specific lands (e.g., Fixen et al. 2015), animal manure conversion factors are changing and fertilizer use efficiencies are improving in some areas (Yang et al., 2016; Sabo et al. 2019). Many farms are also better managed than others. Individual farmers may be applying too little or too much fertilizer or manure, and use efficiencies vary greatly with soil type, moisture, temperature, timing of application, and a myriad of other factors. Practices also vary widely with respect to manure management, including the rate and method by which it is applied to land and environmental conditions at the time of application. Nevertheless, the overarching trends reported here in time and space tell a compelling story of how nutrient pollution is changing and how crop, animal production, and human populations are generally contributing to this pollution throughout the US.

Key trends are that N fertilizer use is increasing relative to that of P, leading to an increase in N:P proportions of total inputs, N fertilizer use exceeds that of manure N inputs, while fertilizer P inputs are more comparable to manure P inputs. Fertilizer P use has been declining in part due to the accumulation of residual P fertilizer in soils over time (e.g., Zhang et al. 2017; Bouwman et al. 2017). Emissions of NH_3, while lower than those of fertilizer input, can be regionally high (even when conservatively estimated), with livestock contributing more than fertilizer volatilization. Greenhouse gas emissions due to manure management have been rising rapidly. Overall, N and P fertilizer input and animal waste far exceeds that of people, except the densely populated northeast and southwest regions. Globally, the ratio of animal feces to human feces has been estimated to be $\sim$ 5 in 2014 and is projected to increase to 6 by 2030 (Berendes et al. 2018). A previous analysis reported that livestock in the US produces 3 times more waste than the US population (US EPA 2003). A similar conclusion was reached by Sabo et al. (2019) for N. Even though total inputs of human waste are less than inputs of fertilizer and manure, the current (2012) estimate is that 45% of municipal wastewater is discharged directly into surface water in the US (Ivahnenko 2017), so this source can be regionally significant.

There have been multiple efforts in recent years to characterize and inventory the N and P budgets at the US national scale, or at a more detailed spatially-explicit level (e.g., Ruddy et al. 2006; Sobota et al. 2015; Houlton et al. 2013; Bouwman et al. 2017; Swaney et al. 2018a, b; Sabo et al. 2019). Ruddy et al. (2006) reported farm and non-farm fertilizer use, livestock manure by animal type and atmospheric deposition for each US county for the years 1982–2001. Yang et al. (2016) examined trends in livestock manure in the US from 1930 to 2012. Swaney et al. (2018a, b) applied the Net Anthropogenic Nitrogen Input model for the US, and more recently, Sabo et al. (2019) reported for each hydrological unit of the US, the N inventories for 2002, 2007 and 2012. The Sabo et al. (2019) approach took into account a comprehensive suite of factors, including human waste, agricultural fertilizer use, and manure production reported here, as well as partial N use efficiency on agricultural lands, N_2 -fixation, lightning, forest fire emissions, fossil fuel combustion, among other factors to derive total N surpluses. Over this time, increased agricultural fertilizer and manure inputs offset estimated reductions in total atmospheric N deposition (Sabo et al. 2019). A similar inventory approach for each hydrologic unit of the US was determined for P (https://doi.org/10.23719/1504278). Global analyses of N and P from agriculture and livestock production have highlighted similar trends (e.g., Bouwman et al. 2013, 2017). That is, N inputs are increasing faster than those of P, they are emitted to the environment via air and water, and due to legacy of nutrient management in agriculture during the 1970s and 1980s, combined with recent changes in inputs, the ratio of N:P exported to fresh and marine waters has increased markedly (Elser et al. 2009; Glibert et al. 2014; Beusen et al. 2015, 2016; Bouwman et al. 2017).

A recent assessment of NH_3 atmospheric concentrations based on passive samplers across the US reported that concentrations have increased over the past decade (Butler et al. 2016). This trend is in spite of the data suggesting little of no trend in NH_3 emissions. The explanation in these contradictory trends may lie in the decline of NO_X and SO_2 emissions, providing less substrate for particulates to form, allowing concentrations of NH_3 to increase even if emissions have not (Butler et al. 2016). Emissions of NH_3 are conservatively estimated here for the most recent animal inventories, using published emission factors (Bowen and Valiela 2001). Estimates of emissions of NH_3 from agricultural system have considerable uncertainty (Beusen et al. 2008), and there are several reasons why new emission factors have been proposed (Pelton et al. 2020). Much larger birds are being grown compared to 15–20 years ago; older estimates are based on European practices of litter management within the flocks and US practices yield twice the NH_3 emission per broiler barn than comparable European barns. Thus, the likely contribution by broilers to NH_3 emissions is a higher percentage relative to other animal sectors and the overall total could be much higher (Fig. 13c).

Eutrophication and algal blooms

Hypoxia and HABs due to eutrophication are increasing in frequency and magnitude in both fresh and marine waters (e.g., Anderson et al., 2002, Heisler et al. 2008; Glibert et al. 2005, 2006, 2014; Glibert and Burkholder 2018). Compared to the 12 million MT of N fertilizer used in the US, it is estimated that 1.15 million MT (or about 10%) of N flows into the Gulf of Mexico annually (von Reusner 2019) contributing to the hypoxia there. The Gulf of Mexico is a prime example of how eutrophication problems can be spatially and temporally displaced from the original nutrient source (Conley et al. 2009; Paerl 2009; Glibert et al. 2011; Glibert 2020). Aside from the nuisance they cause, HAB toxins contaminate drinking water supplies, as was the case in Toledo in 2014 when 500,000 residents were told not to use tap water due to microcystin contamination (e.g., Fitzsimmons 2014), and in coastal waters, HABs also contaminate seafood supplies, cause fish kills, and, depending on species, respiratory distress among many other human and ecosystem health effects (e.g., Landsberg 2002; Backer and McGillicuddy 2006; Basti et al. 2018, Gratton et al. 2018 and references therein).

Control of P has been long been promoted to curtail freshwater HABs because it is easier to control than N, and has long been considered the limiting nutrient in freshwaters (e.g., Schindler et al. 2008, 2016, Schindler and Hecky 2008). It has also been long been thought that if N is reduced well below balanced proportions, there can be growth of N_2-fixing cyanobacteria among which are toxic species and they will compensate for N limitation by accessing the atmospheric source (e.g., Schindler et al. 2008, 2016 and references therein). Thus, it would seem that the trend in increasing N:P should be viewed positively. However, the trend of increasing N:P proportions in fertilizer inputs is particularly concerning for several reasons. Many HAB cyanobacteria are not N_2-fixing, for example, *Microcystis*, and their occurrences are increasing in freshwaters around the world in direct proportion to increasing N loads (Glibert et al. 2014 and references therein). *Microcystis* is increasing throughout the US, but the Midwest is a hot spot for blooms—and for more toxic blooms—due to agricultural impacts (Fig. 18c; Michelak et al. 2013; Loftin et al. 2016). Many marine and estuarine dinoflagellate HABs also have been shown to be more abundant under conditions of increasing N:P. Examples of high biomass HAB dinoflagellates occurrences in environments where N:P loads are in excess of Redfield proportions can be found in the Baltic Sea (Hajdu et al. 2005), Delaware Inland Bay (Handy et al. 2008), Neuse River Estuary (Springer et al. 2005), Chesapeake Bay (Li et al. 2015) and East China Sea (Li et al. 2009; Glibert et al. 2014) among many other regions.

The second problem with a focus on P control over N control is that many cyanobacteria and marine or estuarine dinoflagellate HABs (among other HAB taxa) may be, in fact, more toxic when N is in stoichiometric excess over P. Thus, contrary to the concern that N limitation will promote toxic cyanobacteria, the toxicity of many HABs increases as N:P increases (Glibert 2017 and references therein). Most notably, excess N over P availability has been related to microcystin production under controlled chemostat conditions and in natural populations (Oh et al. 2000; Van de Waal et al., 2009; Harris et al. 2016). In the dinoflagellate *Alexandrium tamarense,* saxitoxin production increased by three- to fourfold under P deficiency (Boyer et al. 1987; Guisande et al. 2002, reviewed by Granéli 2005; Granéli and Flynn 2006), and toxicity of the dinoflagellate *Karlodinium* *veneficum* increased under P limitation, but especially in combination with elevated levels of CO_2 (Fu et al. 2010). Similarly, toxin production by the dinoflagellates *Gymnodinium catenatum, Alexandrium excavatum* and the diatom *Pseudo-nitzschia multiseries* also increased under P stress (Granéli and Flynn, 2006). Many toxins are rich in N and accordingly N-rich toxins can accumulate in excess under P limitation (e.g., Granéli and Flynn 2006; Van der Waal et al. 2014 and references therein).

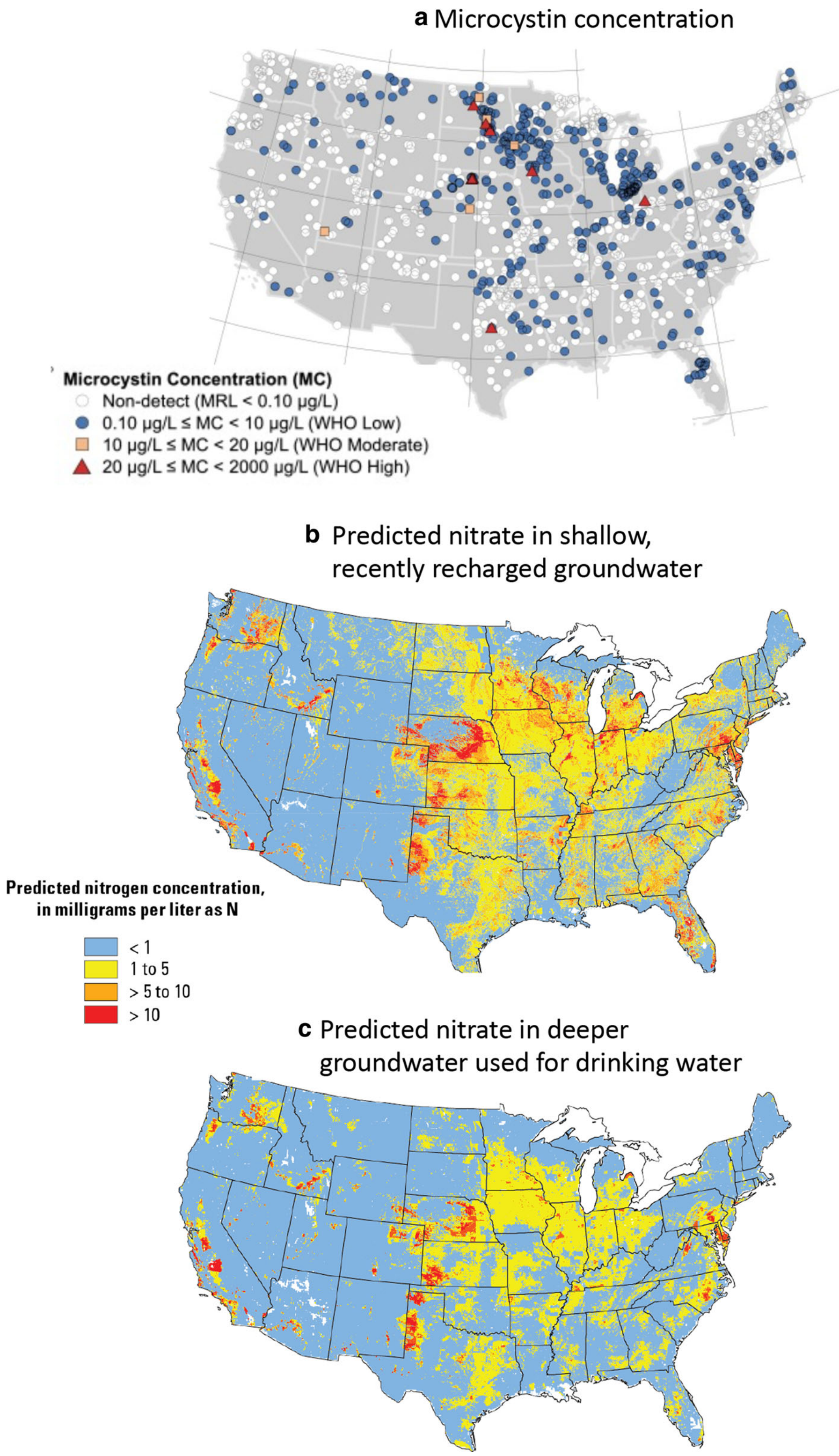
Adding to the trends of increasing N relative to P are the atmospheric NH_3 emissions from animal feeding operations. Most such emissions are deposited within 2.5 km of the source, based on studies of emission from broiler houses on the US eastern seaboard (Baker et al. 2020). These emissions, derived from the animal houses themselves, manure handling, or land applications, have multiple environmental effects. Not only do these emissions contribute to eutrophication (e.g., Mallin and Cahoon 2003; Galloway et al. 2014), but they can form fine aerosols as NH_3 is converted to NH_4 and deposited on particles, contributing to haze, impaired visibility and respiratory problems. These aerosols can also be deposited as NH_3/NH_4 on nearby forests or crops which can, in turn, elicit stress responses from acute NH_3/NH_4 exposure (e.g., Fangmeier et al. 1994). Recent modeling has shown that there has been a threefold increase in soluble N deposition over land and a twofold increase over the ocean due to human activities (Kanakidou et al. 2016), driven largely by NH_3 emissions from agriculture that have traveled from the original source.

These trends all underscore that nutrient reduction efforts must focus on both N and P, even for regional systems that are classically considered to be “limited” by one nutrient or the other (e.g., Burkholder et al. 2006; Howarth and Paerl 2008; Conley et al. 2009; Paerl 2009; Glibert et al. 2011, 2013; Glibert 2017, 2020). Dual nutrient strategies, however difficult they are to achieve, should be the goal, as multiple ecological and ecoservice benefits are met by reducing N input (Vitousek et al. 1997) even in classically P-limited systems, such as lakes. Fragmenting sustainability arguments and focusing on single nutrient reduction measures undermines the need to protect multiple ecosystem services at broad spatial scales, especially given that many eutrophication problems are displaced from the original nutrient source, as previously described for the Gulf of Mexico.

To promote a more environmentally-favorable image, the fertilizer industry has been advocating that farmers apply the “4R” rule for fertilizers: the right source at the right rate, right time and right place (https://www.nutrientstewardship.com/4rs/). This same right-place-right-time principle applies to the kinds of algal species that respond in receiving waters of these wastes. It takes the right nutrients at the right time relative to the needs of the primary producers (algae) for blooms to form (Glibert and Burford 2017). While over-enrichment of both fresh and coastal waters by nutrients is a major pollution problem worldwide, it is not only total nutrient loads that promote HABs and alter microbial biodiversity, it is the right nutrients at the right time.

Many HAB taxa also appear to be favored over diatoms when N is supplied in chemically-reduced relative to oxidized forms—as, for example, in the form of urea (Glibert et al. 2006, 2014). The shift toward increasing use of urea stems from several advantages it has over other N forms (Glibert et al. 2006). It is less explosive than NH_4 and NO_3 when stored, and it can be applied as a liquid or solid. The increase in global use of urea has been related to HAB increases (Glibert et al. 2006, 2014, 2016), and similar conclusions can be drawn for various parts of the US where urea use has increased. For example, cyanobacterial blooms in Florida Bay and on the southwest Florida shelf have been shown to be positively correlated with the fraction of N taken up as urea, and negatively correlated with the fraction of N taken up as NO_3^- (Glibert et al. 2004). Use of slow-release fertilizers has been promoted to reduce leaching of N; slow release fertilizers are coated urea-based granules that may contain a urease inhibitor. The use of urease inhibitors delays the hydrolysis of urea for up to several weeks and thus increases the likelihood that runoff or overland transport will contain urea and not its decomposition products (Prakash et al. 1999). Use of slow-release fertilizers may help to reduce hydrolysis in the soil, but may contribute to runoff of forms of N that are more favorable for at least some HABs.

Recently another environmental consequence of algal blooms has been reported: that is, blooms are an important contributor to CH_4 emissions (Beaulieu et al. 2019 and references therein; Fig. 1b). Production of CH_4 in lakes and eutrophic impoundments is


a Microcystin concentration
Microcystin Concentration (MC)
Non-detect (MRL < 0.10 μg/L)
0.10 μg/L ≤ MC < 10 μg/L (WHO Low)
10 μg/L ≤ MC < 20 μg/L (WHO Moderate)
20 μg/L ≤ MC < 2000 μg/L (WHO High)
b Predicted nitrate in shallow, recently recharged groundwater
Predicted nitrogen concentration, in milligrams per liter as N
< 1
1 to 5
> 5 to 10
> 10
c Predicted nitrate in deeper groundwater used for drinking water

◀ **Fig. 18** Maps of **a** concentrations of microcystins in US lakes, **b** predicted NO_3 in shallow, recently recharged groundwater, and c that of deeper groundwater used for drinking water. Panel a reproduced from Loftin et al. (2016) with permission from Elsevier. Panels **b**, **c** reproduced from USGS (https://www.usgs.gov/media/images/predicted-concentrations-nitrate-us-groundwater; public domain)

directly related to the chlorophyll *a* concentration of the water (DelSontro et al. 2018). Beaulieu et al. (2019) estimate that CH_4 emissions from eutrophic lakes will increase 30–90% over the next century due to continuing eutrophication pressures. Moreover, these authors reported that an increase in P loading by 1.5 times will lead to CH_4 emissions that are equivalent to that from wetlands, currently the largest single source. The continued nutrient pollution from crop and animal production clearly multiplies the impact on greenhouse gases due to accumulations of algal biomass and its decay. It is now abundantly clear that the historic view of algal responses to eutrophication—i.e., that increased nutrients promote increased chlorophyll and high-biomass blooms leading to oxygen deduction and losses in habitat (e.g. Cloern 2001)—is far too simplistic for understanding how harmful taxa develop in response to changes in nutrients.

Human health and community impacts

Numerous studies have documented the many human health impacts of populations living in the shadow of large animal operations. Casey et al. (2015) reviewed the literature of the past 2 decades and reported that four types of health problems were consistent related to life near CAFOs: respiratory issues, methicillin-resistant *Staphylococcus aureus* (MIRS), Q fever (caused by the bacteria *Coxiella burnetii* typically transmitted from animals), and mental health (stress). Occupational-related asthma and bronchitis is not unusual among farm workers or family members, nor is exposure to dangerously high concentrations of NO_3 in drinking water, especially given the fact that many rural areas draw water from local wells rather than municipal supplies (reviewed by Burkholder et al. 2007; Miller and Muren 2019; Fig. 18b,c). High concentrations of NO_3 in water supplies have been associated with increased risks of blue baby syndrome, some cancers (including colon, kidney, stomach, ovarian and bladder), reproductive effects, and diabetes (reviewed by Burkholder et al. 2007; Casey et al. 2015; Miller and Muren 2019). Other contaminants in water near CAFO-impacted communities include veterinary antibiotics or hormones, pesticides, and other pharmaceuticals seep into surface and groundwater from applications to sprayfields or leak from poorly constructed or aging lagoons (Burkholder et al. 2007 and references therein).

Emissions of NH_3 from CAFOs can trigger asthma attacks. Often emissions of H_2S co-occur with NH_3 emissions, especially from poultry houses. It has been reported that people frequently exposed to these emissions were 66% more likely to be diagnosed with pneumonia (Poulsen et al. 2018).

Substantial amounts of fecal bacteria remain in manure when this material is spread on land. While many such microbes may be killed by exposure to ultraviolet radiation (Crane et al. 1983), many remain viable. Viability can be maintained when these materials enter groundwater or surface waters (Mallin and Cahoon 2003). Burkholder et al. (1997) observed that fecal bacteria could be found in river waters and sediments months after a large swine waste spill, but even without large spills, chronic exposure can be problematic.

Economics and trade-offs

Ewing and Runck (2015) modeled the trade-off between the need to optimize high rates of N fertilization of corn and the cost of water quality impacts in the Midwestern US—and highlighted the "deep conflict" between stakeholders involved in food production and those using water resources. Their analysis underscored the importance of understanding regional (less than county level) variabilities where optimizations can be gained and emphasized the importance of stakeholder involvement at local scales. They showed that technological solutions do exist that could increase corn production and improve water quality. Yet, Herrero et al. (2015) argue that even with the efforts over the past decade to quantify impacts of the "gargantuan appetite for livestock products", integrating these efforts with economic and sociocultural efforts is seldom done, when climate, nutrient cycles, biodiversity, land degradation, deforestation are collectively considered.

Costs to reduce and mitigate nutrient pollution are extremely high. A recent estimate from USDA (cited in Ribaudo et al. 2011) suggests that $2 billion annually is spent removing NO_3 that originates with cropland applications and that two-thirds of US cropland is not meeting criteria for good N management. Sobota et al. (2015) estimated the economic costs associated with the leakage of N from the production of food, fuel and fiber in the US. They calculated the damage cost in mitigation, remediation, direct damage or substitution for each N source (focusing on synthetic fertilizers) and human health and environmental impacts by applying methodology described by Birch et al. (2011, Compton et al. (2011) and van Grinsven et al. (2013). They estimated that in the year 2000, the damage costs for N leakage ranged from $1.94 to $2,255 ha^{-1} for different hydrological zones as defined by the USGS. Of these damages, 73–77% were associated with leakage of agricultural N, and areas with the largest damage to aquatic habitat and eutrophication were in the upper Midwest and central California (Fig. 19). Interestingly, they also calculated that much of mid Atlantic, Pacific Northwest, as well as southern California, received less N annually than the Midwest yet had similar damage costs because of the high costs of air pollution on human health. Across the nation, they estimated that damages ranged from $19 billion associated with drinking water impacts to $78 billion associated with freshwater ecosystems, and overall the median estimates in all damages was $210 billion in the early 2000s. This figure represent 21% of the estimated $992 billion that the food and agriculture industry contributes to the US economy (as of 2015; https://www.agweb.com/article/food–ag-industry-contributes-992-billion-to-us-economy-NAA-ben-potter). NOAA published a similar finding, estimating that $82 billion was lost each year in lost fishing revenues and human health problems associated with algal blooms (https://aamboceanservice.blob.core.windows.net/oceanservice-prod/ecoforecasting/noaa-ecoforecasting.pdf). Yet, in keeping with Herrero et al.'s (2015) central point that the economic and societal costs of livestock production must be better understood, undoubtedly, the economic impacts estimated by Sobota et al. (2015) would be higher today and would be higher if the damage from leaked N from the increasing number of animal operations were also considered. A very recent report estimates that the total hidden costs of the food industry across the world to be in range of $12 trillion yr^{-1}, accounting for water scarcity resulting from agriculture use, biodiversity loss and greenhouse gas emissions–a value approaching the domestic product of China (Nature 2019).

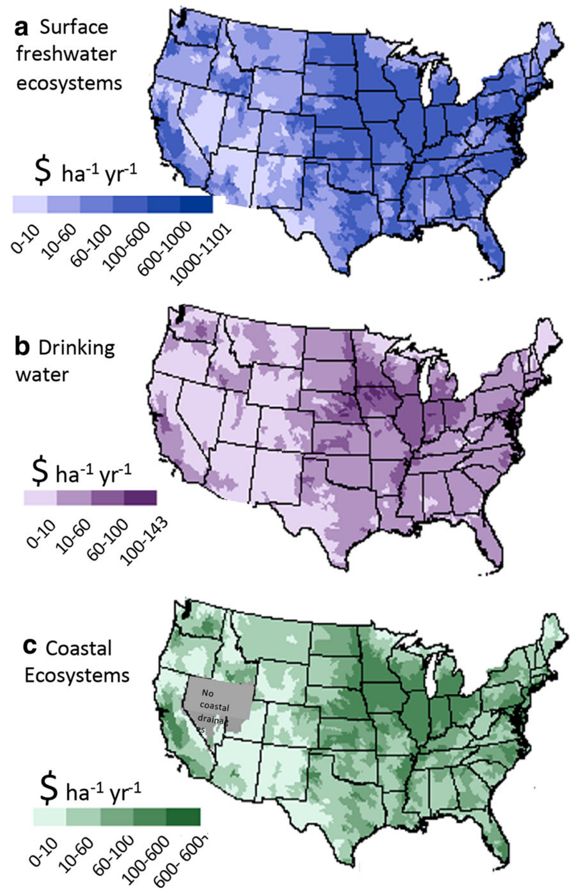


Fig. 19 Estimated costs of N pollution in the US on **a** surface freshwater ecosystems, **b** drinking water, and **c** coastal ecosystems. Reproduced from Sobota et al. (2015) under Creative Commons 3.0 license

Threats to current and future farming

Farming practices will evolve, whether or not such changes will be driven by sound policies, or factors beyond regulatory control. The consolidated, and seemingly highly efficient, food production system is not resistant to change. Its fragility, in both the short term and long term is evident.

The current tariffs on soybean and pork imposed by the Chinese on US exports clearly affect production in the short term. Farmers are being squeezed by these policies in many different ways. Many farmers are going bankrupt. On average, 7 dairy farms went bankrupt per day in 2018 (https://www.farmpolicyfacts.org/2019/04/our-view-trade-can-kickstart-ailing-farm-economy/). Bankruptcies have increased in 9 of the 10 regions of the country; in Regions IV, V, VI and VII, these numbers totaled 81,125, 62 and 87 in 2018 (Wilton and Newton 2019). These bankruptcies are mostly those of the remaining small farmers; large corporations have more capital to buffer these downturns. Subsidies have aided farmers especially in the upper Midwest (Regions VII and V), but have disproportionately aided the industrial farming conglomerates. Consolidation of large farms will only increase.

At the time of this writing, there has been a rapid acceleration in the rate of burning of the Amazon rainforest (Sullivan 2019; Ortiz 2019). The number of fires is about 35% higher than in the first half of the year for all years since 2010, and has risen 79% since 2018 (Ortiz 2019). These fires have been largely deliberately set to clear forests for the planting of soybeans and for cattle grazing. If the US is no longer the world's breadbasket, other countries will take this role. Brazil has currently replaced the US as the major provider of soybeans for China, and as soybean production in Brazil has ratcheted upwards, it is becoming well positioned to become the world's leading supplier (Sullivan 2019). Thus, Brazil burns to create new farmland from the Amazon as small US farmers struggle, both in response to changes in US-China trade relationships, with large international, industrial farms able to capitalize on both of these changes.

One recent factor that industrial farms have not been able to control is the impact of the global 2020 coronavirus pandemic. Many US meat packing plants closed for periods of time due to employee illness. Consequently, many hogs and broilers were euthanized, placing more economic hardships on US producers. These carcasses are being disposed in landfills or composted for fertilizer (Pitt 2020). The full impacts of trade tariffs, the pandemic and other short-term pressures are yet to be seen, and future inventories at local and regional levels will tell those stories.

In the longer term, it is projected that P reserves may be exhausted in a few decades (e.g., Daneshgar et al. 2018). The demand for N, however, is estimated to continue to escalate. For North America, the rate of N use may increase by 32% and that of P use by 24% relative to 2005, based on estimates of Drescher et al. (2011). Globally, urea use is projected to double by mid-century (Millenium Ecosystem Assessment 2005, Glibert et al. 2014 and references therein). This will continue to drive the N:P of runoff higher, with environmental consequences downstream.

The United Nations recently released a report on climate change (IPCC 2019) which details how interactions between climate change, greenhouse gas fluxes, extreme events (floods and droughts), land use change, and desertification may threaten food and nutritional security. Temperatures and CO_2 are rising–factors that may seem beneficial for the growth of some crops. Favorable regions for certain crops may migrate. There is some evidence that higher temperatures are favoring corn production in Minnesota, but disfavoring yields in Illinois, Indiana, and Ohio, and also favoring soybean production in the upper Midwest while disfavoring wheat (Belz 2019). Extreme heat can also alter the timing or rate of flowering, in some cases rendering plants sterile (Dukes and Hertel 2018). Disease and pests may change in frequency. Increased temperatures also reduce the feeding rate by animals and increase their susceptibility to disease.

Under changing climate, precipitation is less predictable, often coming in fewer, more concentrations events. High rainfall makes planting difficult, flooding late in the season can drown plants, but too little rainfall also kills plants (Dukes and Hertel 2018). The extent to which changing precipitation patterns will affect farm production in the long run is yet to be determined. The Midwest experienced massive flooding in 2019, leading to the inability of many farmers to even sow their crops. The 2018–2019 planting season was the wettest in recent history, and the past 5 years have also experienced very wet April–May periods (https://mrcc.illinois.edu/pubs/docs/GL-2018_Climate-trends-and-impacts-summary.pdf). Accordingly, fields were left unplanted, and while this led to higher prices for corn and soybean due to reduced supply, the lack of crop to sell does not balance this loss for farmers. This flooding follows the devastating Midwest drought of 2012. As a crop highly sensitive to heat and water stress, corn is definitely at risk for future and will see more market volatility in the years to come. Recent modeling suggests that in the Midwest, water balance changes due to increased temperature and reduced snowfall may be more important than increased precipitation in the next half decade (Kalcic et al. 2019).

One approach farmers have used to overcome this risk is to forsake traditional crop rotation (corn and soybean) for continuous corn production. In 2012, 22% of corn production was in continuous rotation, a practice that leads to more fertilizer use as well as more soil erosion (Barton and Clark 2014). Moreover, some corn hybrids are becoming more sensitive to drought, requiring higher rates of irrigation during drought periods (Barton and Clark 2014).

Intensive precipitation also leads to greater runoff of both fertilizer and of soil itself. Yet, precipitation events may affect N and P differently. On the one hand, P, which is often bound to particles can be more easily transported by overland flow, whereas N, especially as NO_3, more readily leaches into the ground and may or may not be mobilized to adjacent waters (e.g., Sims et al. 1998). In situ time series of nutrient monitoring in tributaries of the Chesapeake Bay confirmed these different patterns following rainfall events (Glibert et al. 2005). On the other hand, the accumulation of P in soils over time contributes to retention of P relative to N, and a further skewing of the N:P ratio in exported nutrients (Beusen et al. 2016; Bouwman et al. 2017).

Climate changes also pose other risks. There is now considerable emerging evidence that in a higher CO_2 environment, the nutritional quality of plants, including the cellular content of N, protein, and vitamins, is reduced, especially for those plants having C3 metabolism (e.g., rice, wheat) (Loladze 2014; Weigel 2014). This, in turn, may alter the food quality for the animals that are dependent on those crops and may contribute to negative shifts in human nutrition as well. Large, industrialized operations are far less nimble in their ability to adapt to change than smaller operations.

Opportunities and impediments for advancement

Numerous scientists have suggested approaches that can be undertaken globally to mitigate nutrient pollution (e.g., Sutton et al. 2013; Conant et al. 2013; Billen et al. 2015; Bouwman et al. 2017). In the US, legislative efforts related to nutrient pollution from farms are not advancing in the right direction. The Farm Bills of recent years have cut the conservation provisions considerably which were originally included in the 1985 Farm Bill. Moreover, funds available through the Environmental Quality Incentive Program in the 2002 Farm Bill, meant to incentivize farmers to idle lands and to implement other environmental improvements, were allowed to be used for the construction of manure lagoons (Imhoff 2019). Further degradation of waters may result from the current administration's efforts to roll back the definition of "waters of the United States" under the Clean Water Act, thus releasing regulations on many wetlands and tributaries that were protected since 1986 and which was broadly enforced by the EPA since 2015 (Eilperin and Dennis 2019). Wetlands and tributaries are often first recipients of farm runoff.

It is unlikely that the economic and policy drivers favoring large agricultural systems will change any time soon. Much has been written about best management practices, fertilizer use efficiency and potentials for improvement (e.g., Bouwman et al. 2009; Fixen et al. 2015; Mueller et al. 2017; Zhang 2017; Clark and Tilman 2017; Alexander et al. 2017). Davis et al. (2015) modeled the global impacts of livestock intensification, and specifically the shift to dependence on grain. They found that animal calories produced from feed were more efficient than those produced from non-feed sources in terms of land use and greenhouse gas emissions, but conversely production from feed required substantially more N per animal in the overall production chain. Livestock fed poorer quality feed produce more CH_4 than those fed forages that are more nutritious (https://extention2.missouri.edu/g310). Others have suggested other approaches that can be taken to reduce nutrient pollution, such as reduction of food waste and improved processes for mitigating or removing N pollution from the environment (e.g., Houlton et al. 2019 and references therein). While major improvements in use efficiency can be implemented in parts of the world where fertilizer use is less fine-tuned to specific crops and soil types, it is unlikely to ever reach a point where there is no environmental loss. The difficulty in improving efficiency of N use particularly lies in the high mobility of N in the soil–plant system, and the variety of potential loss pathways, ranging from volatilization of NH_4^+, denitrification, leaching and runoff and other pathways (Bouwman et al. 2009). While both P and N have been accumulating in soils (e.g., Van Meter 2016, 2017, Zhang et al. 2017), leading to opportunities for fertilizer reductions, sales of N relative to P fertilizer continue to rise.

Manure management varies by animal operation and by state and there has been a shift toward liquid waste management in both the dairy and swine industries. Anaerobic lagoons and liquid slurry operations (Online Resources Table S3) are most common in dairy and hog operations (e.g., Hunt et al. 2019, Niles and Wiltshire 2019 and references therein; Fig. 20). Managing liquid manures appears to be among the "lowest hanging fruit" of nutrient control in much of the country. Manure spreading should be held to the same strict "4Rs" accounting as chemical fertilizer applications. The lagoons themselves need to be carefully managed. Lagoons, which may be clay or plastic lined, may lose integrity with age (Barth et al. 2004), leading to increased leakage. Many older lagoons are unlined. Volatilization also depends on how farmers manage their lagoons with respect to C content; NH_3 emissions can be reduced if C-rich bedding material is used (Barth et al. 2004). Emissions vary with the bacterial content of the lagoons, especially purple sulfate bacteria (Leytem et al. 2017). Emissions also increase with temperature and pH of the holding lagoon (Arogo et al. 2003; Harper et al. 2004; Doorn et al. 2002; Leytem et al. 2017; Peterson 2018 and references therein). Emissions are also highly variable with short-term wind and precipitation events, with increases in CH_4 emissions from dairy lagoons during rainy days (Grant and Boehm 2015; Leytem et al. 2017). Covers may help to limit these emissions. There are some efforts to use pig manure and corn silage for biogas production (e.g., Gaworski et al. 2017). This technology is beginning to be applied in North Carolina, where Smithfield Foods, now a Chinese company, has partnered with Duke Energy (e.g., Coker 2018). Ultimately, waste treatment may become the only mechanism by which real nutrient reductions can occur. If water quality is valued and if the costs of algal blooms and other environmental impacts are fully recognized, wastewater treatment for animal operations may eventually become economically sound.

Some practices or policies appear to provide favorable environmental outcomes but there can also be unintended consequences. Organic farming, for example, may reduce use of some chemical fertilizers, but this reduction in fertilizer use creates another problem: yields are lower, by as much as 8–25% (Baldos 2018). Therefore, organic farmers have to convert more lands to agricultural fields to produce the same quantity. Moreover, organic nutrients, which favor the growth of many types of HABs, are used to a greater extent in organic farming, leading to increased leakage of these forms to local waters. Weed control on organic farms also requires more mechanical cultivators, leading to more soil erosion and other associated secondary problems (Gunderson et al. 2018). As another example, some animal operations are moving to cage-free operations, particularly in the poultry industry where there is pressure for more humanely-raised products. Many restaurants, including McDonald's, are committed to using eggs only from cage-free systems. Yet, these systems lead to higher NH_3 emissions and other air quality problems due to the greater accumulation of manure and scratching that the birds do while exposed to this dust and litter (Erickson 2018). These changes could have large regional impacts, as chicken producers in the mid-Atlantic (Maryland and Delaware) currently contribute about 17% of the N load of Chesapeake Bay (https://cen.acs.org/environment/pollution/Livestock-emissions-still-air/96/i14). There are no simple solutions that will unravel the profitability and environmental impacts from large agrobusinesses–especially in the current US policy climate.

By definition, CAFO lagoons are "point sources" of pollution and, depending on the size of operation and waste handling procedures, must be permitted under the Clean Water Act, which requires operators to have a nutrient management plan and which defines the limits on the allowable amount of discharge to local waters. Such regulations have been regularly revised (US EPA 2010) and regularly challenged in court. As noted above, state-wide reporting–and therefore the transparency of state-wide statistics–of CAFOs is low for almost every state (Miller and Muren 2019). Permitting can be avoided if the size of the operation falls just under the regulatory limit, and the percentage of CAFOs reporting permits to the EPA (https://www.epa.gov/sites/production/files/2019-09/documents/cafo_tracksum_endyear_2018.pdf) is astonishingly low, especially for those states where hog production is high (Fig. 11b; Online Resources Fig. S6c). Permitting can also be avoided if the facility does not discharge directly to a waterway. Lack of permitting does not imply illegal operation, only that the configuration (i.e., number of confined animals or waste management procedures) of the farm differs from that required to be regulated. The animals from

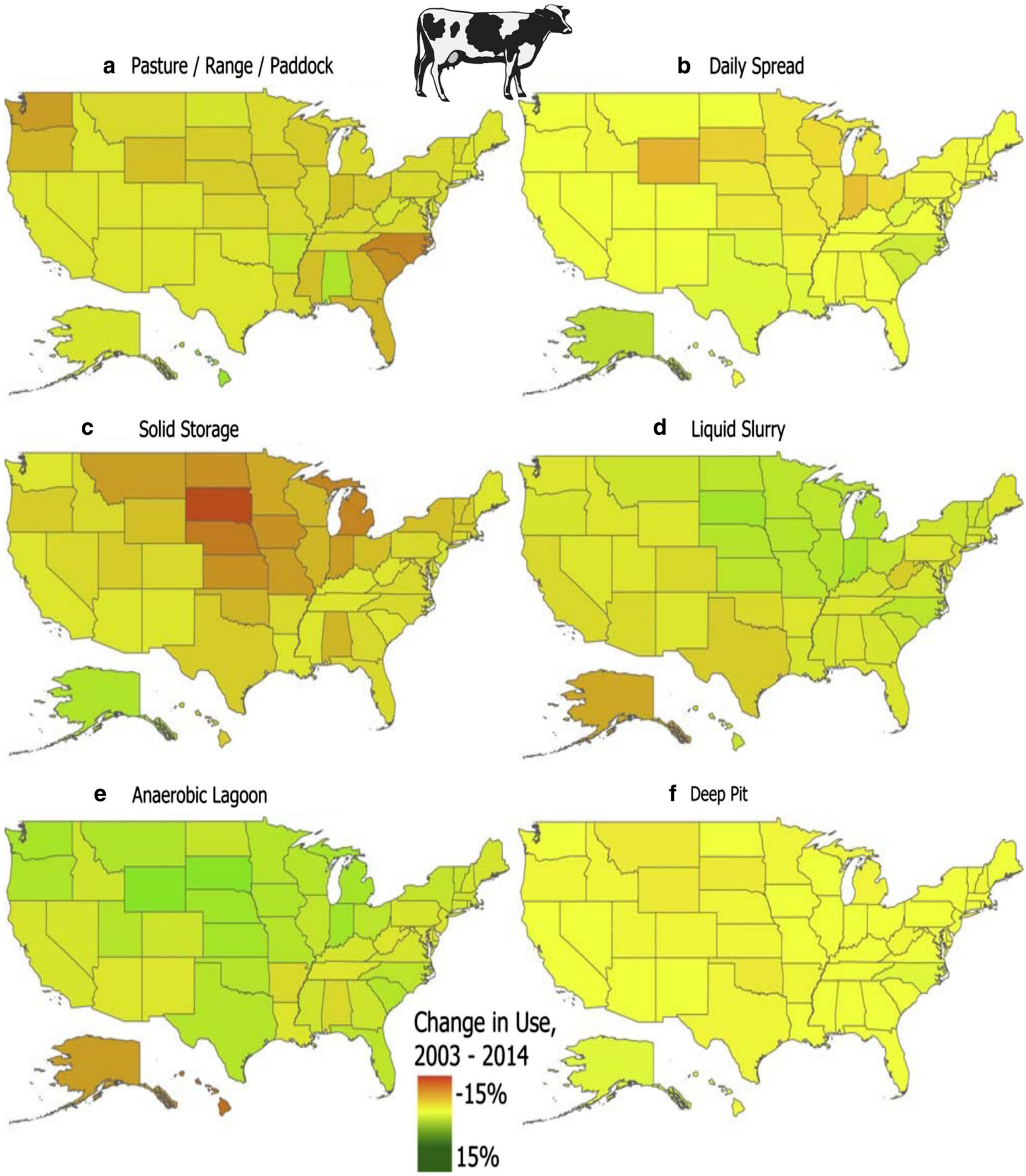


Fig. 20 Change in different waste management strategies of dairy operations in the US from 2003 to 2014 Reproduced and modified from Niels and Wiltshire (2019) under Creative Commons 3.0 license

unpermitted operations nevertheless still release nutrients. Moreover, federal inspections and enforcement of CAFOs have declined every year since 2011; in 2016, enforcement actions were down 75% and inspections down more than 50% compared to those actions taken during the Obama administration (Walton 2016).

There are no federal policies as of yet regarding the emissions of CH_4 or N_2O from CAFO operations (Tomas 2019), nor is this a politically favorable time to suggest new policies or regulations. Because farmers and ranchers are exempted from reporting emissions to federal agencies, the US EPA methodology for estimating emissions is under continual evolution. This exemption from reporting was reaffirmed in the recent Farm Act (Erickson 2018). As seen from the permitting percentages, most farming waste disposal does not fall under the Clean Water Act, but it has been suggested that as emitters of greenhouse gases, farm operations, and especially CAFOs, could, however, fall under some previsions of the Clean Air Act (Tomas 2019). Others (e.g., Ruhl 2000) have argued that the "geographic, economic, and political settings of the farming industry call for approaches that may be outside the box of conventional environmental law. The environmental regulation of farms must incorporate several key features if it is to succeed where traditional models of environmental law surely would not". Such an approach would balance environmental regulation with tax incentives and trading programs. As noted above, it is unlikely that such a sweeping new approach to environmental regulation of farming will happen any time soon.

Conclusions

This paper has attempted a broad review of the patterns and trends in nutrient inputs and greenhouse gas pollution arising from US farming practices. This analysis builds on publicly available and published data and makes use of available detailed inventories. Collectively these efforts have shown that for the entire US: (1) use of N fertilizer is increasing faster than that of P, leading to an increase in the N:P of this source; (2) fertilizer N inputs exceed those of manure, while fertilizer P inputs and those of manure are more comparable; (3) the number of CAFOs has increased over the past decades, including a near 10% increase since 2012, driven largely by a 13% increase in hog production; (4) atmospheric NH_3 release and human wastewater total inputs are less than those of fertilizer and manure, but large regional differences exist across the country (and atmospheric NH_3 may be underestimated); (5) while CH_4 emissions from enteric fermentation remain the largest contributor of this greenhouse gas pollutant, CH_4 and N_2O emissions from manure management are rapidly rising.

At the broad scale, the industrialization of farming, driven by economics rather than a sustainability ethic, will only continue to exacerbate the eutrophication of fresh and coastal waters. There has been an upward trend in N:P of all inputs, conditions that favor many HABs and/or their toxicity. Tariffs and trade disputes are contributing to the destruction of the Amazon as Brazil steps in to lead global soybean production. Together with climate threats and uncertain political trade policies, a near-term future with reductions in nutrient and greenhouse gas emissions by the US farming industry is bleak, and the negative consequences will be felt worldwide for the foreseeable future.

Acknowledgements PMG wishes to thank the editor, Dr. Kate Lajtha, for the invitation to prepare this review, and Lex Bouwman and Arthur Beusen for discussions in the early stages of this analysis. Comments by reviewers were extremely helpful and greatly appreciated. PMG was partially supported by the Ecosystem Science Program of NSF, award number DEB 1911349. This is contribution number 5885 from the University of Maryland Center for Environmental Science.

Funding PMG was partially supported by the Ecosystem Science Program of NSF, award number DEB 1911349.

Data availability All data are from publicly available sources as indicted throughout the manuscript.

Compliance with ethical standards

Conflicts of interest The authors declare that they have no conflict of interest.

References

Alexander P, Brown C, Arneth A, Finnegan J, Moran D, Rounsevell MDA (2017) Losses, inefficiencies and waste in the global food system. Agricul Syst 153:190–200.

Alexander R, Smith R, Schwarz G, Boyer E, Nolan J, Brakebill J (2008) Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. Environ Sci Technol 42:822–830

Anderson DA, Burkholder JM, Cochlan W, Glibert PM, Gobler C, Heil C, Kudela R, Parsons T, Trainer V, Vargo G (2008) Harmful algal blooms in the United States: linkages to eutrophication. Harmful Algae 8:39–53

Anderson DA, Glibert PM, Burkholder JM (2002) Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25:562–584. https://doi.org/10.1007/BF02804901

Arogo J, Westerman PW, Heber AJ (2003) A review of ammonia emissions from confined swine feeding operations. Trans Am Soc Agricul Eng 46:805–817

Backer LC, Kirkpatrick B, Fleming L, Cheng Y-S, Pierce R, Bean J, Clark R, Johnson D, Wanner A, Tamer R, Baden D (2005) Occupational exposure to aerosolized brevetoxins during Florida Red Tide events: impacts on a healthy worker population. Environ Health Perspec 113–115:644–649

Backer LC, McGillicuddy DJ (2006) Harmful algal blooms at the interface between oceanography and human health. Oceanography 19:94–106

Baker J, Battye WH, Robarge W, Arya SP, Aneja VP (2020) Modeling and measuring of ammonia from poultry operations: their emissions, transport, and deposition in Chesapeake Bay. Sci Tot Environ 706:135290. https://doi.org/10.1016/j.scitotenvir.2019.135290

Baldos U (2018) The technology ticket. In: Eise J, Foster K (eds) How to feed the world. Island Press, Washington, DC, pp 77–93

Barth E et al (2004) Risk management evaluation for concentrated animal feeding operations. In: Haines J, Staley L (eds) EPA. Office of Research and Development National Risk Management Research Laboratory, Cincinnati, OH

Barton B, Clark SE (2014) Water & climate risks for U.S. corn production, how companies & investors can cultivate sustainability. Ceres. https://www.ourenergypolicy.org/wp-content/uploads/2014/06/ceres-corn.pdf.

Basti L, Hégaret H, Shumway SE (2018) Harmful algal blooms and shellfish. In: Shumway SE, Burkholder JM, Morton SL (eds) Harmful algal blooms: a compendium desk reference. Wiley, Singapore, pp 135–190

Beaulieu J, DelSontro T, Downing JA (2019) Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. Nat Commun 10:1375. https://doi.org/10.1038/s41167-019-09100-5

Belz A (2019) Climate change surprise: it is helping to grow more corn and soybean in the upper Midwest. Idaho Stateman. July 12, 2019. http:www.idahostateman.com/news/business/article232585007.html.

Berendes DM, Yang PJ, Lai A, Hu D, Brown J (2018) Estimation of global recoverable human and animal faecal biomass. Nat Sustain 1:679–685

Beusen AHW, Van Beek LPH, Bouwman AF, Mogolion JM, Middelburg JJ (2015) Coupling global models for hydrology and nutrient loading to simulate nitrogen and phosphorus retention in surface water– description of IMAGE–GNM and analysis of performance. Geosci Model Dev 8:4045–4067. https://doi.org/10.5194/gmd-8-4045-2015

Beusen AHW, Bouwman AF, Van Beek LPH, Mogolion JM, Middelburg JJ (2016) Global riverine N and P transport to ocean increased during the 20th century despite increased retention along the aquatic continuum. Biogeoscience 13:2441–2451

Beusen AHW, Bouwman AF, Heuberger PSC, Van Drecht G, Van Der Hoek KW (2008) Bottom-up uncertainty estimates of global ammonia emissions from global agricultural production systems. Atmos Environ 42:6067–6077. https://doi.org/10.1016/j.atmosenv.2008.03.044

Billen G, Garnier J, Lassaletta L (2013) The nitrogen cascade from agricultural soils to the sea: modelling nitrogen transfers at regional watershed and global scales. Philos Trans R Soc B 368(1621):20130123. https://doi.org/10.1038/nature11069

Billen G, Lassaletta L, Garnier J (2015) A vast range of opportunities for feeding the world in 2050: trade-off between diet, N contamination and international trade. Environ Res Lett 10:1–15

Birch MBL, Graming BM, Moomaw WR, Doering O, Reeling CJ (2011) Why metrics matter: evaluating policy choices for reactive nitrogen in the Chesapeake Bay watershed. Environ Sci Technol 45:168–174

Bittman M (2019) Iowa crops look like food – but no one's eating. Heated. July 11, 2019. https://heated.medium.com/iowa-crops-look-like-food-but-no-ones-eating-9360661c3664.

Bouwman AF, Beusen AHW, Billen G (2009) Human alteration of the global nitrogen and phosphorus soil balances for the period 1970–2050. Glob Biogeochem Cycle. https://doi.org/10.1029/2009GB003576

Bouwman AF, Beusen AHW, Lassaletta L, Van Apeldoorn D, van Grinsven H, Zhang J, van Ittersum MK (2017) Lessons from temporal and spatial patterns in global use of N and P fertilizer on cropland. Sci Rep 7:40366. https://doi.org/10.1038/srep40366

Bouwman AF, Goldewijk KK, Van der Hoek KW, Beusen AHW, Van Vuuren DP, Willems J, Rufino MC, Stehfest E (2013) Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. Proc Natl Acad USA 110:20882–20887

Bowen JL, Valiela I (2001) Histori changes in atmospheric nitrogen deposition to Cape Cod, Massachusetts, U.S.A. Atmos Environ 35:1039–1051

Boyer GL, Sullivan JJ, Andersen RJ, Harrison PJ, Taylor FJR (1987) Effects of nutrient limitation on toxic production and composition in the marine dinoflagellate *Protogonyaulax tamarensis*. Mar Biol 96:123–128

Bricker S, Longstaff B, Dennison W, Jones A, Boicourt K, Wicks C Woerner J (2007) *Effects of nutrient enrichment in the Nation's estuaries: a decade of change. National estuarine eutrophication assessment update*. NOAA Coastal Ocean Program Decision Analysis Series No.26,

National Centers for Coastal Ocean Science, Silver Spring, MD, 322 pp.
Buford T (2018) A hog waste agreement lacked teeth, and some North Carolinians say they're left to suffer, ProPublica Nov 23, 2018. www.propublica.org/article/a-hog=waste=agreement-lacked-teeth-and-some-north-carolinians-say-left-to-suffer.
Burkholder JM, Mallin MA, Glasgow HB Jr, Larsen LM, McIver MR et al (1997) Impacts to a coastal river and estuary from rupture of a swine waste holding lagoon. J Environ Qual 26:1451–1466
Burkholder JM, Dickey DA, Kinder C, Reed RE, Mallin MA, Melia G, McIver MR et al (2006) Comprehensive trend analysis of nutrients and related variables in a large eutrophic estuary: a decadal study of anthropogenic and climatic influences. Limnol Oceanogr 51:463–487
Burkholder JM, Libra B, Weyer P, Heathcote S, Kolpin D, Thorne PS, Wichman M (2007) Impacts of waste from concentrated animal feeding operations on water quality. Environ Health Perspect 115:308–312
Butler T, Vermeylen F, Lehmann CM, Likens GE, Puchalski M (2016) Increasing ammonia concentration trends in large regions of the USA derived from the NADP/AMoN network. Atmos Environ 146:132–140. https://doi.org/10.1016/j.atmosenv.2016.06.033
Cahoon LB, Mickucki JA, Mallin MA (1999) Nutrient imports to the Cape Fear and Neuse River basins to support animal production. Environ Sci Technol 33:410–415
Casey JA, Kim BF, Larsen J, Price LB, Nachman KE (2015) Industrial food animal production and community health. Curr Environ Health Rep 2:259–271
Clark M (2017) Tilman D (2017) Comparative analysis of environmental impacts of agricultural production systems, agricultural input efficiency, and food choice. Environ Res Lett 12:064016
Cloern JE (2001) Our evolving conceptual model of the coastal eutrophication problem. Mar Ecol Prog Ser 210:223–253
Coker C (2018) Swine manure to biomethane. BioCycle Magazine. 59 (5), p. 25. https://www.biocycle.net/2018/06/07/swine-manure-biomethane/.
Compton JE, Harrison JA, Dennis RL, Greaver T, Hill BH, Jordan SJ, Walker H, Campbell HV (2011) Ecosystem services altered by changes in reactive nitrogen: a new perspective for US decision making. Ecol Lett 14:805–815
Conant RT, Berdanier AB, Grace PR (2013) Patterns and trends in nitrogen use and nitrogen recovery efficiency in world agriculture. Glob Biogeochem Cycles 27:558–566. https://doi.org/10.1002/gbc.20053
Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C et al (2009) Controlling eutrophication: nitrogen and phosphorus. Science 323:1014–1015. https://doi.org/10.1126/science.1167755
Crane SR, Moore JA, Grismer ME, Miner JR (1983) Bacterial pollution from agriculture sources: a review. Trans Am Soc Ag Engin 26:858–872
Daneshgar S, Callegari A, Capodaglio AG, Vaccari D (2018) The potential phosphorus crisis: Resource conservation and possible escape technologies: a review. Resources 7(2):37. https://doi.org/10.3390/resources7020037
Davidson EA, Suddick EC, Rice CW, Prokopy LS (2015) More food, low pollution (MoFoLoPo): A grand challenge for the 21st century. J Env Qual 44:305–311. https://doi.org/10.2134/jeq2015.02.0078
Davis KF, Yu K, Herrero M, Havlik P, Carr JA, D'Odorico P (2015) Historical trade-offs of livestock's environmental impacts. Environ Res Lett 10:125013. https://doi.org/10.1088/1748-9326/10/12/12501
DelSontro T, Beaulieu JJ, Downing JA (2018) Greenhouse gas emissions from lakes and impoundments: upscaling in the face of global change. Limnol Oceanogr Lett 3:64–75
Doorn MRJ, Natschke DF, Thorneloe SA, Southerland J (2002) Development of an emission factor for ammonia emissions from US swine farms based on field tests and application of a mass balance method. Atmos Environ 36:36–37. https://doi.org/10.1016/S1352-23310(02)00689-1
Drescher A, Glaser R, Richert C, Nippes K-R (2011) Demand for key nutrients (NPK) in the year 2050. Draft report, University of Freiberg. https://esdac.jrc.ec.europa.eu/projects/NPK/Documents/Freiburg_Demand_for_key_nutrients_in_2050_Drescher.pdf.
Du H, Chen Z, Mao G, Chen L, Crittenden J, Yi R et al (2019) Evaluation of eutrophication in freshwater lakes: a new non-equilibrium statistical approach. Ecol Ind 102:686–692
DuBois TD, Gao A (2017) Bigmeat: Mega-farming in China's beef, sheep and dairy industries. The Asia-Pacific J: Japan Focus. https://www.globalreach.ca/big-meat-mega-farming-in-chins-beef-sheep-and-dairy-industries/5607796. Accessed 28 Aug 2017
Dukes J, Hertel TW (2018) Our changing climate. In: Eise J, Foster K (eds) How to feed the world. Island Press, Washington, DC, pp 59–76
Díaz RJ, Rosenberg R (2008) Spreading dead zones and consequences for marine ecosystems. Science 321:926–929
Eilperin J, Dennis B (2019) Administration finalizes repeal of 2015 water rule Trump called 'destructive and horrible'. The Washington Post. https://www.washingtonpost.com/climate-environment/administration-finalizes-repeal-of-2015-water-rule-trump-called-destructive-and-horrible/2019/09/11/fddfa49a-d4aa-11e9-9343-40db57cf6abd_story.html?wpisrc=al_news__alert-hse–alert-national&wpmk=1. 11 Sept. 2019
Elser JJ, Anderson T, Baron JS, Bergström A-K, Jansson M, Kyle M, Nydick KR, Steger L, Hessen DO (2009) Shifts in lake N: P stoichiometry and nutrient limitation driven by atmospheric nitrogen deposition. Science 326:835–837
Erickson BE (2018) Livestock emission still up in the air. Chemical and Engineering News April 2, 2018.
Erisman JW, Sutton MA, Galloway J, Klimont Z, Winiwarter W (2008) How a century of ammonia synthesis changed the world. Nat Geosci 1:636–639. https://doi.org/10.1038/ngeo325
Ewing PM, Runck BC (2015) Optimizing nitrogen rates in the Midwestern United States for maximum ecosystem value. Ecol Soc 20(1):18. https://doi.org/10.5751/ES-06767-200118
Fangmeier A, Hadwiger-Fangmeier A, Van der Eerden L, Jager HJ (1994) Effects of atmospheric ammonia on vegetation-a review. Envir Poll 86:43–82
Fitzsimmons EG (2014) Tap water ban for Toledo residents. The New York Times.. https://www.nytimes.com/2014/08/04/

us/toledo-faces-second-day-of-water-ban.html. 3 Aug 2014
Fixen P, Brentrup F, Bruulsema T, Garcia F, Zingore NRS (2015) Nutrient/fertilizer use efficiency: measurement, current situation and trends. In: Drescher PHP, Magen H, Mikkelsen R, Wichelns D (eds) Managing water and fertilizer for sustainable agriculture intensification. International Fertilizer Association (IFA), International Water Management Institute (IWMI), International Plant Nutrition Institute (IPNI), International Potash Institute (IPI), Paris, pp 8–37
Fleming LE, Kirkpatrick B, Bean BL, J, Wanner A, Dalpra D, Tamer R, et al (2005) Initial evaluation of the effects of aerosolized Florida red tide toxins (brevetoxins) in persons with asthma. Environ Health Perspec 113–115:650–657
Foley J (2013) It's time to rethink America's corn system. Sci Am March 5, 2013.
Food and Water Watch (2015) Factory Farm Nation 2015. https://www.foodandwaterwatch.org/sites/default/files/factory-farm-nation-report-may-2015.pdf.
Food and Water Watch (2020) Factory Farm Nation 2020. https://www.foodandwaterwatch.org/sites/default/files/ib_2004_updfacfarmmaps-web2.pdf
Fu F-X, Place AR, Garcia NS, Hutchins DA (2010) CO_2 and phosphate availability control the toxicity of the harmful bloom dinoflagellate *Karlodinium veneficum*. Aquat Microb Ecol 59:55–65. https://doi.org/10.3354/ame01396
Galloway JN, Aber J, Erisman J, Seitzinger SP, Howarth RW, Cowling E, Cosby B (2003) The nitrogen cascade. Bioscience 53(4):341–356
Galloway JN, Cowling EB, Seitzinger SP, Socolow RH (2002) Reactive nitrogen: too much of a good thing? Ambio 31:60–63
Galloway JN, Winiwarter W, Leip A, Leach AM, Bleeker A et al (2014) Nitrogen footprints: past, present and future. Environ Res Lett 9:115003. https://doi.org/10.1088/1748-9326/9/11/115003
Gaworski M, Jablònski S, Pawlaczyk-Graja I, Ziewiecki R, Rutkowski P, Wieczynska A, Gancarz R, Lukaszewicz M (2017) Enhancing biogas plant production using pig manure and corn silage by adding wheat straw processed with liquid hot water and stream explosion. Biotechnol Biofuels 10:259. https://doi.org/10.1186/s13068-017-0922-x
Glibert PM (2017) Eutrophication, harmful algae and biodiversity- challenging paradigms in a world of complex nutrient changes. Mar Poll Bull 124:591–606. https://doi.org/10.1016/j.marpolbul.2017.04.027
Glibert PM (2020) Harmful algae at the complex nexus of eutrophication and climate change. Harmful Algae. https://doi.org/10.1016/j.hal.2019.03.001
Glibert PM, Burford MA (2017) Globally changing nutrient loads and harmful algal blooms: recent advances, new paradigms, and continuing challenges. Oceanography 30(1):58–69. https://doi.org/10.5670/oceanog.2017.110
Glibert PM, Burkholder JM (2018) Causes of harmful algal blooms. In: Shumway SE, Burkholder JM, Morton SL (eds) Harmful algal blooms: a compendium desk reference. Wiley, Singapore, pp 1–38
Glibert PM, Legrand C (2006) The diverse nutrient strategies of HABs: focus on osmotrophy. In: Granéli E, Turner J (eds) Ecology of harmful algae. Springer, Cham, pp 163–176
Glibert PM, Heil CA, Hollander D, Revilla M, Hoare A, Alexander J, Murasko S (2004) Evidence for dissolved organic nitrogen and phosphorus uptake during a cyanobacterial bloom in Florida Bay. Mar Ecol Prog Ser 280:73–83
Glibert PM, Seitzinger S, Heil CA, Burkholder JM, Parrow MW, Codispoti LA, Kelly V (2005) The role of eutrophication in the global proliferation of harmful algal blooms: new perspectives and new approaches. Oceanography 18(2):198–209
Glibert PM, Harrison JA, Heil CA, Seitzinger S (2006) Escalating worldwide use of urea: a global change contributing to coastal eutrophication. Biogeochemistry 77:441–463
Glibert PM, Fullerton D, Burkholder JM, Cornwell J, Kana TM (2011) Ecological stoichiometry, biogeochemical cycling, invasive species and aquatic food webs: San Francisco Estuary and comparative systems. Rev Fish Sci 19:358–417
Glibert PM, Kana TM, Brown K (2013) From limitation to excess: consequences of substrate excess and stoichiometry for phytoplankton physiology, trophodynamics and biogeochemistry, and implications for modeling. J Mar Syst 125:14–28. https://doi.org/10.1016/j.jmarsys.2012.10.004
Glibert PM, Manager R, Sobota DJ, Bouwman L (2014) The Haber-Bosch-Harmful algal bloom (HB-HAB) link. Environ Res Lett 9:105001. https://doi.org/10.1088/1748-9326/9/10/105001
Glibert PM, Wilkerson FP, Dugdale RC, Raven JA, Dupont C, Leavitt PR, Parker AE, Burkholder JM, Kana TM (2016) Pluses and minuses of ammonium and nitrate uptake and assimilation by phytoplankton and implications for productivity and community composition, with emphasis on nitrogen-enriched conditions. Limnol Oceanogr 61:165–197
Godfrey HCJ, Aveyard P, Garnett T, Hall JW, Key TJ, Lorimer J, Pierrrehumbert RP, Scarborough P, Springmann M, Jebb SA (2018a) Meat consumption, health, and the environment. Science 361:5324. https://doi.org/10.1126/science.aam5324
Glibert PM, Beusen AHW, Harrison JA, Durr HH, Bouwman A, Laruelle GG (2018b) Changing land, sea- and airscapes: Sources of nutrient pollution affecting habitat suitability for harmful algae. In: Glibert PM, Berdalet E, Burford M, Pitcher G, Zhou M (eds) Ecology and oceanography of harmful algal blooms (GEOHAB). Springer, Cham, pp 53–76
Grant RH, Boehm MT (2015) Manure NH_3 and H_2S emissions from a western dairy storage basin. J Envir Qual 44:127–136. https://doi.org/10.2134/jeq2014.05.0196
Granéli E, Flynn KJ (2006) Chemical and physical factors influencing toxin content. In: Granéli E, Turner J (eds) Ecology of harmful algae. Springer, Dordrecht, pp 229–241
Granéli E (2005) Harmful algal blooms. In: Wassmann P, Olli K (eds) Drainage basin inputs and eutrophication: An integrated approach. University of Tromsø, Norway, pp 99–112
Gratton LM, Schumacker J, Reich J, Holobaugh A (2018) Public health and epidemiology. In: Shumway SE, Burkholder

JM, Morton SL (eds) Harmful algal blooms: a compendium desk reference. Wiley, Singapore, pp 355–376

van Grinsven HJM, Holland M, Jacobsen BH, Klimont Z, Sutton MA, Willems WJ (2013) Costs and benefits of nitrogen for Europe and implications for mitigation. Environ Sci Technol 47:3371–3379

Grossi G, Goglio P, Vitali A, Williams V (2019) Livestock and climate changes: impact of livestock on climate and mitigation strategies. Anim Front. https://doi.org/10.1093/af/vfy034

Guisande C, Frangópulos M, Maneiro I, Vergara AR, Riveiro I (2002) Ecological advantages of toxin production by the dinoflagellates *Alexandrium minutum* under phosphorus limitation. Mar Ecol Prog Ser 225:169–176

Gunderson M, Torres A, Boehlje M, Phillips R (2018) Systems. In: Eise J, Foster K (eds) How to feed the world. Island Press, Washington, DC, pp 94–115

Hajdu S, Pertola S, Kuosa H (2005) *Prorocentrum minimum* (Dinophyceae) in the Baltic Sea: morphology, occurrence: a review. Harmful Algae 4:471–480

Handan-Nader C, Ho DE (2019) Deep learning to map concentrated animal feeding operations. Nat Sustain 2:298–306

Handy SM, Demir E, Hutchins DA, Portune KJ, Wherat EB, Hare CE, Rose JM, Warner M, Farestad M, Cary SC, Coyne KJ (2008) Using quantitative real-time PCR to study competition and community dynamics among Delaware Inland Bays harmful algae in field and laboratory studies. Harmful Algae 7:599–613

Hanson JD, Hendrickson JR (2009) Towards a sustainable agriculture. In: Franzluebbers AJ (ed) Farming with grass: achieving sustainable mixed agricultural landscapes. Soil Water Conserv Soc, Ankeny, pp 26–36

Harper LA, Sharpe RR, Parkin TB, de Visscher A, van Cleemput O, Byers M (2004) Nitrogen cycling through swine production systems: Ammonia, dinitrogen, and nitrous oxide emissions. J Environ Qual 33:1189–1201

Harris TD, Smith VH, Graham JL, Van de Waal DB, Tedesco LP, Clercin N (2016) Combined effects of nitrogen and phosphorus and nitrate to ammonia ratios on cyanobacterial metabolite concentrations in eutrophic Midwestern USA reservoirs. Inland Waters 6:199–210

Heisler J, Glibert PM, Burkholder JM, Anderson DM, Cochlan W, Dennison WC, Dortch Q, Gobler CJ, Heil CA, Humphries E, Lewitus A, Magnien R, Marshall HG, Sellner K, Stockwell DA, Stoecker DK, Suddleson M (2008) Eutrophication and harmful algal blooms: a scientific consensus. Harmful Algae 8:3–13

Herrero N, Wirsenius S, Henderson B, Rigolot C, Thornton P, Havlik P, de Boer I, Gerber PJ (2015) Livestock and the environment: what have we learned in the past decade? Ann Rev Envir Res 40:177–202. https://doi.org/10.1146/annurev-environ-031113-003503

Hertel TW (2018) Tangled trade. In: Eise J, Foster K (eds) How to feed the world. Island Press, Washington, DC, pp 115–131

Houlton BZ, Almaraz A, Aneja V, Austin AT, Bai E, Cassman KG, Compton JA, Davidson EA et al (2019) A world of cobenefits: solving the global nitrogen challenge. Earth's Future. https://doi.org/10.1029/2019EF001222

Houlton BZ, Boyer E, Finzi A, Galloway J, Leach A, Liptzin D, Melillo J, Rosenstock TS, Sobota D, Townsend AR (2013) Intentional versus unintentional nitrogen use in the United States: trends, efficiency and implications. Biogeochemistry 114:11–23. https://doi.org/10.1007/s10533-012-9801-5

Howarth RW (2008) Coastal nitrogen pollution: a review of sources and trends globally and regionally. Harmful Algae 8:14–20

Howarth RW, Paerl H (2008) Coastal marine eutrophication: control of both nitrogen and phosphorus is necessary. Proc Natl Acad Sci USA 105:E103

Howarth RW, Sharpley A, Walker D (2002) Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. Estuaries 25:656–676

Hribar C (2010) Understanding concentrated animal feeding operations and their impact on communities. National Association of Local Boards of Health, Bowling Green, OH, https://www.cdc.gov/nceh/ehs/docs/understanding_cafos_nalboh.pdf.

Hunt D, Bittman S, Chantigny M, Lemke R (2019) Year-round N_2O emissions from long-term applications of whole and separated liquid dairy slurry on a perennial grass sward and strategies for mitigation. Front Sustain Food Syst. https://doi.org/10.3389/fsufs.2019.00086

IPCC (Intergovernmental Panel on Climate Change) (2007) Fourth Assessment Report. https://www.ipcc.ch/assessment-report/ar4/.

IPCC (Intergovernmental Panel on Climate Change) (2019) Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems.https://www.ipcc.ch/report/srccl/

Imhoff D (2019) The farm bill: a citizen's guide. Island Press, Washington DC, p 258

Ivahnenko T (2017) Evaluation and use of US Environmental Protection Agency clean water needs survey data to quantify nutrient loads to surface water, 1978–2012, Rep. 2328–0328. US Geological Survey.

Kalcic MM, Muenich RL, Basile S, Steiner AL, Kirchhoff C, Scavia D (2019) Climate change and nutrient loading in the western Lake Erie basin: warming can counteract a wetter future. Environ Sci Tech 53: https://doi.org/10.1021/acs.est.9b01274

Kanakidou M, Myriokefalitakis S, Daskalakis N, Fanourgakis G, Nenes A, Baker A, Tsigaridis K, Mihalopoulon N (2016) Past, present and future atmospheric nitrogen deposition. J Atmos Sci 73:2039–2047. https://doi.org/10.1175/JAS-D-15-0278.1

Kemp WM, Testa J, Conley DJ, Gilbert D, Hagy J (2009) Temporal responses of coastal hypoxia to nutrient loading and physical controls. Biogeosciences 6:2985–3008

Ketchum BH (ed) (1972) Contamination and coastal pollution through waste disposal practices. The water's edge: critical problems of the coastal zone. The MIT Press, Cambridge, pp 146–186

Kundu P, Debsarkar A, Mukherjee S (2013) Treatment of slaughter house wastewater in a sequencing batch reactor: performance evaluation and biodegradation kinetics. Biomed Res Int 2013:134872

Landsberg JH (2002) The effects of harmful algal blooms on aquatic organisms. Rev Fish Sci 10:113–390. https://doi.org/10.1080/20026491051695

Leytem AB, Bjrneberg DL, Koehn AC, Moraes LE, Kebreab E, Dungan RS (2017) Methane emissions from dairy lagoons in the western United States. J Dairy Sci 8:6785–6803. https://doi.org/10.3168/jds.2017-1277a

Li J, Glibert PM, Gao Y (2015) Temporal and spatial changes in Chesapeake Bay water quality and relationships to *Prorocentrum minimum, Karlodinium veneficum*, and CyanoHAB events, 1991–2008. Harmful Algae 42:1–14

Li J, Glibert PM, Zhou M, Lu S, Lu S (2009) Relationships Between nitrogen and phosphorus forms and ratios and the development of dinoflagellate blooms in the East China Sea. Mar Ecol Prog Ser 383:11–26

Likens GE (ed.). (1972) Eutrophication and aquatic ecosystems. Limnol Oceanogr Spec Symp 1: 328. https://doi.org/10.1043/lo.1972.17.6.0965.

Loftin KA, Graham JA, Hilborn ED, Lehmann SC, Meyer MT, Dietz JE, Griffith CB (2016) Cyanotoxins in inland lakes of the United States: occurrences and potential recreational health risks in the EPA National Lakes Assessment 2007. Harmful Algae 56:77–90. https://doi.org/10.1016/j.hal.2016.04.001

Loladze I (2014) Hidden shift of the ionome of plants exposed to elevated CO2 depletes minerals at the base of human nutrition. eLife 3:e02245. https://doi.org/10.7554/eLife.02245

Lorimer J, Powers W, Sutton A (2014) Manure characteristics. MidWest Plan Service, Iowa State University.Report MWPS-18 Section 1. https://www.canr.msu.edu/uploads/files/ManureCharacteristicsMWPS-18_1.pdf

Mallin MA (2000) Impacts of industrial-scale swine and poultry production on rivers and estuaries. Am Sci 88:26–37

Mallin MA, Cahoon LB (2003) Industrialized animal production: a major source of nutrient and microbial pollution to aquatic ecosystems. Pop Environ 24:369–385

Mallin MA, McIver MR, Robuck AR, Dickens AK (2015) Industrial swine and poultry production causes chronic nutrient and fecal microbial stream pollution. Water Air Soil Pollut 226: https://doi.org/10.1007/s11270-015-2669-y

Martin KL, Emanuel RE, Vose JM (2018) Terra incognita: the unknown risks to environmental quality posed by the spatial distribution and abundance of concentrated animal feeding operations. Sci Total Environ 642:887–893. https://doi.org/10.1016/j.scitotenv.2018.06.072

McCabe RM, Hickey BM, Kudela RM, Lefebve KA, Adams NG, Bill BD, Gulland FMD, Thomson RE, Cochlan WP, Trainer VL (2016) An unprecedented coastwide toxic algal bloom linked to anomalous ocean conditions. Geophys Res Lett. https://doi.org/10.1002/2016GL070023

Van Meter KJ, Basu NB, Van Cappellen P (2017) Two centuries of nitrogen dynamics: Legacy sources and sinks in the Mississippi and Susquehanna River Basins. Glob Biogeochem Cycles 31(1):2–23. https://doi.org/10.1002/2016GB005498

Van Meter KJ, Basu NB, Veenstra JJ, Burras CL (2016) The nitrogen legacy: Emerging evidence of nitrogen accumulation in anthropogenic landscapes. Environ Res Letts 11(3):035014. https://doi.org/10.1088/1748-9326/11/3/035014

Michelak AM, Anderson EJ, Beletsky D et al (2013) Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. Proc Natl Acad Sci US. https://doi.org/10.1073/pnas.1216006110

Millenium Ecosystem Assessment (2005) Ecosystems and human well-being: Current state and trends: findings of the condition and trends working group (millennium ecosystem assessment series). Island Press.

Miller DL (2017) A seat at the table: new voices urge Farm Bill reform. Yale Law J. Forum 395 https://www.yalelawjournal.org/pdf/Miller_hierxcys.pdf.

Miller L, Muren G (2019) CAFOs: what we don't know is hurting us. National Resources Defense Council. https://www.nrdc.org/sites/default/files/cafos-dont-know-hurting-us-report.pdf.

Mueller ND, Lassaletta L, Runck BC, Billen G, Garnier J, Gerber JS (2017) Declining spatial efficiency of global cropland nitrogen allocation. Glob Biogeochem Cycles.

Nature (editorial) (2019). Counting the hidden $12 trillion cost of a broken food system. Nature 286. https://doi.org/10.1038/d41586-019-03117-y

Niles MT, Wiltshire S (2019) Tradeoffs in US dairy manure greenhouse emissions, productivity, climate, and manure management strategies. Environ Res Commun 1:075003. https://doi.org/10.1088/2515-7620/ab2dec

Oh H-M, Lee SJ, Jang M-J, Yoon B-D (2000) Microcystin production by *Microcystis aeruginosa* in a phosphorus-limited chemostat. Appl Environ Microbiol 66:176–179

Ortiz JL (2019) The Amazon hasn't stopped burning. There were 19,925 fire outbreaks last month, and 'more fires' are in the future. USA Today https://www.usatoday.com/story/news/nation/2019/10/18/amazon-rainforest-still-burning-more-fires-future/4011238002/. 18 Oct 2019

Paerl HW (1997) Coastal eutrophication and harmful algal blooms: importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. Limnol Oceanogr 42:1154–1165

Paerl HW (2009) Controlling eutrophication along the freshwater-marine continuum: dual nutrient (N and P) reductions are essential. East Coasts 32:593–601

Paerl HW, Scott JT (2010) Throwing fuel on the fire: Synergistic effects of excessive nitrogen inputs and global warming on harmful algal blooms. Environ Sci Technol 44:7756–7758. https://doi.org/10.1021/es102665e

Pelton T, Lamm M, Russ A (2020) Poultry industry pollution in the Chesapeake region: ammonia air emissions and nitrogen loads higher than EPA estimates. Environment Integrity Project. https://environmentalintegrity.org/wp-content/uploads/2020/04/Chesapeake-Poultry-Report-EMBARGOED-for-4.22.20.pdf.

Peterson SO (2018) Greenhouse gas emissions from liquid dairy manure: Prediction and mitigation. J Dairy Sci 101:6642–6654. https://doi.org/10.3168/jds.2017-13301

Pingali P (2012) Green revolution: impacts, limits, and the path ahead. PNAS 109(31):12302–12308. https://doi.org/10.1073/pnas.0912953109

Pitt D (2020) Tens of thousands of healthy pugs being killed daily as meatpacking backlog hits hog farms. https://time.com/

5831061/pigs-killed-coronovirus-meatpacking-shutdowns/. Accessed 20 May 2010
Pollen M (2006) The Omnivore's Dilemna: a natural history of four meals. Penguin press, New York, p 450
Potter P, Ramankutty N, Bennett EM, Donner SD (2010) Characterizing the spatial patterns of global fertilizer application and manure production. Earth Interact 14: https://doi.org/10.1175/20099E1288.1
Poulsen MN, Pollak J, Sills DL, Casey JA, Nachman KE, Cosgrove SE, Stewart D, Schwatz BS (2018) High-density poultry operations and community-acquired pneumonia in Pennsylvania. Environ Epidemiol 2: https://doi.org/10.1097/EE9.0000000000000013
Prakash O, Alva AK, Paramasivam S (1999) Use of urease inhibitor N-(n-butyl)-thiophosphoric triamide decreased nitrogen leaching from urea in a fine sandy soil. Water Air Soil Pollut 116:587–595
Rabalais NN, Díaz RJ, Levin LA, Turner RE, Gilbert D, Zhang J (2010) Dynamics and distribution of natural and human-caused coastal hypoxia. Biogeosciences 7:585–619
Rabalais NN, Turner RE, Díaz RJ, Justic D (2009) Global change and eutrophication of coastal waters. ICES J Mar Sci 66:1528–1537
Ribaudo M, Delgado J, Hansen L, Livingston M, Mosheim R, Williamson J (2011) Nitrogen in agricultural systems: Implications for conservation policy. Economic Res. Report No. ERR-127. 89 pp. https://www.ers.usda.gov/publications/pub-details/?pubid=44919.
Ruddy BC, Lorenz DL, Mueller DK (2006) County-level estimates of nutrient inputs to the land surface of the conterminous United States, 1982–2001. US Geological Survey Scientific Investigations Report 2006–5012, 17 pp.
Ruhl JB (2000) Farms, their environmental harms, and environmental law. Ecology Law Quart 27:263–349
Rumpler J (2016) Corporate agribusiness and the fouling of America's waterways. Environmental Virginia Research and Policy Center. Richmond: 46p. https://environmentamerica.org/sites/environment/files/reports/CorpAgFoulingWaterways2016-web_0.pdf
Russ A, Schaeffer E (2018) Ammonia emissions from broiler operations higher than previously thought. 2. Environmental Integrity Project, January 2018. https://www.environmentalintegrity.org/wp-content/uploads/2017/02/Ammonia-Report.pdf
Sabo RD, Clark CM, Bash J, Sobota D, Cooter E, Dobrowolski JP, Houlton BZ, Rea A, Schwede D, Morford SL, Compton JE (2019) Decadal shift in nitrogen inputs and fluxes across the contiguous United States: 2002–2012. JGR Biogeosci 124:3104–3124. https://doi.org/10.1029/2019JG005110
Sakadevan K, Nguyan M-L (2016) Livestock production and its impact on nutrient pollution and greenhouse gas emissions. Adv Agron. https://doi.org/10.1016/bs.agron/2016.10.002
Scavia D, Rabalais NN, Turner RE, Justic D, Wiseman WJ (2003) Predicting the response of Gulf of Mexico hypoxia to variations in Mississippi River nitrogen load. Limnol Oceanogr 48:951–956
Schneider SA (2010) A reconsideration of agricultural law: a call for the law of food, farming and sustainability. William Mary Environ Law Policy Rev. 34:935–963
Schindler DW, Hecky RE (2008) Reply to Howarth and Paerl: is control of both nitrogen and phosphorus necessary? Proc Natl Acad Sci USA 105:E104. https://doi.org/10.1073/pnas.0809744105
Schindler DW, Hecky RE, Findlay DL, Stainton MP, Parker BR, Paterson MJ, Beaty KG, Lyng M, Kasian SEM (2008) Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proc Natl Acad Sci USA 105:11254–11258
Schindler DW, Carpenter SR, Chapra SC, Hecky RE, Orihel DM (2016) Reducing phosphorus to curb lake eutrophication is a success. Environ Sci Technol. https://doi.org/10.1021/acs.est.6b02204
Sheldon I (2019) Why China's soybean tariffs matter. Salon, April 8, 2018. https://www.salon.com/2018/04/08/why-chinas-soybean-tariffs-matte_partner/.
Sims JT, Simard RR, Joern BC (1998) Phosphorus loss in agricultural drainage: historical perspective and current research. J Environ Qual 29:111–116
Smil V (2001) Enriching the earth: fritz haber, carl bosch, and the transformation of world food. The MIT Press, Cambridge, p 338
Smith VH, Joye SB, Howarth RW (2006) Eutrophication of freshwater and marine ecosystems. Limnol Oceanogr 51:351–355.
Sobota DJ, Compton JE, McCracklin ML, Singh S (2015) Cost of reactive nitrogen release from human activites to the environment in the United States. Environ Res Lett 10:025006. https://doi.org/10.1088/1748-9326/10/2/025006
Springer JJ, Burkholder JM, Glibert PM, Reed RE (2005) Use of a real-time remote monitoring network and shipborne sampling to characterize a dinoflagellate bloom in the Neuse Estuary, North Carolina, U.S.A. Harmful Algae 4:533–551
Stauber KN, Hassebrook C, Bird EAR, Bultena GL, Hoiberg EO, MacCormack H, Menanteau-Horta D (1995) The promise of sustainable agriculture. In: Bird EAR, Bultena GL, Gardner JC (eds) Planting the future: developing an agriculture that sustains land and community. Iowa State Univ Press, Ames, pp 3–15
Steffen MM, Davis TW, McKay RML, Bullerjahn GS, Krausfeldt LE, Stough JM et al (2017) Ecophysiological examination of the Lake Erie *Microcystis* bloom in 2014: linkages between biology and the water supply shutdown of Toledo, OH. Environ Sci Technol 51:6745–6755. https://doi.org/10.1021/acs.est.7b00856
Sullivan Z (2019) The real reason the Amazon is on fire. Time. https://time.com/5661162/why-the-amazon-is-on-fire/. 26 Aug 2019
Sutton MA et al (2013) Our nutrient world: the challenge to produce more food and energy with less pollution. Centre for Ecology and Hydrology (CEH) on Behalf of the Global Partnership on Nutrient Management and the International Nitrogen Initiative, Edinburgh
Swaney DP, Howarth RW, Hong B (2018a) Nitrogen use efficiency and crop production: patterns of regional variation in the United States, 1987–2012. Sci Total Environ 635:498–511. https://doi.org/10.1016/j.scitotenv.2018.04.027
Swaney DP, Howarth RW, Hong B (2018b) County, subregional and regional nitrogen data derived from the Net

Anthropogenic Nitrogen Inputs (NANI) toolbox. Data in Brief 18:1877–1888

Tomas KA (2019) Manure management for climate change mitigation: Regulating CAFO greenhouse gas emissions under the Clean Air Act. Univ. Miami Law Review. pp. 531–568. https://repository.law.miani.edu/umlr/vol73/iss2/8/

Townsend AR, Howarth RW, Bazzazz FA, Booth MS, Cleveland CC, Collinge SK et al (2003) Human health effects of a changing global nitrogen cycles. Front Ecol Environ 1:240–246. https://doi.org/10.1890/1540-9295(2003)001[0240:HHEOAC]2.0.CO;2

Turner RE, Rabalais NN, Justic D (2006) Predicting summer hypoxia in the northern Gulf of Mexico: Riverine N, P, and Si loading. Mar Poll Bull 52:139–148

US EPA (2003) National pollution discharge elimination system permit regulation and effluent limitation guidelines and standards for concentrated animal feeding operations (CAFOs). https://www.federalregister.gov/documents/2003/02/12/03-3074/national-pollutant-discharge-elimination-system-permit-regulation-and-effluent-limitation-guidelines

US EPA (US Environmental Protection Agency) (2010) Implementation guidance on CAFO regulations–CAFOs that discharge or are proposing to discharge. https://www3.epa.gov/npdes/pubs/sector_table.pdf

US EPA (2017) Inventory of greenhouse gas emissions and sinks, 1990–2017. EPA430-R-19-001. www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks

USDA (US Department of Agriculture) (2008) Farm and Ranch irrigation survey https://www.nass.usda.gov/Publications/AgCensus/2007/Online_Highlights/Farm_and_Ranch_Irrigation_Survey/

USDA (2016) US Agriculture and Forestry Greenhouse gas inventory 1990–2013. Office of the Chief Economist, Climate Change Program Office, Technical Bulletin No. 1943, Sept 2016. https://doi.org/10.15482/USDA.ADC/1264344.

USDA (2019) Nutrient management. https://www.ers.usda.gov/topics/farm-practices-management/crop-livestock-practices/nutrient-management/

Vilmin L, Mogollón JM, Beusen AHW, Bouwman AF (2018) Forms and subannual variability of nitrogen and phosphorus loading to global river networks over the 20th century. Glob Planet Change 163:67–85. https://doi.org/10.1016/j.gloplacha.2018.02.007

Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schelsinger WH, Tilman DG (1997) Human alteration of the global nitrogen cycle: sources and consequences. Ecol Appl 7:737–750. https://doi.org/10.1890/1051-0761(1997)007[0737:HAOTGNJ2.0.CO;2

Van de Waal DB, Smith VH, Declerck SAJ, Stam ECM, Elser JJ (2014) Stoichiometric regulation of phytoplankton toxins. Ecol Lett 17:736–742

Van de Waal DB, Verspagen JM, Lurling M, Van Donk E, Visser PM, Huisman J (2009) The ecological stoichiometry of toxins produced by harmful cyanobacteria: an experimental test of the carbon-nutrient balance hypothesis. Ecol Lett 12:1326–1335

von Reusner L (2019) Mystery meat II: The industry behind the quiet destruction of the American Heartland. www.mightearth.org

Walker KD (2019) The grand food bargain and the mindless drive for more. Island Press, Washington DC, p 327

Walljasper C (2018) Large animal feeding operations on the rise. https://investigateMidwest.org/2018/06/07/large-animal-feeding-operations-on-the-rise/.

Walton B (2016) EPA turns away from CAFO water pollution. Circle of Blue Water News. https://www.circleofblue.org/2016/water-policy-politics/epa-turns-away-cafo-water-pollution/. 22 Dec 2016

Weigel H-J (2014) Plant quality declines as CO_2 levels rise. eLife 3:e03233. https://doi.org/10.7554/eLIfe.02245

Whitall D, Hendrickson B, Paerl H (2003) Importance of atmospherically deposited nitrogen to the annual nitrogen budget of the Neuse River Estuary, NC. Environ Intern 29:393–399

Wilton A, Newton J (2019) Farm loan delinquencies and bankruptcies are rising. Market Intel. https://www.fb.org/market-intel/farm-loan-delinquencies-and-bankruptcies-are-rising. 31 July 2019

Yang Q, Tain H, Li X, Ren W, Zhang B, Zhang X, Wolf J et al (2016) Spatiotemporal patterns of livestock manure nutrient production in the conterminous United States from 1930 to 2012. Sci Total Environ 541:1592–1602. https://doi.org/10.1016/j.scitotenv.2015.10.044

Zhang X (2017) A plan for efficient use of nitrogen fertilizers. Nature 543:322–333

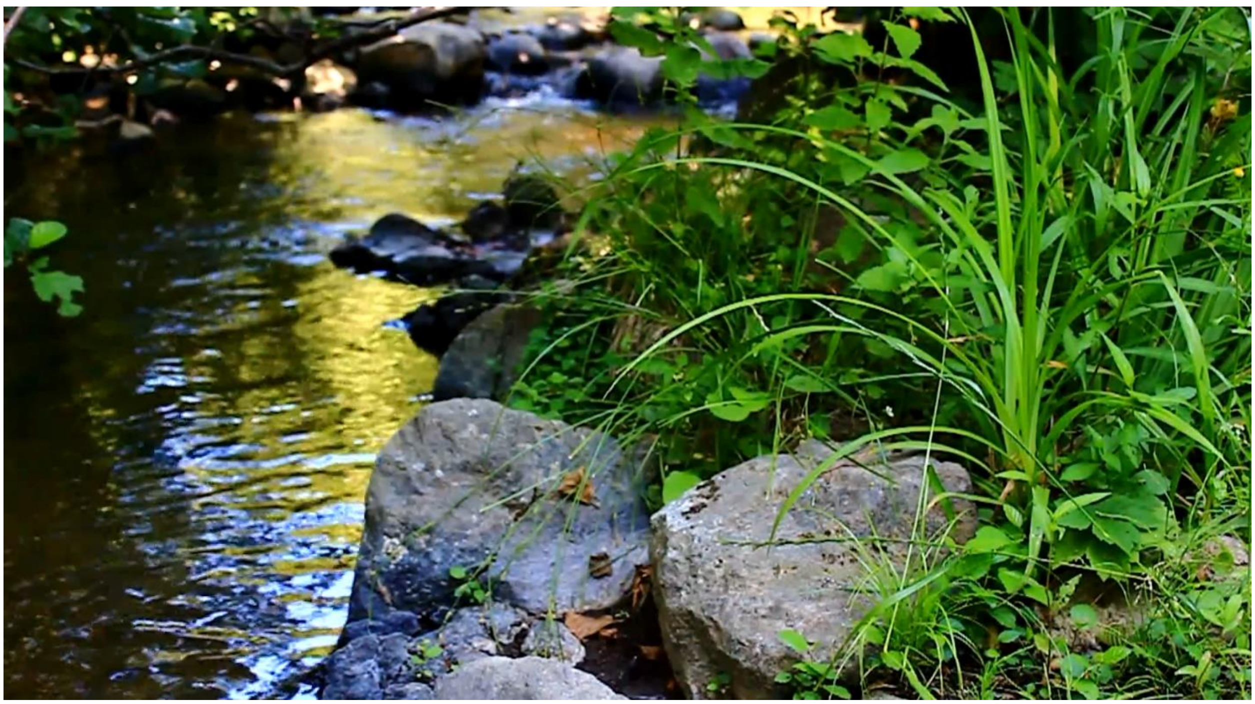
Zhang J, Beusen AHW, Van Apeldoorn DF, Mogollón JM, Yu C, Bouwman AF (2017) Spatiotemporal dynamics of soil phosphorus and crop uptake in global cropland during the 20th century. Biogeosciences 14(8):2055–2068

Zhou X, Passow FH, Rudek J, von Fisher JC, Hamburg SP, Albertson JD (2019) Estimation of methane emissions from the US ammonia fertilizer industry using a mobile sensing approach. Elementa 7:19. https://doi.org/10.1525/elementa.358

Iowa's water quality strategy is not working. Here's what should be done instead.

Iowa's Nutrient Reduction Strategy has a solid scientific foundation, but it relies on farmer altruism. It's clear this approach will require generations to produce measurable improvements.

NEIL HAMILTON, MATT LIEBMAN, SILVIA SECCHI, CHRIS JONES | IOWA VIEW CONTRIBUTORS | 10:33 am CST February 7, 2020



A study shows that nitrates in drinking water may be tied to 300 cases of cancer in Iowa each year.
OLIVIA SUN, DES MOINES REGISTER

In 2019, Iowa streams carried away a billion pounds of nitrogen and 50 million pounds of phosphorus, constituting an enormous financial loss to farmers, a serious degradation of drinking water and recreation, and a threat to human health and fisheries.

More than 90% of the nitrogen and 75% of the phosphorus in Iowa waters come from farm fields and livestock operations. Since the 2013 adoption of the Nutrient Reduction Strategy, **Iowa's water quality has not improved**.

How can substantial improvements in Iowa's water quality be achieved? As Iowans with a long involvement in agricultural science and policy, we believe there are three components for rapidly catalyzing water quality improvements: 1) Iowa should reconfigure its livestock industry; 2) regulation must play a parallel role with voluntary adoption of conservation practices; and 3) policies should be tailored to respond to changing climate and production systems.

While the number of Iowa farms and farmers continues to decline, since 1997 the population of **hogs has grown from 14 million to 25 million** and that of laying chickens from 29 million to more than 80 million now. The area cropped to corn and soybeans has increased by 2 million acres since 1982. This more intensive agricultural system requires more conservation just to maintain the water quality we have now. We believe Iowa's existing crop and livestock production framework is not, and will not ever be, consistent with our state's water quality objectives.

RELATED: Iowa could need hundreds of years to reach nutrient goals

Animals are so overpopulated in some areas that manure-borne nutrients far exceed crop needs. The current system, which decouples animal and crop production, prevents efficient nutrient recycling. Balancing the absorptive and productive capacity of the land with even mediocre water quality is impossible for water bodies from the Floyd River of northwest Iowa to Lake Darling of southeast Iowa, especially when commercial fertilizer sales continue unabated in watersheds with dense livestock populations.

Iowa's livestock industry has grown far beyond our agencies' capacity to enforce the weak regulations that we have. And our counties' citizens and elected officials have no power to guide continued expansion.

It's time to admit the obvious and regroup.

Iowa's Nutrient Reduction Strategy has a solid scientific foundation, but it relies on farmer altruism. It's clear this approach will require generations to produce measurable improvements. We think Iowans deserve better from an industry indemnified by the taxpayer through billions of dollars spent on trade mitigation payments, crop insurance subsidies, and disaster relief.

Poor water quality is not the result of callous, poorly informed or rogue farmers; rather it is the predictable result of land use policies, vulnerabilities of the corn-soybean-animal confinement scheme, and an economic system tyrannically ruling farmer decisions. If the public is to get the environmental outcomes they deserve, the system must change to support diversified and integrated crop and livestock production. This would benefit Iowa's water, help revitalize rural Iowa and breathe life into hundreds of our small towns.

Des Moines Register reporter Mackenzie Elmer collects data from a water sample she pulled during a water quality workshop at Jester Park in Granger on Friday, June 10, 2016.
BRYON HOULGRAVE/THE REGISTER

Without such change, and as long as the taxpayer is expected to prop up the system, then we say the public has a right to expectations for how the present system is operated. These expectations should include taxation of purchased fertilizer and animal feed, regulations that restrict or ban practices that degrade water quality, and requirements for practices that improve it, such as:

- Restrict cropping on frequently flooded land and planting up to the stream edge.
- Align fertilizer and manure inputs with Iowa State University recommendations by requiring farmers to implement nutrient management plans.
- Digitize land records for manure management plans so that fields aren't used too frequently for manure disposal.
- Replace the current livestock Master Matrix regulations with a system that allows governments to manage manure nutrients at the watershed scale.
- Ban manure application to snow-covered and frozen ground.

State leaders also need to recognize that economic and environmental resilience is intimately connected to weather, and that climate change is blowing Iowa into uncharted waters. As our weather gets warmer, wetter and more extreme, our current production systems will increasingly rely on expensive engineering solutions, just to maintain the status quo.

Who's going to pay for this? We think public dollars would be better used by reconfiguring the system in resilient ways that benefit all Iowans. New and existing funding should not be allocated to water quality measures without adequate monitoring and other mechanisms to assess effectiveness.

The challenge presented by our degraded water is enormous. We know of no problems approaching this magnitude that have been solved through individual actions. Iowans deserve more than meaningless platitudes and dogmatic devotion to voluntary approaches. Now is the time to act if we are to avoid another century of degraded water.

Neil Hamilton is emeritus professor of law at Drake University and former director of the Agricultural Law Center. Matt Liebman is professor of agronomy and H.A. Wallace Chair for Sustainable Agriculture at Iowa State University. Silvia Secchi is an associate professor in the Department of Geographical and Sustainability Sciences and Public Policy Center at the University of Iowa. Chris Jones is a research engineer with IIHR Hydroscience and Engineering at the University of Iowa.

Natural Resources Conservation Service

CONSERVATION PRACTICE STANDARD

NUTRIENT MANAGEMENT

CODE 590

(ac)

DEFINITION

Manage rate, source, placement, and timing of plant nutrients and soil amendments while reducing environmental impacts.

PURPOSE

This practice is used to accomplish one or more of the following purposes:

- Improve plant health and productivity.
- Reduce excess nutrients in surface and ground water.
- Reduce emissions of objectionable odors.
- Reduce emissions of particulate matter (PM) and PM precursors.
- Reduce emissions of greenhouse gases (GHG).
- Reduce emissions of ozone precursors.
- Reduce the risk of potential pathogens from manure, biosolids, or compost application from reaching surface and ground water.
- Improve or maintain soil organic matter.

CONDITIONS WHERE PRACTICE APPLIES

All fields where plant nutrients and soil amendments are applied. Does not apply to one-time nutrient applications at establishment of permanent vegetation.

CRITERIA

General Criteria Applicable to All Purposes

Develop a nutrient management plan for nitrogen (N), phosphorus (P), and potassium (K), which accounts for all known measurable sources and removal of these nutrients.

Sources of nutrients include, but are not limited to, commercial fertilizers (including starter and in-furrow starter/pop-up fertilizer), animal manures, legume fixation credits, green manures, plant or crop residues, compost, organic by-products, municipal and industrial biosolids, wastewater, organic materials, estimated plant available soil nutrients, and irrigation water.

When irrigating, apply irrigation water in a manner that reduces the risk of nutrient loss to surface and ground water.

Follow all applicable State requirements and regulations when applying nutrients near areas prone to contamination, such as designated water quality sensitive areas, (e.g., lakes, ponds, rivers and streams, sinkholes, wellheads, classic gullies, ditches, or surface inlets) that run unmitigated to surface or groundwater.

Soil and tissue testing and analysis
Base the nutrient management plan on current soil test results in accordance with land grant university (LGU) guidance, or industry practice when recognized by the LGU. Use soil tests no older than 2 years when developing new nutrient management plans. Use tissue testing, when applicable, for monitoring or adjusting the nutrient management plan in accordance with LGU guidance, or industry practice when recognized by the LGU.

For nutrient management plan revisions and maintenance, take soil tests on an interval recommended by the LGU or as required by local rules and regulations.

Collect, prepare, store, and ship all soil and tissue samples following LGU guidance or industry practice. The test analyses must include pertinent information for monitoring or amending the annual nutrient plan. Follow LGU guidelines regarding required analyses and test interpretations.

For soil test analyses, use laboratories successfully meeting the requirements and performance standards of the North American Proficiency Testing Program under the auspices of the Soil Science Society of America and NRCS or use an alternative NRCS- or State-approved certification program that considers laboratory performance and proficiency to assure accuracy of soil test results. Alternative certification programs must have solid stakeholder support (e.g., State department of agriculture, LGU, water quality control entity, NRCS State staff, growers, and others) and be State or regional in scope.

Maintain soil pH within ranges which enhance the adequate level for plant or crop nutrient availability and utilization. Refer to State LGU documentation for guidance.

Manure, organic by-product, and biosolids testing and analysis
Collect, prepare, store, and ship all manure, organic by-products, and biosolids following LGU guidance or industry practice when recognized by the LGU. In the absence of such guidance, test at least annually, or more frequently if needed to account for operational changes (e.g., feed management, animal type, manure handling strategy, etc.) impacting manure nutrient concentrations. If no operational changes occur and operations can document a stable level of nutrient concentrations for the preceding 3 consecutive years, manure may be tested less frequently, unless Federal, State, or local regulations require more frequent testing. Follow LGU guidelines regarding required analyses and test interpretations. Analyze, as a minimum, total N, total P or P_2O_5, total K or K_2O, and percent solids.

When planning for new or modified livestock operations, and manure tests are not available yet, use the output and analyses from similar operations in the geographical area if they accurately estimate nutrient output from the proposed operation or use “book values” recognized by the NRCS (e.g., NRCS Agricultural Waste Management Field Handbook) and the LGU.

For manure analyses, use laboratories successfully meeting the requirements and performance standards of the Manure Testing Laboratory Certification program under the auspices of the Minnesota Department of Agriculture or other NRCS-approved program that considers laboratory performance and proficiency to assure accurate manure test results.

For nutrient management plans developed as a component of a comprehensive nutrient management plan for an animal feeding operation (AFO) follow policy in NRCS directive General Manual (GM) 190, Part 405, “Comprehensive Nutrient Management Plans.” These plans must include documentation of all nutrient imports, exports, and on-farm transfers.

Nutrient loss risk assessments
Use current NRCS-approved nitrogen, phosphorus, and soil erosion risk assessment tools to assess the site-specific risk of nutrient and soil loss.

Complete an NRCS-approved nutrient risk assessment for N on all fields where nutrient management is planned unless the State NRCS, in cooperation with State water quality control authorities, has determined specific conditions where N leaching is not a risk to water quality, including drinking water.

Complete an NRCS-approved nutrient risk assessment for P when any of the following conditions are met—

- P application rate exceeds LGU fertility rate guidelines for the planned crop(s).
- The planned area is within a P-impaired watershed.
- The site-specific conditions equating to low risk of P loss have not been determined by the NRCS in cooperation with the State water quality control authority.

Any fields excluded from a P risk assessment must have a documented agronomic need for P, based on soil test P and LGU nutrient recommendations.

For fields receiving manure, where P risk assessment results equate to—

- LOW risk.—Manure can be applied at rates to supply P at greater than crop requirement not to exceed the N requirement for the succeeding crop.
- MODERATE risk.—Manure can be applied at rates not to exceed crop P removal rate or the soil test P recommended rate for the planned crops in rotation.
- HIGH risk.—Manure can be applied at rates not to exceed crop P removal rate if the following requirements are met:
    - A soil P drawdown strategy has been developed, documented, and implemented for the crop rotation.
    - Implementation of all mitigation practices determined to be needed by site-specific assessments for nutrients and soil loss to protect water quality.
    - Any deviation from these high-risk requirements that would increase the risk of P runoff requires the approval of the Chief of the NRCS.

The 4Rs of nutrient stewardship
Manage nutrients based on the 4Rs of nutrient stewardship—apply the right nutrient source at the right rate at the right time in the right place—to improve nutrient use efficiency by the crop and to reduce nutrient losses to surface and groundwater and to the atmosphere.

<u>Nutrient source</u>
Choose nutrient sources compatible with application timing, tillage and planting system, soil properties, crop, crop rotation, soil organic content, and local climate to minimize risk to the environment.

Determine nutrient values of all nutrient sources (e.g. commercial fertilizers, manure, organic by-products, biosolids) prior to land application.

Determine nutrient contribution of cover crops, previous crop residues, and soil organic matter.

For operations following USDA's National Organic Program, apply and manage nutrient sources according to program regulations.

For enhanced efficiency fertilizer (EEF) products, use products defined by the Association of American Plant Food Control Officials as EEF and recommended for use by the State LGU.

In areas where salinity is a concern, select nutrient sources that limit the buildup of soil salts. When manures are applied, and soil salinity is a concern, monitor salt concentrations to prevent potential plant or crop damage and reduced soil quality.

Apply manure or organic by-products on legumes at rates no greater than the LGU estimated N removal rates in harvested plant biomass, not to exceed P risk assessment limitations.

For any single application of nutrients applied as liquid (e.g., liquid manure, nutrients in irrigation water, fertigation)—

- Do not exceed the soil's infiltration rate or water holding capacity.
- Apply so that nutrients move no deeper than the current crop rooting depth.
- Avoid runoff or loss to subsurface tile drains.

Nutrient rate
Plan nutrient application rates for N, P, and K using LGU recommendations or industry practices when recognized by the LGU. Lower-than-recommended nutrient application rates are permissible if the client's objectives are met.

At a minimum, determine the rate based on crop/cropping sequence, current soil test results, and NRCS-approved nutrient risk assessments. Where applicable, use realistic yield goals.

For new crops or varieties where LGU guidance is unavailable, industry-demonstrated yield and nutrient uptake information may be used.

Estimate realistic yield potentials or realistic yield goals using LGU procedures or based on historical yield or growth data, soil productivity information, climatic conditions, nutrient test results, level of management, and/or local research results considering comparable management and production conditions.

Nutrient application timing and placement
Consider the nutrient source, management and production system limitations, soil properties, weather conditions, drainage system, soil biology, and nutrient risk assessment to develop optimal timing of nutrients. For N, time the application as closely as practical with plant and crop uptake. For P, time planned surface application when runoff potential is low. Time the application of all nutrients to minimize potential for soil compaction.

For crop rotations or multiple crops grown in one year, do not apply additional P if it was already added in an amount sufficient to supply all crop nutrient needs.

To avoid salt damage, follow LGU recommendations for the timing, placement, and rate of applied N and K in starter fertilizer or follow industry practice recognized by the LGU.

Do not surface apply nutrients when there is a risk of runoff, including when—

- Soils are frozen.
- Soils are snow-covered.
- The top 2 inches of soil are saturated.

Exceptions for the above criteria related to surface-applied nutrients when there is a risk of runoff can be made when specified conditions are met and adequate conservation measures are installed to prevent the offsite delivery of nutrients. NRCS, in cooperation with the State water quality control authority, will define adequate treatment levels and specified conditions for applications of manure if soils are frozen and/or snow covered or the top 2 inches of soil are saturated. At a minimum, must consider the following site and management factors:

- Climate (long-term)
- Weather (short-term)
- Soil characteristics
- Slope

- Areas of concentrated flow
- Organic residue and living covers
- Amount and source of nutrients to be applied
- Setback distances to protect local water quality

Additional Criteria to Minimize Agricultural Nonpoint Source Pollution of Surface and Groundwater

Apply conservation practices to avoid nutrient loss and control and trap nutrients before they can leave the field(s) by surface, leaching, or subsurface drainage (e.g., tile, karst) when there is a significant risk of transport of nutrients.

Additional Criteria to Reduce the Risk of Potential Pathogens From Manure, Biosolids, or Compost Application From Reaching Surface and Groundwater

When applicable, follow proper biosecurity measures as provided in NRCS directives GM-130, Part 403, Subpart H, "Biosecurity Preparedness and Response."

Follow all applicable Federal, Tribal, State, and local laws and policies concerning the application of manure, biosolids, or compost in the production of fresh, edible crops.

Apply manure, biosolids, or compost with minimal soil disturbance or by injection into the soil unless it is being applied to an actively growing crop, a minimum of 30 percent residue exists, or there is a living cover that has a fibrous root system with 75 percent or more cover. Do not surface apply manure if a storm event is forecast within 24 hours.

Additional Criteria to Reduce Emissions of Objectionable Odors, PM and PM Precursors, and GHG and Ozone Precursors

To address air quality concerns caused by odor, N, sulfur, and particulate emissions; adjust the source, timing, amount, and placement of nutrients to reduce the negative impact of these emissions on the environment and human health.

Do not surface apply solid nutrient sources, including commercial fertilizers, manure, or organic by-products of similar dryness/density when there is a high probability that wind will blow the material and emissions offsite. Do not surface apply liquid nutrient sources when there is a high probability that wind will blow the liquid droplets applied from sprinklers or other applicable methods offsite.

Reduce the potential for volatilization by applying sources subject to volatilization during cooler, higher humidity conditions or by placement that minimizes vulnerability to volatilization.

Additional Criteria to Improve or Maintain Organic Matter

Design the plant or crop management systems so the soil conditioning index (SCI) organic matter subfactor is positive.

Apply manure, compost, or other organic nutrient sources at a rate and with minimal disturbance that will improve soil organic matter without exceeding acceptable risk of N or P loss.

For low residue plant or cropping systems, apply adequate nutrients to optimize plant or crop residue production to maintain or increase soil organic matter.

CONSIDERATIONS

General Considerations

Consider development of nutrient management plans by conservation management unit (CMU). A CMU is a field, group of fields, or other land units of the same land use and having similar treatment needs and planned management. A CMU is a grouping by the planner to simplify planning activities and facilitate development of conservation management systems. A CMU has definitive boundaries such as fencing, drainage, vegetation, topography, or soil lines.

Develop site-specific yield maps using a yield monitoring system, multispectral imagery or other methods. Use the data to further delineate low- and high-yield areas, or zones, and make the necessary management changes. Use variable rate nutrient application based on site-specific factor variability. See NRCS directive Agronomy Technical Note (TN) 190, AGR.3, "Precision Nutrient Management Planning."

Use the adaptive nutrient management learning process to improve nutrient use efficiency on farms as outlined in NRCS' national nutrient policy in GM-190, Part 402, "Nutrient Management." Consider using an adaptive approach to adjust nutrient rate, timing, form, and placement as soil biologic functions and soil organic matter changes over time. See NRCS directive Agronomy Technical Note (TN) 190, AGR.7, "Adaptive Nutrient Management Process."

When developing new nutrient management plans, consider using soil test information no older than 1 year rather than 2 years.

Develop a whole farm nutrient budget (nutrient mass balance), including all imported and exported nutrients. Imports may include feed, fertilizer, animals and bedding, while exports may include crop removal, animal products, animal sales, manure, and compost.

Modify animal feed diets to reduce the nutrient content of manure following guidance contained in Conservation Practice Standard (CPS) Feed Management (Code 592).

Provide a nutrient analysis of all nutrient source exports (manure or other materials).

Excessive levels of some nutrients can cause induced deficiencies of other nutrients, (e.g., high soil test P levels can result in zinc deficiency in corn).

Use soil tests, plant tissue analyses, and field observations to check for secondary plant nutrient deficiencies or toxicity that may impact plant growth or availability of the primary nutrients.

Do not apply K in situations where an excess (greater than soil test K recommendation) causes nutrient imbalances in crops or forages.

Use bioreactors and multistage drainage strategies to mitigate nutrient loss pathways, as applicable.

Use legume crops and cover crops to provide N through biological fixation. Cover crops with a carbon to nitrogen ratio below 20:1 can release a large amount of soluble N after being plowed or tilled into the soil when an actively growing crop is not present to take up nutrients, leading to increased risks of nitrate movement and nitrous oxide emissions. The nitrous oxide emissions often occur in high soil moisture conditions, such as when a legume cover crop is plowed down in fall or early spring. To avoid these losses, use grass-legume or grass-legume-forbs mixtures with a more balanced carbon to nitrogen ratio.

Use winter hardy grass cover crops to take up excess N after the cash crop growing season and promote contribution of the nitrogen to next plant or crop.

Use conservation practices that slow runoff, reduce erosion, and increase infiltration (e.g., filter strip, contour farming, or contour buffer strips).

Use application methods, timing, technologies or strategies to reduce the risk of nutrient movement or loss, such as—

- Split nutrient applications.
- Banded applications.
- Injection of nutrients below the soil surface.
- Incorporate surface-applied nutrient sources when precipitation capable of producing runoff or erosion is forecast within the time of a planned application.
- High-efficiency irrigation systems and technology.

- Enhanced efficiency fertilizers
  - Slow or controlled release fertilizers
  - Nitrification inhibitors
  - Urease inhibitors.
- Drainage water management.
- Tissue testing, chlorophyll meters, or real-time sensors.
- Pathogen management considerations.

When a recycled product (e.g., compost) is to be used as a nutrient source on food crops or as food for humans or animals, make sure that pathogen levels have been reduced to acceptable levels (reference the Food and Drug Administration's Food Safety Modernization Act at www.fda.gov/FSMA). When the recycled product has come from another farming operation, implement biosecurity measures and evaluate the risk of pathogen transfer that could cause plant or animal diseases.

Use manure treatment systems that reduce pathogen content from manure.

Implementing a soil health management system that reduces tillage or other soil disturbance, includes a diverse rotation of crops and cover crops, keeps roots growing throughout the year, and keeps the soils covered to reduce nutrient losses, and improves—

- Nutrient use efficiency, rooting depth, and availability of nutrients.
- Soil organic matter levels.
- Availability of nutrients from organic sources.
- Aggregate stability and soil structure.
- Infiltration, drainage, and aeration of the soil profile.
- Soil biological activity.
- Water use efficiency and available moisture.

Use targeted or prescribed livestock grazing to enhance nutrient cycling and improve soil nutrient cycling functions.

Elevated soil test P levels may lead to reduced mycorrhizal fungal associations and immobilize some micronutrients, such as iron, zinc, and copper.

Apply manure, compost, or other nutrient sources with minimal soil disturbance and at a rate that will improve soil organic matter without exceeding acceptable risk of N or P loss.

PLANS AND SPECIFICATIONS

In the nutrient management plan, document—

- Aerial site photograph(s), imagery, topography, or site map(s).
- Soil survey map of the site.
- Soil information including: soil type, surface texture, drainage class, permeability, available water capacity, depth to water table, restrictive features, and flooding and ponding frequency.
- Location of designated sensitive areas and the associated nutrient application restrictions and setbacks.
- Location of nearby residences, or other locations where humans may be present on a regular basis, that may be impacted if odors or PM are transported to those locations.
- Results of approved risk assessment tools for N, P, and erosion losses.
- Documentation establishing the application site presents a low risk for P transport to local water if P is applied in excess of crop requirement.

- Current and planned plant production sequence or crop rotation.
- All available test results (e.g. soil, water, compost, manure, organic by-product, and plant tissue sample analyses) upon which the nutrient budget and management plan are based.
- When soil P levels are increasing above an agronomic level, include a discussion of the risk associated with P accumulation and a proposed P draw-down strategy.
- Realistic yield goals for the crops (where applicable for developing the nutrient management plan).
- Nutrient recommendations for N, P, and K for the entire plant production sequence or crop rotation.
- Listing, quantification, application method and timing for all nutrient sources (including all enhanced efficiency fertilizer products) that are planned for use and documentation of all nutrient imports, exports, and onsite transfers.
- Guidance for implementation, operation and maintenance, and recordkeeping.

For variable rate nutrient management plans, also include—

- Geo-referenced field boundary and data collected that was processed and analyzed as a GIS layer or layers to generate nutrient or soil amendment recommendations per management zone. Must include site-specific yield maps using soils data, current soil test results, and a yield monitoring system with GPS receiver to correlate field location with yield.
- Nutrient recommendation guidance and recommendation equations used to convert the GIS base data layer or layers to a nutrient source material recommendation GIS layer or layers.
- After implementation, provide application records per management zone or as applied map within individual field boundaries (or electronic records) documenting source, timing, method, and rate of all nutrient or soil amendment applications.

If increases in soil P levels are expected above an agronomic level (i.e., when N-based rates are used), document—

- Soil P levels at which it is desirable to convert to P-based planning.
- A long-term strategy and proposed implementation timeline for soil test P drawdown from the production and harvesting of crops.
- Management activities or techniques used to reduce the potential for P transport and loss.
- For AFOs, a quantification of manure produced in excess of crop nutrient requirements.

OPERATION AND MAINTENANCE

Review or revise plans periodically to determine if adjustments or modifications are needed. At a minimum, review and revise plans as needed with each soil test cycle, changes in manure management, volume or analysis, plants and crops, or plant and crop management.

Monitor fields receiving animal manures and biosolids for the accumulation of heavy metals and P in accordance with LGU guidance and State law.

For animal feeding operation, significant changes in animal numbers, management, and feed management will necessitate additional manure analyses to establish a revised average nutrient content.

Calibrate application equipment to ensure accurate distribution of material at planned rates. For products too dangerous to calibrate, follow LGU or equipment manufacturer guidance on proper equipment design, plumbing, and maintenance.

Document the nutrient application rate. When the applied rate differs from the planned rate, provide appropriate documentation to explain the difference.

Protect workers from and avoid unnecessary contact with nutrient sources. Take extra caution when handling anhydrous ammonia or when managing organic wastes stored in unventilated tanks, impoundments, or other enclosures.

Use material generated from cleaning nutrient application equipment in an environmentally safe manner. Collect, store, or field apply excess material in an appropriate manner.

Recycle or dispose of nutrient containers in compliance with State and local guidelines or regulations.

Maintain records for at least 5 years to document plan implementation and maintenance. Records must include—

- All test results (soil, water, compost, manure, organic by-product, and plant tissue sample analyses) upon which the nutrient management plan is based.
- Listing and quantification of all nutrient sources (including all enhanced efficiency fertilizer products) that are planned for use and documentation of all nutrient imports, exports and onsite transfers.
- Date(s), method(s), and location(s) of all nutrient applications.
- Weather conditions and soil moisture at the time of application, elapsed time from manure application to rainfall or irrigation event(s).
- Plants and crops planted, planting and harvest dates, yields, nutrient analyses of harvested biomass, and plant or crop residues removed.
- Dates of plan review, name of reviewer, and recommended adjustments resulting from the review.

For variable rate nutrient management plans, also include—

- Maps identifying the variable application location, source, timing, amount, and placement of all plant and crop nutrients applied.
- GPS-based yield maps for crops where yields can be digitally collected.

REFERENCES

Association of American Plant Food Control Officials (AAPFCO). 2017. AAPFCO Official Publication no. 70. AAPFCO Inc., Little Rock, AR.

Follett, R.F. 2001. Nitrogen transformation and transport processes. In Nitrogen in the environment; sources, problems, and solutions, (eds.) R.F. Follett and J. Hatfield, pp. 17–44. Elsevier Science Publishers. The Netherlands. 520 pp.

Schepers, J.S., and W.R. Ruan, (eds.) 2008. Nitrogen in agricultural systems. Agron. Monogr. no. 49, American Society of Agronomy (ASA), Crop Science Society of America (CSSA), Soil Science Society of America (SSSA). Madison, WI.

Sims, J.T. (ed.) 2005. Phosphorus: Agriculture and the environment. Agron. Monogr. no. 46. ASA, CSSA, and SSSA, Madison, WI.

Stevenson, F.J. (ed.) 1982. Nitrogen in agricultural soils. Agron. Series 22. ASA, CSSA, and SSSA, Madison, WI.

USDA, NRCS. Agronomy Technical Note 3, Precision Nutrient Management Planning. 2010. Washington, DC. NRCS eDirectives under Technical Notes, Title 190 (https://policy.nrcs.usda.gov/).

USDA, NRCS. Agronomy Technical Note 7, Adaptive Nutrient Management Process. 2013. Washington, DC. NRCS eDirectives under Technical Notes, Title 190 (https://policy.nrcs.usda.gov/).

USDA, NRCS. Nutrient Management Technical Note 7, Reducing Risk of E. coli O157:H7. 2007. Washington, DC. NRCS eDirectives under Technical Notes, Title 190 (https://policy.nrcs.usda.gov/).

USDA, NRCS. Title 190, General Manual, (GM), Part 402, Nutrient Management. 2011. Washington, DC. NRCS eDirectives under General Manual, Title 190 (https://policy.nrcs.usda.gov/).

USDA, NRCS. Title 190, National Instruction (NI), Part 313, Nutrient Management Policy Implementation. 2017. Washington, DC. NRCS eDirectives under National Instruction, Title 190 (https://policy.nrcs.usda.gov/).

Journal of Great Lakes Re arch 45 (2019) 1162-1170

Contents lists available at ScienceDirect

Journal of Great Lakes Research

journal homepage: www.elsevier.com/locate/ijglr

Manure Management at Ohio Confined Animal Feeding Facilities in the Maumee River Watershed

Jeffrey B. Kast ,a.b• , Colleen M. Long e, Rebecca Logsdon Muenich d . Jay F. Martin a, e , Marga ret M. Kalcic a

ʼ Deparrment of Food. Agriculcural and Biological Engineerin&', The Ohio State UnlvtrSiry, S90 Woody Hayes Dr. Columbus.OH 43210, United Stoces
ʼ Environmencal Science Graduate Program, The Ohio State Unlverslry, 174 West 18th Ave..Columbus, OH 43210, United Stores
'Graham Sustolnablllry lnsdwte, University of Mlchij:an.214 S. StareSI.. Ann Arbor, MI 481OS, United Scares
• School of Susralnable Engineering and the Bulle Environment, Arizona Scare University, 660 S. College Ave.. Tempe, AZ 85281, United Stotes
• Th• Sustainabfliry /nsrfrute at Ohio State, 174 West 18th Ave.. Columbus, OH 432IO, United States

ARTICLE INFO

Article hf<tory:
Received 9 November 2018
Accepted 2 September 2019
Available on line 2 November 2019
Communicated by R. Michael McKay

Keywords:
CAFO
M•nur
Nutrients
Wme MaDAgement
Maumee River wate rshed

ABSTRACT

In 2015, 48 permitted Confined Animal Feeding Operations (CAFOs) housed approximately 90% of poultry and 20% of swine and cattle within the Ohio portion of the Maumee River watershed. Recently, concerns about the impact CAFOs may have on nutrient loading in the watershed have been raised. In this study. we used manure management plans and inspection reports obtained from the Ohio Department of Agriculture Division of Livestock Environmental Permitting (ODA-DLEP) to assess how these CAFOs managed their manure for the years 2014 and 2015. A majority of liquid manure was applied between April and October, closely matching the amount of liquid manure planned to be applied during this period. Approximately 79% of the acres under control of the CAFOs that received manure had Bray Pl soil test phosphorus values below 50 ppm. The average distance between a swine CAFO's livestock holding barn to the fields they control that can receive manure was 1.43 miles while for cattle CAFOs this distance was 1.91 miles. Approximately 78% of manure phosphorus generated on CAFOs was planned to be transferred through Distribution and Utilization. a process in which owner.;hip of manure changes hands, including virtually all solid poultry manure phosphorus. While publicly available data show that. in general, CAFOs in the region are adhering to their state-approved permits. a knowledge gap regarding the management or approximately 80% of manure phosphorus exists due to manure transferred through Distribution and Utilization and manure produced from non-permitted livestock operations.

Introduction

Coastal eutrophication is occurring throughout the world with impacts such as the development of harmful algal blooms (HABs : Paerl et al., 2016; Li et al., 2014). HABs can cause substantial economic losses (Hoagland et al.. 2002), impaired water quality (Brooks et al.. 2016; Davis er al., 2019), public health risks (Grattan er al.. 2016). and ecosystem degradation (Sukenik et al., 2015). Lake Erie is a notable example where HABs have affected drinking water supplies and had measurable economic impacts (Carmichael and Boyer, 2016; Wolf et al.. 2017). In freshwater systems, such as Lake Erie, phosphorus (P) is typically the limiting nutrient controlling the production and size of HABs (Correll, 1999; Blomqvist et al.. 2004; Kane et al., 2014 ; Schindler et al.. 2016). In response to eutrophication and other water quality concerns in the 1960s and 1970s. the United States and Canada agreed to the 1972 Great Lakes Water Quality Agreement (GLWQA), which successfully reduced the P loading into Lake Erie, largely by decreasing point source discharges (Depinto et al., 1986). Despite this improvement in Lake Erie's water quality and resulting reduction in HABs. discharges of Dissolved Reactive Phosphorus (DRP). primarily from nonpoint sources of pollution, have steadily increased since the late 1990s (Scavia et al.. 2014) leading to a reemergence of HABs. In response. a revised version of the GLWQA now calls for a 40% reduction in both total phosphorus (TP) and DRP entering the lake by 2025 (USEPA, 2018).

The Maumee River watershed, the largest of Lake Erie's tributary watersheds by size, contributes the largest TP and second largest DRP loads to Lake Erie (Maccoux et al., 2016). Recent mass-balance studies have estimated that most of the TP load leaving the watershed (85-88%) originates from nonpoint sources

• Co1Tesponding author at: Department or Food. Agricultural and Biologiul Engineering, The Ohio Sme University, 590 Woody Hayes Dr., Columbus.OH 43210, United States.
E-mail addr,ss: kast.140o su.edu, UB.Kast

https:/ldol.o,a:/I0.1016/j.Jglr.2019.09.015

including farm fertilizers and manures (Ohio EPA. 2016: Scavia et al., 2016; Ohio EPA. 2018). Studies have also estimated that approximately 19-23% of the P input in the watershed is from manure (International Joint Commission, 2018: Scavia *et* al., 2017). The number of large-scale livestock operations, including permitted Concentrated Animal Feeding Operations (CAFOs). has increased in the United States (USGAO. 2008: Hribar, 2010 lead - ing to higher spatial concentrations of animals (Copeland, 201O: Key et al., 2011). Mirroring national trends, the three states in the Maumee River watershed (Ohio, Indiana, and Michigan) have seen an increase in the number of large-scale livestock facilities (Kee ney, 2008: USDA, 2012 : Lenhardt and Ogneva- Himmelberger , 2013). In watersheds across the United States, including the Maumee River watershed, there is uncertainty regarding nutrient delivery and water qualitydownstream of these operations where large amounts of manure are produced, stored. and applied to cropland.

Confined Animal Feeding Operations contain populations greater than the livestock-equivalent of 1000Animal Units and confine animals for more than 45 days per year. Additionally, a CAFO is defined as any animal feeding operation that discharges manure or wastewater directly into surface water , or is designated by a permitting authority as an operation that contributes significant amounts of pollutants, if the above two conditions are not met (USEPA,2008). States can further refine this definition and set their own reporting requirements, which can lead to different naming conventions, different state agencies directing regulations and inspections of facilities, and different reporting requirements. All of this leads to a lack of common, publicly available data across states. which makes it difficult to identify where. when. and how manure generated from CAFOs is app lied across watersheds.

Confined Animal Feeding Operations can use nutrients in livestock manure in multiple ways including energy production and land application. A CAFO can apply manure on their own fields, engage in land use agreements with nearby landowners. or use a manure broker to apply on fields at further distances. If cro pland requiring manure application is not available, a CAFO may leave manure in storage or over-apply it on cropland which may not need fertilization (Ribaudo et al .. 2003) which can lead to elevated nutrient losses from those fields (Aronsson et al.. 2014). In the Maumee River wat ershed, application of manure from CAFOs is legally limited to fields with soil phosphorus levels <150 ppm (Bray Pl) in Michigan (MDEQ, 2015) and <200 ppm (Bray Pl or Mehlich-111) in Indiana (IDEM. 2014). Application of manure from CAFOs in OH is recommend ed not to occuron fields with soil phosphorus levels greater than 150 ppm(Bray Pl): however , if a field's potential for phosphorus movement {P-Jndex) is low enough it may be applied (ORÇ2014).

The primary objective of this study was to characterize manure generation as well a.s manure application timing, incorporation, and distribution from the 48 permitted livestock operatio ns located within the Ohio portion of the Maumee River wa te rshed. In pani cular, we seek to understand when manure was applied; where it was applied in relation to the livestock facility; and field soil phosphorus levels. Compiling this information will improve our understanding of manu re management within the waters hed and provide insight as to whether practices related to application timing and placement. two elements of 4Rs of Nutri ent Stewardship (Vollmer-Sanders et al., 2016). are being used to minimize nutrient runoff.

Although livestock operations in Ohio permitted by the Ohio Depan ment of Agric ulture-Division of Livestock Environmental Permitting (ODA-DLEP)are called Confined Animal Feeding Facilities (CAFFs) by the Stat e of Ohio, they will be referred to as CAFOs throughout thismanuscrip,tas that is the morecommon term used nationwide.

Methods

CAFO data-permits and manure management plans

Ohio requires all CAFOs (livestock operations permitted by ODADLEP) to submit a manure management plan (MMP) to ODA-DLEPwhen a facility submits a Permit to Operate (PTO). We reviewed the most recent permits and MMPs for all operational CAFOs (n • 48) located in th e Ohio ponion of the Maumee River wa ters hed in 2015 (Fig. 1). Manure management plans and l'TOs submitted between 2012 and 2016 as well as Permits to Install (PTI) submitted in 2015 for these CAFOs were obtained through a Freedom of Informat ion Act (FOIA) request and other informal requests to ODA-DLEP {Ohio Environmental Council, 2017; ODA-DLEP pe rso nal communication, 2017). Manure management plans included information on the planned timing and method of manure ap plication. t he different manure storage structures a CAFO use s and their manure composition analyses, the amount of manure nutrients planned to be applied to fields controlled by a CAFO and fields controlled by others, and current (as of time the MMP was written) as well as historical soil phosphorus tests for fields controlled by a CAFO. Manure management plans also contained information regarding the maximum number of animals a CAFO may contain as well as an estimate of the annual volume of manure produced. Confined Animal Feeding Operations are not required to repon all information related to manure applications originating from their operation. These reponing rules particularly impact manure managed under Distribution and Utilization as information on land (soil phosphorus levels and field maps) and manure application (planned timing and manure application met hod) are not required to be reported. Sixteen CAFOs voluntarily provided this information and, when available, these data were analyzed.

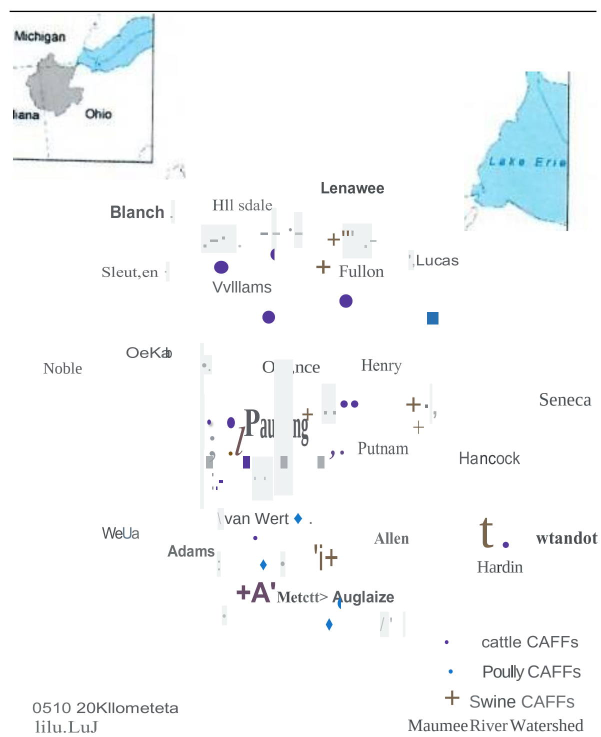


Fig. 1. Locations of catt le, poultry, and swine CAFOs loca ted within the Mau mee River watershed and in Ohio counties that were Included in the study.

Distribution and Utilization refers to CAFOs distributing manure to farm operators and land not under control of the permitted facility. Because of missing data and data reported in non-standard formats multiple CAFOs were excluded from the analyses (Electronic Supplemental Material (ESM) Table S1).

CAFO data-inspection reports

Ohio requires that all CAFOs be annually inspected by ODA-DLEP. We obtained Inspection Reports from 2014 and 2015 for 18 cattle and 18 swine CAFOs through a FOIA request and other informal requests to ODA-DLEP (Ohio Environmental Council, 2017; ODA-DLEP personal communication, 2017). Inspection Reports include soil test phosphorus (STP) data from the fields under control of a CAFO in which manure was applied during the year as well as the timing and volume of liquid manure and mass of solid manure applied on the fields. Due to Inspection Reports not being provided at the time of this study multiple swine and cattle CAFOs were omitted from this analysis (ESM Table S2).

Calculations and assumptions for livestock populations

The 48 CAFOs were characterized by their primary livestock constituency: cattle, poultry, or swine, based on the total number of animals each CAFO indicated they were permitted to house in their most recent PTO submitted prior to 2017 or PTI submitted prior to 2016. Cattle CAFOs were further characterized by the type of cattle permitted to be housed on the facility, dairy or beef. Livestock numbers from CAFO reports and the NASS 2012 Census were used to estimate the number of livestock not housed in CAFOs within the boundary of the study area (ESM Appendix S1 County-Level Livestock Estimates and Manure Phosphorus Produced and ESM and Tables S3 - S6).

Calculations and assumptions for manure application and distribution

The planned amount (in either volume or mass) of manure to be applied during each application and incorporation schedule was calculated by multiplying the planned percent of yearly manure applied by the annual volume of mass of manure removed from each storage structure. Application and incorporation schedules indicate either the range of months or the month in which manure application was planned as well as the number of days after application when the manure was planned to be incorporated into the soil. Volumes and masses were summed for each CAFO to calculate the total amount of liquid and solid manure that was planned for each of the reported manure application timing and incorporation event.

The total monthly volume of liquid manure and mass of solid manure applied to fields in 2014 and 2015 were summed for each CAFO. Respectively, 19 and 22 fields in 2014 and 2015 were reported as receiving a total amount of manure applied over several months rather than on explicit days. In these cases, the total amount was divided equally over the months.

Maps of fields a CAFO can use to apply manure were digitized in ArcGIS. Distances were calculated from the location of the facility's animal housing barn to the centroid of all the digitized fields for each CAFO. The average mean, median, minimum, and maximum distance were calculated for all fields under control of each swine and cattle CAFOs. Approximately 40 fields from five swine CAFOs (13% of all fields listed in swine CAFO MMPs) and 136 fields from seven cattle CAFOs (33% of all fields listed in cattle CAFO MMPs) were excluded from this analysis because they either were noted as D&U fields, were illegible in the MMPs, or were not explicitly identified as fields under control of the CAFO.

The total amount of manure nitrogen and phosphorus (reported as P_2O_5) planned to be applied on fields controlled by and fields not controlled by each CAFO were reported in each facility's MMP. Manure nutrients planned to be applied on fields controlled by and fields not controlled by a CAFO were summed for each livestock-designated CAFO.

Calculations and assumptions for manure nutrient compositions

Manure composition analyses reported for 2007 to 2015 were analyzed by manure storage structure for CAFOs who provided manure nutrient analyses and who were located within an Ohio county in the Maumee River watershed regardless of if the facility was located within the watershed (n = 97: data from 97 CAFOs within and outside the boundaries of the Maumee River watershed were analyzed). Permits to Operate were used to link manure storage structures to their corresponding manure nutrient composition analysis reported when naming conventions for the storage structure was not consistent throughout an MMP. Student T-tests assuming unequal variance were used to determine if total nitrogen (TN) and phosphorus (as P_2O_5) compositions of manure stored within different swine manure storage structures and between the most common manure storage structures for dairy, swine, and poultry manure significantly differed.

Calculations and assumptions for field soil test phosphorus

Soil test phosphorus (STP) values (Bray P1) were recorded for each field under control of a CAFO and, when provided, for fields not under control of a CAFO as distinguished in the MMPs. An area-weighted STP value was calculated for fields in which multiple soil phosphorus tests were conducted. Soil test phosphorus values reported in Mehlich-III were converted to Bray P1 (Watson and Mullen, 2007), and STP values reported in pounds per acre were converted to parts per million (Liu et al., 2013). Soil test phosphorus values of the fields which received manure in 2014 and 2015 were recorded along with the acreage represented in the corresponding Inspection Reports. If inspection reports did not include the STP testing method, Bray P1 or Mehlich-III, the STP testing method detailed in the CAFO's MMP was used. When multiple STP were provided for a field receiving manure (16 cases in 2014 and 20 cases in 2015) the average STP value was used.

Results

County level livestock populations in the watershed

In 2015, 18 dairy, 1 beef, 23 swine, and 6 poultry livestock operations were permitted as CAFOs within the watershed boundary (Fig. 1). Approximately 605,000 swine (242,616 Animal Units), 164,000 cattle (235,171.4 Animal Units), and 3,500,000 poultry (35,312.1 Animal Units) were in the Ohio portion of the watershed in 2012, Table 1. Approximately 20% of swine and cattle and 90% of poultry were housed in CAFOs.

Planned application and incorporation schedules for liquid and solid manure

In 2015, over 324,000,000 gallons of liquid manure from 26 of the 48 CAFOs and approximately 36,000 tons of solid manure from 18 of the 48 CAFOs were planned to be applied to fields under control of CAFOs. These manure volumes and masses represented 65% and 19% of all liquid and solid manure, respectively, planned to be removed from the storage structures of the 48 CAFOs in the study area. Of the remaining liquid manure planned to be removed from

Tab!@1

li vest ock population estim•tes for cattle, poultry, and swine within the Ohio boundaries or the Maumee River watershed. Animal units arc estimated assuming all swine are breeding hogs, all cattle are milk cows, and all poultry arc chicken l.iyers (Kellogg.2002).

	Non--CAFO livestock population	CAFO livestock population	Tot.al livestock population
Swine	471,158(78%) 18, 8 463.2 Animal Unil5	135382 (22%) 54.152.8Animal Unil5	606,540 242,616 Animal Units
Cattle	123,277 (**751**) 176,1lOAnimal Units	41,343(25S) 59,061.4 AnimalUnits	164,620 235,171.4 Animal Unlu
Poultry	225,575 (**61**) **2,255.8 Animal** Units	33 05,625 (94S) 33,056.3 Animal Units	3,53i,200 35.3121 Animal Units

CAFO manure storage structures, 4% (from one CAFO) was planned to be transferred through D&U, 14 % (from seven CAFOs) prov ided information not consistent with forms provided by ODADLEP, and **4%**(from two CAFOs} provided information for onlya portion of their manure storage structures. Of the remaining solid manure planned to be removed from CAFO manure **storage** structures, 72% (from 17 CAFOs) was planned to be transferred through D&U, 9% (from six CAFOs} provided information not consistent with forms provided by ODA-DLEP. Five CAFOs indicated no solid manure was planned ro be removed from their manure storage structures.

Approximately 59% of the liquid manure applied on CAFO controlled fields with data available was planned to be applied between July and Octoberand 46% was planned to be incorporated into the soil within one day of application. Approximately 59% of the solid manure applied on CAFO cont rolle d fields with data avail- able was planned to be applied between April and July with a majority (82%} being planned to be incorporated into the soil within one day of application .

Relationship between manur e nutrient *compositionand* manure storage *structure*

Liquid swine manure stored in anaerobic treatment lagoons was found to have significantly lower phosphorus *($p < 0.0001$)* and total nitrogen ($p < 0.0001$) compositions than liquid swine manure stored in concrete pits. Liquid dairy manure stored in earthen ponds was found to have significantly less phosphorus and nitrogen than liquid swine manure stored in concrete pits ($p < 0.0001$} and significantly more phosphorus and nitrogen than liquid swine manure stored in anaerobic treatment lagoons *($p < 0.0001$)*.

Median phosphorus contents for liquid swine manure stored in anaerobic treatment lagoons and concrete pits, the two primary storage structures found for swine manure among the CAFOs, contained 0.07 g P_20_5/L (0.6 lbs P_20_5 per1000 gallons} and 1.96 g $P20_5$/L(16.4 lbs P 0 s per 100 0 gallon s) , respe ct ively. Median phosphorus concentrations for liquid dairy manure stored in earthen ponds was 0.56g P_20 s/L (4.7 lbs P_20_5 per 1000 gallons}. Median concentrations for total nitrogen among the two swine storage structures were 0.60 g N/L (5.0lbs N per1000gallons) for anaerobic treatment lagoons and 4.39 g N/L (36 .7 lbs N per 1000 gallons) for concrete pits. Median total nitrogen concentrations for the dairy storage structure was 1.40g N/L (11.7 lbs N per 1000 gallons; Table 2).

Manure *transfer through* distribution *and* utilization

Nutrien ts from manure produced on CAFOs and transferred to farms and fields controlled byother operators through D&U varied between the primary livestock found at each CAFO. as shown in Table 3. The proportion of CAFOs transferring manure to others also varied between the primary animal designations for each CAFO. A majority of cattle CAFOs (78%) and half of swine CAFOs (50 %) planned to transfer some of their manure with 7% of cattle CAFOs planning to transfer all of their manure and 50% of swine CAFOs planning on transferring noneof their manure. All five poultry CAFOs planned to transfer 100% of their manure nutrients.

Manure application surrounding *a CAFO*

Twenty-three swine CAFOs and 15 ca ttle CAFOs provided usable field maps and distinguished the fields under their control that can receive manure in their MMPs. The average mean, median, minimum,and maximum distances for all fields able to receive manure controlled by each cattle CAFO from the livestock holding barn was found to be larger than those of swine CAFOs. Table 4. Approximately half (53%) of swine CAFOs had a mean distance between the fields they control to the location of the livestock holding barn less than one mile while17% had a mean distance greater than two miles{ESMTible S7). An equal fraction of cattle CAFOs (40%) had a mean distance between the fields they control to the location of the livestock barn less than one mile and greater than two miles (ESM Table S8). Although the largest maximum distance manure could travel to a field under control of a CAFO was for a swine CAFO, (ESM Table S7). cattle CAFOs were found to generally have larger maximum distances manure could travel between the livestock holding barn and the fields they control than swine CAFOs (Tab le 4).

Soil *test phosphorus (SfP)* on *fields able* to *receive*manure

Thirty-nine CAFOs provided STP results between May 2007 and January 2017 for fields under their control totaling over 32,000 acres (ESM Tables S9 and S10). A m.1jority of acres (79%) were found to have a Bray Pl STP value of 50 ppm or less while 3% of acres were found to have a Bray Pl STP values greater than 100 ppm (Fig. 2). Fifteen CAFOs provided STP res ults between November 2004 and August 2014 for fields not under their control that are used in D&U totaling over 37,000 acres. Over half of the acres (57%} were found to have a Bray P1 STP value of 50 ppm or less while 5% of acres had Bray Pl STP value greater than 100 ppm (Fig. 2). The majority of the O&U acres (78%} provided estimated STP va lue s where the date of the soil test wasexcluded. Six CAFOs reported transferring all or most of their manure offsite through D&U and did not provide soil test results for these fields.

Planned versus actual manure management

Planned applications of liquid manure and actual liquid manure applications in 2014 and 2015 were more similar than planned application of solid manure and actual solid manure applications in 2014 and 2015 (Fig. 3). The total amount of liquid and solid manure applied in 2014 and 2015 did not exceed the total planned amount of manure removed from the storage structures in the MMPs suggesting that CAFOs may not have fully emptied their storage structures each year, maintained less livestock then they

T•ble 2

Nutrient contents or swine, dairy, and poultry manure 0111:anlzed by primary manure storage structures reported in CAFO MMPs and PTOs. The number or nutrient composition analyses (n: sample size for phosphorus, nitroaen) is shown with each storage structure. Swine and Dairy manure are liquid manure and reponed In lbs/1000 gallons. Poultry manure is solid manure and reponed in lbs/ton. Leners represent • sm lstically signiftcant difference at P !, O.OS betwe e n manure nutrient compositions.

	Swine		Dairy	Poult ry
	Concrete pit (n • 117/118)	Anaerobic treatment lagoon (n • 45/44)	Earthen pond (n • 243/229)	Barn (n • 357/355)
Median P:,Os	16.4	0.6	4.7	69.2
Mean P,.0s	19.o"	0.7[1]	6.1<	70.6°
Std. Dev. P_2O_5	16.S	0.7	5,3	19,8
Med ia n TN	36.7	5,0	11.7	62.3
Mean TN	34.9•	5.0"	13,7'	68.1•
Std. Dev. TN	13.8	1.9	9.1	34 6

Table 3

Total amount and percent of nutrients found in CAFO-genc rated manure that was planned 10 be transferred to fields not under CAR> control through D&U.

CAFO type	Total nitrogen (lbs)	P20s (lbs)
Swine (n • 22)	364,655(21:1:)	384,236(34:1:)
Canle (n • 18)	5 ,137,6 57 (7U,)	1,752,895 (63%)
Poultry (n • 5)	5,35J.468(1OO,;J	4.454 ,902(I 00%)
Total	10,853,780	6.592.033

were permitted to house, or over-es timated manure volumes by using conservative calculations.

A majority of the iden tified fields that received CAFO manu re in 2014 (64%) and 2015 (69%) had Bray Pl STP values below SO ppm (Fig. 4). Confined Animal Feeding Facilities did not use all their available acreage when applying manure each year. Manure was app lied to approximately 55% of the available acreage in 2014 and 61% in 2015 . Over the two-year period of study, only one CAFO reported applying manure on a field with a Bray Pl STP level above 150ppm.

Discussion

Knowledge gap of manure generation and application

Understanding where manure is applied, when it is applied, and how it is applied can furthe r the understandi ng of manu re management within the region and infonn conservation strategies. There have been numerous knowledge gaps regarding how manure is managed in the Maumee River watershed , and this study has presented data to address some, but not all, of those unknowns. One such knowledge gap was the distance manure is applied from the livestock holding barn where it is produced, and results from this study indicate that these distances may be dependent on the type of livestock or type of manure in a CAFO. All poultry CAFOs transported their solid poultry litter off-site while cattle and swine CAFOs applied manure on-site and transported manure off-sit e (Table 2). Our results align with other studies which have shown non-pouloy manure does not travel far from its source. Lory et al. (2001) found that swine manure could travel as far as 5.2 miles from its source and Long et al. (2018) found that 70% of man - ure applied from CAFOs was within 5 miles of the facility. Furt hermore, Long et al. (2018) found that 51% of field acreage used for manure application in the northern part of the Maumee River wat ers hed. from primarily dairy CAFOs in Michigan, was within 3.1 miles of the facilities. Our results show that swine manure is likely applied closer to its source than cattle manure, including dairy manure, when it is not transferred to others through D&U (Electronic Supplemental Materia l (ESM) - Tables S7 and S8).

When manure is transferred through D&U there is less certainty of the locations where the manure is land applied. However, this manure must be handled by a Certified Livest oc k Manager (CLM) who is trained by ODA or another certified fertilizer applicator, so it would be possible to learn more about manure management through surveying CLMs and further analyzing facility Ins pectio n Reports.

Iden tifying periods in which manure is applied and when it is incorporated into the soil is important in determining the amount of nutrients available to crops and the potential impact on downstream water quality following application (Gowda et al., 2008: Hooda et al.. 2000: Watts et al., 2011). The risk of nutrient runoff in surface water has been found to decrease when avoiding either application in winter or application immediately preceding a rain event (Vadas et al. 2017) and when nutrients are incorporated into the soil rather than broadcasted on the soil surface (Gildow et al.. 2016: Williams et al., 2018). Liu et al. (2017) found that simulated manure application in the spring resulted in up to 16% less TP discharge and 40% less DRP discharge than manure application in the fall and wint er.

In Ohio. the 4R's of Nutrie nt Stewardship address these two factors by recommending nutrients be placed at the right time(when nutrients are applied) and at the right place (where in the soil pro- file nut rients are applied; Vollmer-Sanders et al., 2016). Our results found wide-ranging planned application schedules, spanning all four seasons of the year. A majority of liquid manure was planned to be applied between July and October.This pattern was found to be similar to the liquid manure applied in 2014 and 2015. A major- ity of solid manure was planned to be applied between April and June. Large portions of solid manure applied in 2014 and 2015 were applied between April and June as well as in July and August. CAFOs largely avoided winter applications of manure that result in higher nutrient losses. A majority of the manure from CAFOs was applied was between April and October (Fig.3 This indicates that CAFOs applied manure during months that minimize time manure would be on a fallow field and maximize opportunity for nutrient uptake by crops. Results show that a majority of the solid manure was planned to be incorporated into the soil within one day of application and approximately 50% o f th e liquid manure was planned to be incorporated within one day of application. Smith et al. (2007) found that plots experiencing their first runoff even t

Table 4

Avcr e distances from the livestockb.lm *to* the fields under contr0I of swine and catt le CAFOs th .at can receive manure .

	Mean (miles)	**Median** (miles)	Minimum (miles)	Maximum (miles)
Swine (n • 23)	1.43	1.67	0.19	2.86
CAttle(n• 15)	1.91	1.76	0.28	4.23

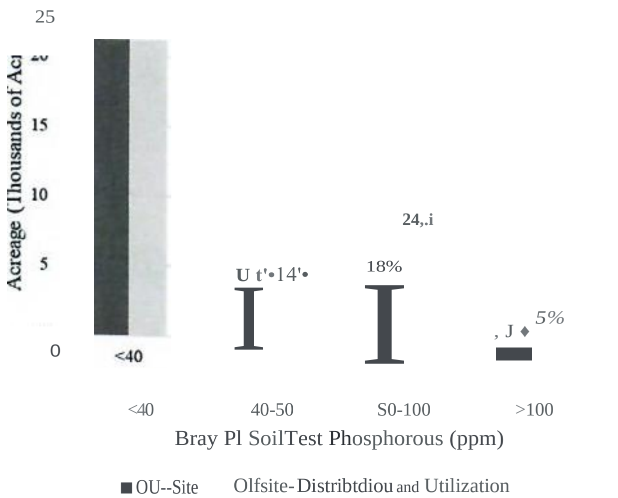


Fig.2. Br.iy Pl STP threshol ds o r flelds •v• llable ror land applicorion underconrrol ora CAFO•nd thosenot underaCAFO-s control. STP valu es or 40 ppm Bray Pl is the lowe r limit in which the Tri-State Fertilizer recommendations ror com •nd soybe•ns recommend no additional phosphorus ren iliu r. For Wheat and alralra. 50 ppm Bray Pl is the lower limit (Vitosh et al.. 1995). Values above each bu represent the fraction of acres within each STP range and within the two groups: on-site and offsite.

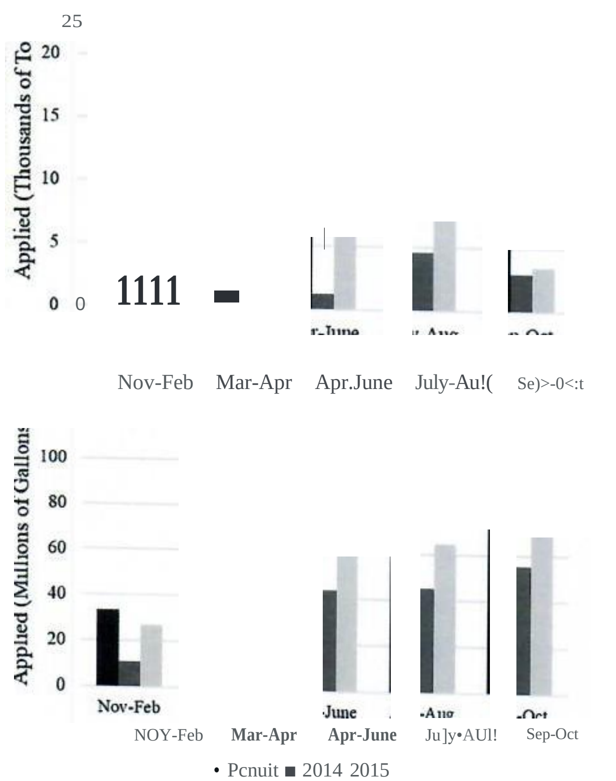
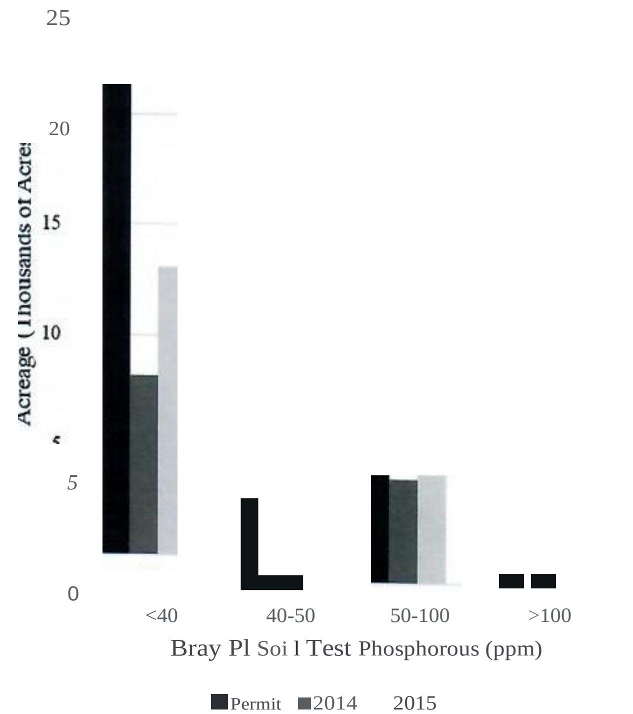


Fig. 3. Planned manure application in permits compar<d ro actual manure application methods employed In 2014 •nd 2015.

one day after surface swine manure application had surface water discharged with higher concentrationsof soluble phosphorus than plots experiencing their first runoff event further after the swine manure application. These results indicate for the manure applied to CAFO controlled fields, CAFOs are employing manure management practices which support downstream water quality by limiting the amount of time a rain event could impact nutrient



Ftg.4. Bray Pl soil phosphorus test levels orfields under control or CAFOs receivin g manure in 2014 and 2015 compared to soil phosphorous test levels ofRelds under control or CAFOs a ble to be land applied as reported in permits. STP value s or 40 ppm Bray Pl is the lower limitin which the Tri-State Fertili,er recommendations ror com and soybeans recommend no additional phosphorus fertilizer. For wheat and alfalfa, SO ppm Bray Pl is the lowe r limit (V11oshet al. 1995).

discharge, as well as getting the nutrients to the agricultural fields at times to support crop uptake.

A crit ical knowledge gap for manure management in Ohio is the management of manure that is distributed offsite through O&U. Manure application methods as well as field locations and their

STP levels need to be maintained by the CLM or certified fertilizer applicator but do not need to be reported to the state. Approximately 75% of all manure nutrients generated on CAFOs in northwest Ohio is planned to be land applied on fields in which CAFOs use the D&U process to transfer their manure, including all six poultry CAFOs that wereanalyzed. This results in manure management from the majority of CAFO-produced manure being unavailable for analysis and largely unknown.

Manure composition and *storage* structure

Manure nutrient composition has been found to vary within manure storage structures for both swine (Waskom and Davis. 1994; Honeyman , 1996 ; Lorimor and Kohl, 1998) and dairy (Chastain and Camberato, 2004: MWPS-18. 2004) making it difficult to detennine a manure's typical nutrient composition. The differences among the nutrient compositions stored in a variety of structures (Table2) show the importance of storage conditions in addition to the livestock diet (Nah m, 2002). Our results show the importance of using local manure analyses to estimate the amount of manure nutrients applied (Long et al., 2018) as well as the challenges of calculating an average nutrient composition of manure. Although typical manure nutrient compositions can be used to estima te the amount of nutrients land applied during manure applications (Bentley et al., 2016), local manure analyses will yield more accurate estimates. Liquid swine manurestored in concrete pits, the most common storage structure utilized by swine CAFOs in the study, was found to have, on average, over seven-fold the amount of P20 s and nearly five-fold the amount of total nitrogen as liquid swine manure treated in anaerobic treatment lagoons. It should be noted that this result was found when disregarding the phosphorus in sludge accumulations in both concrete pit and lagoon storage structures.

Limirations *of information regarding manure management*

Analyses on manure management in this study do not include manure produced from livestock operations with amounts of animals below the legal threshold to be considered a CAFO in Ohio.

Using methods described in this manuscript and expanding the analysis to include livestock operations in Michigan and Indiana, approximately 71% of swine, 76% of cattle, and 15% of poultry within the Maumee River wa tershed and proportional volumes of manure were housed in and generated on non-permitted operations. In the Ohio portion of the watershed, non-permitted operations account for approximately 80% of swine and cattle, and 6%of poultry (Table 1), and thus similar percentages of manure generated. Manure managed by non-permitted operations differs from manure managed by CAFOs in a number of ways in Ohio.These discrepancies include (1) ODA-DLEP does not regulate these facilities or manure produced from these facilities, (2) a Manure Management Plan is not required to be submitted to the state, and (3) manure does not need to be handled by a CLM who is trained by ODA However, non-permitted operations, like CAFOs, are subject to other regulation s enacted by the state. One such regulation is Ohio Senate Bill 1 which bans manure and fertilizer application on frozen soil or when there is a 50% chance of exceeding 1/2 in. of rain within a 24-h period, unless it is applied on a growing crop, injected intothe soil,or incorporated intothe soil within 24-hof its application. If livestock operations of all sizes follow these rules, nutrient runoff from manure applications will be limited. as previous work has shown the effectiveness of these practices (Schuster et al., 2017; Vadas et al., 2017). While the publicly available data used in this study show that CAFOs are generally adhering to guidelines and regulations when applying to fields they control, overall knowledge of manure management and impact is limited by key unknowns. The situation for livestock operations below CAFO thresholds, combined with aforementioned limitation related to D&U, results in an overall knowledge gap for management of approximately 80% of the manure produced from swine and cattleand 95% of solid manure produced from poultry and produced in the Ohio portion of the Maumee River watershed.

Approach to calculating livestock populations and manure *generated*

A recent report by the Environmental Working Group (EWG) analyzed livestock populations and manure produced in the Maumee River watershed (EWG, 2019). While this report had similar findings of unknown management of a majority of manure in the watershed, differences in estimates of the number of livestock and amount of manure produced exist between the two studies. These differences are likely due to variations in methods and time-periods analyzed. The 2012 Agricultural Census and CAFOs' Annual Reports and Inspection Reports up to 2015 were used to derive livestock pcpulations in this study (ESM Sl).This approach allowed for the estimation of the number of livestock held in permitted operations in the region as well as the number of livestock in non-permitted operations within each county of the region. Similar to thisstudy, EWG (20 19) used CAFO permit data to aid in their livestock population estimates. However, additional livestock barns were located with aerial imagery and assigned animal counts based on the square-footage and other physical charac ter istics of the mapped barns (EWG, 2019). This study limits its investigation of livestock populations to 2015 bounded by county-level livestock estimates of the 2012 Agricultural Census and of livestock populations detailed in CAFOs' Annual Reports and Inspection Repcrts. EWG (2019) investigates livestock populations up to 2018, not bounded by the 2012 Agricultural Census, leading to a larger estimate of swine and poultry in the Ohio portion of the watershed (swine: +30%, poultry: +544%, cattle [dairy and beef): - 54%.) One explanation for the difference in poultry estimates, in addition to increases in the inventory of poultry from the 2012 Agriculture Census (+16% in Ohio; USDA, 2017), is that EWG (2019) includes turkeys in their analyses while this study focuses on layers, broilers, and pullets. In 2012, turkeys accounted for approximately 10% of the poultry population within the region (EW,G 2019).

Another explanation for lower animal numbers in this study is that livestock were subtracted from county totals that were within a county partially residing in the watershed but located in the portion of the county outside the watershed.

In addition to differences in estimating livestock populations, this study and EWG (2019) use different methods in estimating the amount of manure phosphorus produced by livestock. Methods described by Ruddy et al. (2006) were used to derive the amount of phosphorus in manure in this study when estimating the amount of manure nutrients produced throughout the watershed (ESM Appendix S1). For manure nutrients produced in the watershed, EWG (20 19) used the Midwest Planning Service (MWPS-18 . 2004 ; Vitosh et al., 1995) publication, which primarily differs from Ruddy et al. (2006) because of the need to estimate the size of the animal(s) of interest to estimate the amount of manure and corresponding nutrients produced.The differences between the studies in estimating livestock populations and manure nutrients contributes to differences in the amount of manure phosphorus generated from livestock in the region. Comparing results from EWG (2019) to this study, basin-wide manure phosphorus generated in 2015 were larger for poultry (352%) and similar for swine (95%) and combined cattle (100%).

Although EWG (2 019) found similar percentages of manure phosphorus produced from non-permitted swine facilities (79%} as this study (78%) the differences in the two methodological approaches of the stu dies may contribute to differences in manure produced from non-permitted poultry and cattle operations. EWG (2019) found 51% of poultry manure phosphorus was from non-permitted operations while this study found only 6%. Further, this study combined dairy and beef livestock operations into *a* single cattlecategory while EWG (2019) reported dairy and beef livestock separately. This resulted in differences in the fraction of manure from non-permitted operations for these livestock in the two studies (25% for combined cattle in this study; 84% for beef cattle and 34% for dairy cattle in EWG (2019)).

Watershed modeling applications

Results from this study can be used to improve models of the Maumee River watershed that have been used to simulate the impact s of agricultural practices on Lake Erie (Cousino et al.. 2015; Gildow et al., 2016; Scavia et al., 2017) and aid in estimating the impact manure applications have on phosphorus loadings from the watershed. Past studies applying the Soil and Water Assessment Tool (SWAT) model have utilized livestock permits to develop assumptions about how manure was applied (Saleh et al.. 2001), downscaled county -level livestock populations to small spatial scales Uha et al. , 2007), and distributed manure to agricultural fields closer to CAFOs (Muenich et al., 2016) to improve the spatial resolution of manure applications. In the Maumee River wate rshed, previous watershed modeling studies have employed simplifications to the spatial and temporal distributions of manure as well as the manure's nutrient composition (Kak ic et al., 2016: Scavia et al.. 2017; Muen ich et al., 2016). Using region-specific manure nutrient analyses and spatial and temporal analyses of management practices of large livestock operations can improve these modeling efforts by more accurately representing practices within the study area.

Ac.know ledgem e nts

This work was supported by a Harm ful Algal Bloom Research Initiative grant from the Ohio Department of Higher Educatio n as well as through the University Fellowship Program, and the College of Food, Agricultural and Environmental Science Fellowship program. The authors would like to thank Ashley Gignac, E'lise Hardin . Awoke Dagnew.Yu-Chen Wang,and Noel Aloysius for their data collection efforts and guidance, as well as three anonymous reviewers who helped improve this manuscript.

Appendix A.Supple me nta ry data

Supplementary data to this article can be found online at https://doi.org/10.1O16/j.jglr.2019.09.015.

Referenc es

Aronsson. H.. Liu. J.. Ekne. E.. Tome nssonm,G.. Salomon. E.. 201 4. Effects or pig and dairy slurry application on N and P leaching from crop rot.itions wil h spring cereals .nd forage leys. Nutr. Cyd . Agroecosyst. 98 (3 281- 293.

Bentley,J.. limnu, LL.Tranel, LF.. Lenth. R..2016. Manure Ma nagementSystems - Iowa Dairy Producer Surveys. Anim.Ind. Rep.. AS 662 . ASL R3074.

Blomqvist, S.. Gunnars. A., Elmgren . R.. 2004. Why the li mitin1 nutrient diffors 1><,,woen temperate coasial seas and freshwater lakes: a matter of salt. Limno. Oceanogr. 49 (6), 2236- 2241.

Brooks. B.W.. Lazorcha k,J.M.. Howard, M.D..Jo hnson,M.V.V.. Mortin,S.L, Perkins,D. A.. Re.avie, E.D., Scott.G.I., Smith, SA, Steevens.JA, 2016. Are harmful alg•I blooms becoming the greatest inland waterquality threat to public health and aquatic ecosystems?. Env. Tox. Chem. 35. 6-13. https://doi.org/10.1002/er/3220.

Carmichael. W.. Boyer. G., 2016. Heallh impacts from cyanobacteria harmful algae blooms: implications forthe Nonh AmericanGreat Lakes. Harm. Algae 54. 194 - 212.

Chastain.J.P.,Camberato,J.J.2004.Dairy Manure Production and Nutrient Content.

Copeland, C.. 2010. CRS Report for Congress Animal Waste and Water Quality: EPA Regulat ion of Concentrated Anim• I Feeding Operations (CAfOs).

Correll. D.L.. 1 999. Phosphorus: a rate limiting nutrient in swfacewarers. Poult.Sd. 78. 674- 682.

Cousino. LK. Becker, R.H.. Zmijewski, K.A., 20 Is. Modeling the effects of climate change on water. .sediment, and nutrient yields from the Maumee River wat ershed. J. Hydrol. Rei. Stud.hnps:/ldOI.Or&/10 1016 /j.J rh..201S.06.017 .

Dav_is,T.W.. Stumpf. R.. Bullerjahn,G.S.. McKay, R.M.. Chaffin, J.D.. Bridgeman.T.B.. Winslow. C.. 2019. 5elence meets policy: a framework for determining impairment designation criteria for large watertlodies affected by cyanob.lcterial harmful •lgal blooms. Harm. Algae Bl. 59-64. hnps://doi.org/10.1016/j.hal.2018.11.016.

Depinto, J.V.. Young. T.C.. Mcilroy. LM., 1986. Creat laku water quality improvoment. Environ. Sci.Technol. https:l/dol.org/10.1021/esOOl SOaOOl .

EWG (Environmental Working Group), 2019. Explosio n of Unregulated Factory Farms In Maumee Watershed Fuels Lake Erie's Tocis Blooms.

Cildow. M., ysius. N. Gebremanam. S..Man u,.J. 2016.Fertilizer placomentand applic• tton timing as strategies ro reduce phosphorus loading ro (Ake Erie. J. Great Lakes Res. 42 (6), 1281-1288 .

Cowda. P.. Mu,lla D.. Jaynes, D.. 2008. Modeling impact of timing and amount of nitrogen fertillzor o.n water quality for an Iowan agricultural watershed. Agrlc. Water Manag. 95, 616- 624.

Grattan. LM.. Holo baugh.S.. MorrisJr..J-G..2016. Harmful algal blooms and public health. Hann. Alga• 57, 2-8. https:l/doi.org/l0.1016/jhal.2016.05.003.

Hoagland. P.. Andorson, D.M,. Kaoru, Y., White, A.W., 2002 . The economic effects of am1ful algal blooms In lhe United Stares: estimates. assessment Issues, and information needs. Esrua.ries. hnps://do1.org/10.1007/BF02804908.

Honeyman, M.S.. 1996. Sus1amability Issues or u.s. swine productton 1. Exp. Sta, J. Anim. Sci. 74.1410--1417.

Hood•.P.S.,Edw rds. A.C., Anderson, HA, Miller,A..2000.A review of waterquality concerns m livestockfanning.uus.Sci. Total Environ. hnps://doi.org/10.1016/50048-9697(00)00373-9.

Hribar,C., 2010. Understanding Concentrated Animal feeding Operations andTheir Impact on Communities.

IDEM (I ndiana Department of Envfronmontal Managemont). 2014. Guidance Manual for Indiana's Confined feeding Program.

International Joint Commission, 2018. fertilizer Applic.irion Patterns and Trends and Their Implications for Water Quality in1he Western Lake Erie Basin.

Jha. M. Gassman. P., Amold. J.. 2007. Water quality modeling for the Raccoon River watershed usingSWAT. Trans. ASABE SO. 479 - 493 .

K.llclc, M.M.C., Kirchhoff, C.,Bosch, N.. Muenich, R.L. Murray, M.. Griffith Gardner. J.. Scavia, D., 2016. Engaging stakeholders 10 define feasible and dosirable agricultural conservation in Western Lake Erie Watersheds. Environ. Sci. Technol. hrtps:l/doi.org/10.1021/acs.est.6b01420.

Kane, 0.0 .. Conroy, J.D.. Poter Richards. R.. Baker. 0.8.. CUiver. DA. 2014. Ro-eutroph1c.iaon of Lake Erie: Correlanons between tributary nutrient loadsand phytoplankto n biomass. J. Creat Lakes Res. 40 (3). 496 - 501.

Keeney. R.. 2008. CAFOs Community impacts of CAFOs: Growing Trends Social/ Economic iSSuES.

Kellogg, R.L, 2002. Proflie of farms with livestock In the United St<1tes: a srarlstlcal summary.U.S. DepartmentofAgriculture,NaturalResourcesConse,vationService.

Key, N., Mcbndc. W.D.. Ribaudo M_Sneeringer. S. 2011. Trends and Developments m Hog M nure Ma nagement. U.S. Dept. Agrlc. Econ. Res. Sc rv.

Lenhardt,J., Ogneva-Hlmmelberger,Y.. 2013. Environmental injustice in the spatial distributionofconcentrated animal fecdin& operations In Ohio. Environ. Just.6, 133- 139. https://doi.org/10.1089/env.2013.0023.

Ll. H.M.. Tang. H.J.,Sh i, X.Y,. Zhang, c.s.. Wang. X.L. 2014. Increased nutrient loads from the Changjiang (Yangtze) Rive r have led to incneased Harmful Algal Bloonu. Harm. Algae. https :l/ do i.orz /l0 .1016 /j.hal.2014.07 .002.

Liu,J.V.. Collick. T.. Kleinman, A.. Beegle. P.. Bryant. D.R.. 2017. Seasonal manure application timing and storage effects on field-.and watershed -level phosphorus losses. J. Environ. Q)Jal. 46 , 1403 - 1412. https :/l do1.org/10.2134/joq2017.04.0150.

Liu. G.. Li, Y.. Ga1.Ula. A . 2013 . Convenions of Pans per Million on Soil Test Repons to Pounds per Acre. HS 1229. Unlvorsity of Florida Institu te of food and Agricultural 5eiences. Gainesville. Retrieved January l S,2018, from hltp:1/ed is.lfas.un.edu/hsl 229.

Long. C.M.. Logsdon,R..Kalcic. M.M.•Scavia. D.. 2018. Useor manure nutrients from concentrated animal feeding operations. J. Great Lakes Res. 44 , 245- 252. https://doi.org10.1Ol6/jJgJr.2018.01.006.

Lorimor.J.C.. Kohl. K.. 1998 . Liq uidSwine Manure Nutrients.sw ine Res. Rep. Anim. Sci Res. Ropons.

Lory, J.A., Massey, R.. Zulovich, J.. Millmior, A. Hoehne.J.. And, P.E. Case, C.. 2001, Chapter 4 on Farm Evaluation of Adopting Phosphorus Versus Nitrogen Limits forManure Applicationon USSwineOperations, E<:onomic Viability of USSwine farms ImplementingWaler Quality Best Available Technologies.

Macco ux, MJ., Dove. A.. Backus. S.M., Dolan, D.M.. 2016. Total and soluble reactive phosphorus loadingsto Lake Ene: a detailed accountingby year. basin, country. .ind tnbulary. j. Gneat Lakes Ros. 42 (6). ll 51-1 165.

MDEQ(Michig.n Department of Environmental Quality), 2015, Natio nal Polluranr Discharge Elimination System Wastewater Discha rge General Permit for Concentrated Animal Feeding Operations. Permit No . MIGOI000.

Muenich, R.L, K.lld c. M., Scavia, D.. 2016. Evaluating the impact or legacy p and agricultural consetvation practices on nutrient loads from the Maumee River wa tershed. Environ. Sci. Technol. https:}/doi.org/10.1021/acs.u t.6b01 2 l.

Nahm, K.H. 2002. Efficient feed nutrient utiliu tion to reduce pollutants in poultry and swine manure. Crit. Rev. Environ. Sci. Technol, https://doi.org/10.1080/10643380290813435.

Ohio Environmental Council,2017. Ohio'sConcentrated Animal Feeding Facilities:A Review of Statewide ManureManagement and Phosphorous Applications in the Western Lake Erie Watershed.

Ohio EPA. 2016. Nutrient Mass Balanu Study for Ohio's Major Rivors.

Ohio EPA. 201 8. Nutrient Mass Balan« Study for Ohio's Major Rivers.

ORC (Oh io Revised Code). 2014. 901:10-2-14 Contents of Manure Management Plan: Land Application Methods, Appendix E.

Paerl. H.W.• Scott, J.T. Mcurthy. MJ.. Newell. S.E.. Gardner. W.S., Havens, K.E.. Hoffman, D.K..Wilhelm. S.W.. Wurtsbaugh. W.A., 2016. 11takes two ro tango: when and where dual nulrlent(N&P)red uctionsarc needed to protect lakes and downstream ecosystems, Environ. 5el.Technol.https:}/doi.org/10.1021/acs.est.6b02575

Riba udo. M.. Kaplan, J.. Chris1ensen. L. Gollehon. N., Johansson, R.. Breneman, V.. A1lery, M.. Agapoff, J.. PetersmM.. 2003. Manure management forwater quality costs10animalfeeding operations or applyingmanure nutrients to land.

Ruddy. B.C.. Loremnz, D.L, Muellor, D.K.. 2006. Count-level estimates of nutrienl inputs 10 the land surface of the conterminous United Stat es. 1982- 2001. U.S. Geologic.iiSurvey Scientific lnvesrieations Report 2006-SOl2.

MWPS-18, 2004. Manure Characteristics. MidWest Plan Service (MWPS). Ames, Iowa.

Saleh. A.. Arnold.J.C., Gassman. P.W., Hauck. LM. R ntha l. W.D.. Williams. j.R, McFarland, A.M.S. 2001.Applic.irio n of SWAT fo r th• Upper North Bosque River Watershed. Trans. Am. Fish. Soc . 43, 1077- 1087.

Scavia, D. David Allan J., Arend, K.K. Bartell.S. Beletslcy. D.. Bosch, N.S., Brandt. S.B. Briland, R.D.. DalotJu, L, DePinto, j.V.. Dolan. D.M.. Evans, MA . Fanne r. T.M.. Goto, D.. Han. H.. HMk, T.O., Knigh t, R.. Ludsin. S.A.. Maso n, D.. Micha lak.A.M., Peter Rlc:hards, R.. Roberts , J.J.. Rucinsk.i D.K.. Ruthe rford, E., Schwab. D.J.. Sesterh nn_T.M.. Zhang. H.. Zhou. Y.. 2014 . Assessing and addressin, th,, re-outrophlcauon or Lake Ene: Central baSin hypoxia. J. Great Lakes Res 40 (2), 226- 246.

Seavia. 0., Ka lcic. M.. Logsdon Muenich, R, Aloysius, N.. Soles, C., Confosor, R.. Depinto. J.. Gildow. M., M• rtin, J.. Read. J.. Redder, T., Robenson, D.. Sowa. S.. Wang, Y.--C... Ye n. H.. 2016. Informing Lake Erie Agriculture Nutr ie nt Management via Scenario Evaluation.

Savia, D.. Kalcic, M.. Muenlch, R.L, Re•d. J.. Aloysius, N., Bertani, I.. Boles, C.. Con feso r, R., OePlnto, J.. Gildow. M.. Martin,]., Redder,T.. Robertson, D.,Sowa,S., Wang, V.C. Yen, H.. 2017. Multiple models guide s1megies for agricultur•I nutrien1 reductions. Front. Ecol. Environ. https :J/dol ors/ 10.1002/Fee.1472.

Schindler, D.W., Carpenter, S.R., Chapra. S.C.. Hecky, R.E.. Orihel. O.M., 2016. Reducing phosphorus ro curb lake eutrophication is a success. Environ. Sci. Technol. https :// doi.org/10 .1021/acs.est.6b02204.

Schus1e,rNR.. Bartelt-Junt . S.L. Ourso. LM.. Giley .J.E.. U.X., Marx.D.8..Schrmdt. A. M., Snow. D.D.. 2017. Runoff water quality characteristics following swine slurry application und er broadcast and injec1ed condi1Jons. Tr• ns. ASABE 60 , 53-66 .

S mith. 0 .. OWens. P. Leytem. A.. Wamcmuende. E.. 2007. Nutrienl losses from manure and fertilizer applications as impacted by time 10 first runoff event. Environ. Poll. 147 (1). 131- 137.

Sukenik, A.,Quesada, A. 5almaso. N., 2015.Glob.ii expansion of toxic and non-toxic cyanobacteria: effect on ecosystem functionin&- Biod iv . Conserv. 24, 889-908 . https://doi.ori/ 1o.1007/sl 0531-0 l S-0905-9.

USDA. 2012. National Agriculture s1a1is1lcs Survey .

USDA. 2017. National Agriculture Statistics Survey.

USEPA. 2008.The Oean Water Act. 33 US.C 1251 § 122.23.

USEPA. 2018. U.S. Action Plan for Lake Erie: Commitments and Stra tegy for Phosphorus Reduction.

USGAO. 2008. Concentrated Animal Fttding Operations.

Vadas, P., Good, L, Joke la. W.. Karthikeyan, K.G.. Arriaaa. F., Stock. M., 2017. QuantlFying the Impact Or seasonal and short-term manure a pplication decisions on phosphorus loss in surface runoff. J. Environ. Qua!. 46. 1395-1402. https :/Jdo1.org/ I0.2 l 3 /jeq20 l 6 .0 li 0220 .

Vilosh. M..Johnson, J., Mengel, O..1995. Tri-State fertilizer Recommendations for corn. soybeans. wheat, and alfalfa. Ext.Bull E-2 567. Michigan sme University.

Vollmer-S,mders, C. Allman, A..Busdeker,0 .. Beal, L. Stanley , W.G.• 2016. Building partnerships to scale up conservation: R Nutrient Stewardship Certification Program In the l..lke Erie watershed. J. Great l..lkes Res. 42, 1395-1402. https:// doi.org/I0.1016/JJglr.2016.09.00.4

Waskom. R., Davis. J.. 1994 . Best Managem,nt Practices for Manure Utilization Bulletin 568A.

Watson, M.. Mullen, R., 2007. Understanding Soil Tests for Plant-An ilab le Pho,phorus.

Watts. 0.8.• Way. T.R.. Torben , HA, 2011. Subsurface application of poultry litte r and its influence on nutrient losses in runoff waterfrom permanent pastures.J. Environ. Qua!. https:l/ dol.org/l0 .2134/jeq 20 I0.0089.

Williams, M.R,. King. K.W.. Duncan. f..W., Pease, LA.. Penn , CJ, 2018. Fertlllzcr placement and tillage effects on phosphoru.s concentration in l,a chate from fine-textured soils.Soil Tillage Res. htrps :f/dol.org/10.1016/j.still.2017 .12.010.

Wolf, D., Georgie. W.. Kla ibe r, HA , 2017. Reeling In the damages : harmfu l alg.,I blooms' impactson LakeErie's recrear,onal fishing industry.J.Environ.Manage 199, 1 8 - 157 ,

REVISION OF THE 590 NUTRIENT MANAGEMENT STANDARD:

SERA-17 RECOMMENDATIONS

SERA – 17 Members:

- Andrew Sharpley, Dept. Crop, Soil & Environmental Sciences, Univ. of Arkansas, Fayetteville, AR (Chair)
- Doug Beegle, Dept. Crop and Soil Sciences, Pennsylvania State Univ., State College, PA
- Carl Bolster, USDA-ARS, Animal Waste Management, Unit, Bowling Green, KY
- Laura Good, Dept. Soil Science, University of Wisconsin, Madison, WI
- Brad Joern, Dept. Agronomy Purdue Univ., West Lafayette, IN
- Quirine Ketterings, Dept. of Animal Science, Cornell University, Ithaca NY
- John Lory, Division of Plant Sciences, University of Missouri, Columbia, MO
- Rob Mikkelsen, International Plant Nutrition Institute, Davis, CA
- Deanna Osmond, Soil Science Dept., North Carolina State University, Raleigh, NC
- Peter Vadas, USDA-ARS, Dairy Forage Research Unit, Madison, WI

Background

NRCS's short-term goals for a revised Phosphorus Index (P Index) or Phosphorus Risk Assessment Tool (PRAT) are to:

1. Prevent the gradual loading of phosphorus (P) to high water quality risk levels.
2. Assist producers in mitigating existing high water quality risk situations to lower sustainable P levels.
3. Determine and implement a "cutoff" to identify those conditions where no additional P shall be applied.
4. In order to accomplish the above goals, the P Index should include the following:
    a. A tool built on a national platform with scientific underpinnings.
    b. A tool to assess the potential for edge-of-field P runoff and leaching.
    c. A tool based on the best available science that can be refined / improved as better technology or science becomes available.
    d. A tool that can utilize local soil, hydrology, and climate data (these data already reside in wind and water erosion prediction tools used in NRCS field offices) that can track erosion and sediment transport to concentrated flow, to a point of deposition, or edge of field.
    e. A tool that can address, where needed, irrigation-induced erosion, runoff, and leaching.
    f. A tool that can assess risk from manure and/or P fertilizer.
    g. Although the proposed P Index would be quantitative, it is not necessary that the results be delivered numerically. A narrative or category rating (Very Low, Low, Medium, High, Very High, etc.) would be satisfactory.
    h. The minimum criteria for edge-of-field P runoff should be that nutrient concentrations in runoff reaching a stream or water body will not cause water quality impairment (algae,

aquatic habitat, etc.). The tool will also need to identify those fields/situations where even with the best conservation, no additional P should be applied.

The Charge to SERA-17

Based on the above requirements the SERA-17 subgroup had the following charges (Figure 1):

1. Define criteria establishing the range of soil test P (STP) values where a P Index risk assessment is needed.
2. Define the upper P Index threshold that limits P application.
3. Define the minimum requirements of P Indices.
4. Define a process to evaluate P Indices.
5. Define long-term goals for development of the next generation P Indices.

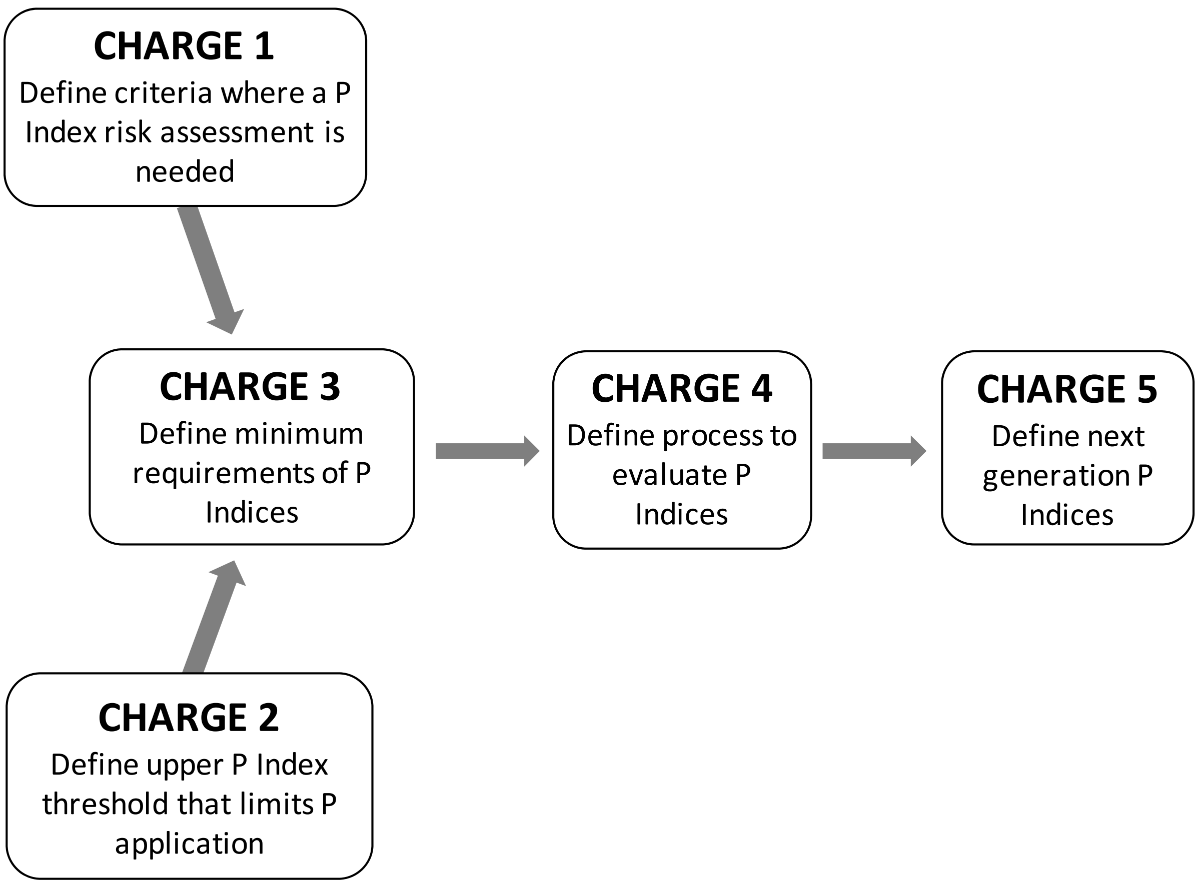


Figure 1. Organization scheme of the 590 revision charges.

EXECUTIVE SUMMARY

- The goal of a P Index is to estimate the potential for P loss from any agricultural field. Phosphorus Indices were not designed to address or solve the broader issue of regional P surpluses. Many P Indices force a P balance approach on individual fields at some point; however, this point varies greatly and P Index cutoff values (the P Index value where no additional P is recommended) are not tied directly to water quality. A separate effort to address P balance (i.e., inputs equal to or less than outputs) at a watershed scale is needed. A P-balance approach will involve alternative technologies for manure utilization and export of manure from many farms in some watersheds.
- Many states have developed adequate tools to estimate the potential for P loss by describing the main factors and conditions controlling P loss in their state. However, there is substantial variation among P Indices in their structure, algorithms, and cutoff values used to delineate very low, low, medium, high, and very high risk of P loss. More importantly, there is a great deal of inconsistency in results and interpretation regardless of the details of the tool used.
- States may find it appropriate to eliminate the requirement of a P Index assessment when P applications are based on land-grant university nutrient recommendations and appropriate Best Management Practices (BMPs) for land application of P sources as defined by NRCS Conservation Practice Standards. For P application in excess of recommended rates, a P Index assessment will need to be conducted.
- All P Indices should "zero out" at some point. That is, there is a point above which the risk of P loss from a field is too great to warrant the application of P in any form. Each state should demonstrate that its P Index meets this criterion. We provide several approaches to determine this point, and where field-based research has been conducted to develop upper limits, state specific information should take precedence.
- There are too many legitimate differences in soils, climate, cropping systems, water body sensitivities, etc., and insufficient progress in modeling of all processes to support development and use of a single National P Index that addresses all of these differences, especially if a National Index must be user-friendly and require minimal input data and training for end-users at this time. Development of a National P Index will require a long-term commitment of time and resources similar to that required for the development of the USLE. Development of a P loss assessment tool that addresses the P loss issues specific to a physiographic region is desirable and should be a long-term goal of SERA-17 and NRCS collaboration.
- Although there is no scientific evidence to support the use of STP or P saturation alone to determine the risk of P loss; because P is a finite resource, states should consider establishing an upper limit of STP above which manure cannot be applied, regardless of P Index assessment.
- There needs to be a concerted training effort on how to use P Indices in the context of nutrient management planning and how to address any concerns identified by the P Index used during the planning/implementation process.

CHARGE 1

DEFINE CRITERIA ESTABLISHING THE RANGE OF SOIL TEST PHOSPHORUS VALUES WHERE A PHOSPHORUS INDEX RISK ASSESSMENT IS NEEDED

Recommendation

The lower limit of the range of STP values where a P Index risk assessment is needed can be based on land-grant university P application recommendations. States may find it appropriate to eliminate the requirement of a P Index assessment when P applications are based on land-grant university nutrient recommendations and appropriate Best Management Practices (BMPs) for land application of P sources (NRCS Conservation Practice Standards). For P application in excess of recommended rates, a P Index assessment will need to be conducted. States could develop a screening tool or other resources to identify high risk areas where a P Index assessment should be conducted even if STP results in a P application recommendation.

Because P is a finite resource, states should establish an upper limit of STP above which manure cannot be applied, regardless of P Index assessment. However there is no scientifically defensible way to set a uniform national upper STP bound based solely on water quality goals.

Considerations

Setting the lower STP limit when no P Index assessment is required

- The P Index (or pre-screening tool) should only be optional for fields with an agronomic need for P, based on STP and land-grant university nutrient recommendations.
- Producers are required to meet all other field-specific NRCS conservation objectives and standards, including erosion control, manure application setbacks, proper timing of manure application, and annual N limits for the crop. These conservation requirements apply to all nutrient applications independent of source according to the NRCS National Nutrient Management Standard.
- A low STP level does not mean there is no risk for P loss from manure or fertilizer application. For instance, the application of P to critical risk areas, such as fields adjacent to a stream with a high transport risk should be avoided. States that do not require the use of the P Index when an agronomic P need exists, could develop and use a screening tool to identify any local high risk situations (e.g., 303(d) listed waters for P or other state designated P-related impairment, erosion greater than T, high runoff potential, and within 30 m of flowing water) where the P Index should be used even when P applications are recommended.
- In some states, the P Index may allow repeated N-based applications, which can lead to a buildup of STP in excess of soil test P-driven nutrient recommendations. Because the recommended approach of Charge 1 never allows P applications to exceed crop rotation requirements, it is more restrictive than repeated N-based application rates.

- This approach promotes use of manure as a nutrient resource and ensures that farmers who manage manure P in this way can avoid conducting a P Index assessment when developing a nutrient management plan or adjusting a manure application rate based on new information, such as information from regular and ongoing soil or manure test results. This allows limited planning resources to be targeted to higher priority areas.
- Manure P can be applied at a rate to meet the recommendation for multiple crop years (length to be determined by each state) without the need to do a P Index assessment. For example, with a three-year limit, a farmer could apply manure (based on the total P concentration of manure) in one year to meet three years of crop P need, as long as crop N requirements are not exceeded. No additional P is applied in the current and two additional years. However, given the short-term over application of P, states may want to provide additional guidance requiring agronomic practices that have been shown to minimize P runoff (e.g., subsurface placement, injection).
- It is theoretically possible that this approach would allow a manure or fertilizer application when the P Index recommends no application of manure. Reviewing current P loss assessment strategies from 21 states, shows that the P Indices in six of these states may indeed prevent manure application to fields when STP values are below the agronomic threshold (Table 1). In most cases, this would occur under specific and limited conditions (e.g., organic soils, high transport potential, proximity to a stream, specialty crops) for manure application and/or when manure application rate was high. Soil test P values at which no additional P is recommended are summarized in Table 2 for 24 states.
- Given the urgent need for improvements in P recommendations for environmental risk assessment purposes, continued efforts to use accurate data are essential. Private soil testing laboratories should be encouraged, if they are not already doing so, to participate in a laboratory certification program to verify that analytical procedures are performed correctly. T hey should also be encouraged to work with land-grant universities to ensure testing methods are consistent with extraction protocols established by the land-grant university in the state where the soil sample was taken. In addition, NRCS 590 standards should require soil test laboratories be certified and use land-grant university nutrient recommendations for both N and P. For states that do not have this requirement in their NRCS 590 standard, soil testing analysis and recommendations can vary significantly. See Appendix A for more information.

Setting the upper STP limit when no more P should be applied because of limited P resources

- There is no scientific evidence to support the use of STP or P saturation alone to determine the potential for P loss from a field. A wealth of scientific evidence is available documenting that agronomic STP or soil P saturation is only one of several factors influencing the risk of P loss from a field. Use of agronomic STP or P saturation alone will not capture a site's risk for P loss (see Appendix B for more information). Any effort to set regional or national limits based solely on STP or P saturation will encounter the following challenges:

1. Inability to define cutoff values based on water quality criteria because of the lack of a correlation between STP or P saturation and edge-of-field runoff water quality.
2. Because several different STP methods and depths of soil sampling are used across the U.S., equivalent values for each method would have to be determined.

- There are legitimate reasons to set an upper STP boundary not directly associated with current P loss potential of a field:
  1. Phosphorus is a finite natural resource that needs to be conserved. Thus, we support achieving on-farm and regional P balance with the long-term goal of meeting agronomic requirements. The unlimited over-application of P to soils is not a sustainable use of this finite resource. Limited buildup of STP above agronomic thresholds (Table 2) can achieve both agronomic and economic goals by maintaining agronomic P levels through a rotation or as a hedge against volatile fertilizer prices. At some point, continued buildup of STP has no possible agronomic value and can only be classified as a waste disposal P application.
  2. There is no guarantee that conditions currently limiting P transport on low P index fields will be maintained in perpetuity.
- The P index in many (if not all) states allows build up of STP above agronomic need on most fields. States should consider defining where STP buildup transitions above "insurance" applications. Such a boundary may be considered as a limit to P application to meet resource conservation goals or as an educational tool so farmers understand there is little or no expectation of utilization for applied P to fields with STP above that limit.

The following are possible approaches states may use if they choose to set an upper STP threshold above which no manure application is allowed:

1. **<u>Select a multiple of agronomic STP optimum</u>**. The resulting limit could be interpreted correctly independent of the extraction procedure. States using a specific extraction procedure could later translate the guidance into specific extract concentrations.
2. **<u>Select a draw down STP level</u>** that would require no more than a set number of years to be drawn down to optimum under normal cropping conditions.

Table 1. Conditions under which P Indices could limit P applications on a field with an agronomic need for P in selected states.

State	Can state P Index restrict P applications on soils with an agronomic need for P?	Basis of Determination	Reference
AK	Yes	Can limit agronomic applications where site, transport, methods of application and timing factors are all at very high or worst-case scenario levels.	NRCS Alaska PI Index. May 2002.
AR	No	Restrictions most likely to occur on soils with high rates of P application coupled with high transport potential.	Moore, P.A., Jr., A. Sharpley, W. Delp, B. Haggard, T. Daniel, K. VanDevender, A. Baber, and M. Daniel. 2010. The Revised Arkansas Phosphorus Index. Arkansas Natural Resources Commission Title 20. http://www.anrc.arkansas.gov/Title%2020%2012-10-09.pdf .
CO	No	P index does not need to be run if STP is less than 10 mg kg^{-1} AB-DTPA, 30 mg kg^{-1} Bray-I P, 40 mg kg^{-1} Mehlich-3 P or 20 mg kg^{-1}Olsen P. This will result in no restriction on agronomic P applications except for potatoes.	USDA-NRCS State of Colorado. Agronomy Technical Note No. 95 (revised). Colorado Phosphorus Index Risk Assessment (Version 4). October 1, 2008.
CT	No	State has no P-Index, but P applications are not restricted if soil test recommends P applications.	http://efotg.sc.egov.usda.gov/references/public/CT/CT_590_2010_F.pdf
DE	No	The State of Delaware's Nutrient Management Commission has established a Mehlich 3 P threshold of 150 mg kg^{-1} (3 times the University of Delaware M3 P critical value of 50 mg kg^{-1}) as the basic definition of a "high P" soil. By state law (Delaware Nutrient Management Act of 1999), soils that are "high" in P can continue to receive manure or fertilizer P in any given year at the rate that will	Sims, J. T. and Leytem, A. B. 2002. The Phosphorus Site Index: A Phosphorus Management Strategy for Delaware's Agricultural Soils. Nutrient Management Fact Sheet No. 5. University of Delaware College of Agriculture and Natural Resources, Newark, DE 19717-2303.

		be removed by crop harvest in the next 3 years, but no additional P can then be applied for 3 years (i.e., P is applied once at a "3-year crop P removal" rate, then again 3 years later). However, farmers are given the option to use a P Site Index for soils with M3-P > 150 mg kg^{-1} and to apply manure and fertilizer P in accordance with the recommendations of the P Site Index. The University of Delaware recommends that no manure or fertilizer P be applied if a field has a "Very High" P Index rating. For soils with a "High" P Index value, the recommendation is that "…fertilizer P, other than a small amount used in starter fertilizers, will not be needed. Manure may be in excess on the farm and should only be applied to fields with a lower P Site Index value." It is possible, but highly unlikely, that soil erosion or artificial drainage could result in a Very High P Index value and restrict manure applications to a soil with an agronomic need for P.	
GA	Yes	P Index could restrict agronomic applications in soils with high transport potential.	Cabrera, M.L., D.H. Franklin, G.H. Harris, V.H. Jones, H.A. Kuykendall, D.E. Radcliffe, L.M. Rise, and C.C. Truman. 2002. The Georgia phosphorus index. Cooperative Extension Service, Publications Distribution Center, University of Georgia, Athens, Georgia, 4pp.
IN	No	Application rate bases for nutrient applications are determined by STP according to Chart B if the Indiana off-site risk pre-screening tool value is <6. If the Indiana off-site risk pre-screening tool is >6, the Indiana Off-Site Risk Index (ORI) must be completed and all risk components identified must be addressed. After all risk components identified by the ORI have been addressed nutrient applications are determined by STP according to Chart B.	Indiana Nutrient Management Standard. July 2001.
KY	No	P Index is not required until Mehlich-3 STP values exceed 200 mg kg^{-1} which is ~ 7 times greater than the agronomic recommendation for most crops.	Kentucky Nutrient Management Standard, May 2001.

MD	Yes	P Index may restrict agronomic applications for sites with very high off-site transport potential (e.g. high erosion potential) and close proximity to surface water and/or surface application of manure.	Coale, F.J. 2005. The Maryland Phosphorus Site Index Technical Users Guide. Soil Fertility Management Series, SFM-7. Maryland Cooperative Extension. http://www.anmp.umd.edu/files/SFM-7.pdf.
ME	No	Restrictions affect soils with soil test P greater than 20 mg kg^{-1} where no P application is recommended.	
MO	No	P Index is designed to insure rating of no higher than "medium" on fields with agronomic need and soil loss less than 2T. Therefore, the P index should never limit agronomic applications on fields where erosion limits of the 590 standard are being met.	Lory, J.A., R. Miller, G. Davis, D. Steen and B. Li. 2007. The Missouri Phosphorus Index. MU Extension Pub. G9184.
NC	Yes	P Index almost always restricts agronomic applications on organic soils at the agronomic cutoff for P. Most manure, however, is not applied to organic soils.	Johnson, A.M., D.L. Osmond, and S.H. Hodges. 2005. Predicted Impacts of North Carolina's Phosphorus Loss Assessment Tool. *J. Environ. Qual*. 34:1801-1810.
NY	No	Restrictions most likely to occur on soils with high rates of P application coupled with high transport potential.	Czymmek, K.J. Q. M. Ketterings, L. D. Geohring, G. L. Albrecht. 2003. The New York Phosphorus Runoff Index. User's Manual and Documentation. CSS Extension Publication E03-13. 64 pages.
OK	No	Nutrient Management Standard states that no manure application only on fields with Mehlich3-P >150 mg kg^{-1} (STP Index >300).	Oklahoma Nutrient Management Standard. March 2007.
PA	Yes	Using all the worst-case scenarios leads to no application if the P application rate from all sources exceeds 100 lbs $acre^{-1}$. Result only applicable in special protection watersheds and applications within 150 feet of receiving water.	2007. The Pennsylvania Phosphorus Index, Version 2.
SC	No	P Index cannot be used to limit or deny applications of P when it is recommended for crop growth through soil test results	The Phosphorus Index: South Carolina. 210-AWMFH, SC Supplement, July 2004.

TN	No	The P Index assessment is required for P applications where no further P additions are agronomically needed as defined by Mehlich-1 soil test P.	Tennessee Phosphorus Index: A Planning Tool to Assess & Manage P Movement. 2001.
TX	No	When the Mehlich-3 soil test P reaches 200 mg kg^{-1} in East Texas (counties with greater than 25 inches of precipitation) or 350 mg kg^{-1} (counties with less than 25 inches of precipitation and named streams greater than 1 mile away), the maximum application would be 1.0X P annual crop removal rate, not to exceed the annual N rate of application for PI ratings of Very Low, Low, Medium, or High and for Very High it is 0.5X the annual P crop removal rate.	Texas Nutrient Management Practice Standard. July, 2007.
UT	No	Nutrient management guidance states that Olsen-P of 50 mg kg^{-1} manure can be applied according to the agronomic N need. Between 50 and 100 mg kg^{-1}, manure should be applied according to the agronomic P need. Above 100 mg kg^{-1} Olsen P, manure should only be applied at 50% of agronomic P need.	Utah 590 Standard: http://extension.usu.edu/files/publications/publication/AG_Soils_2008-01pr.pdf
VA	No	P Index does not come into effect until Mehlich 1 P above agronomic optimum	http://p-index.agecon.vt.edu/
WI	Yes	It is possible to have particulate P loss that exceeds the WI target P Index value with STP in the optimum range for high P demand crops (e.g., potato) even when erosion is below T; these crops rarely receive manure.	2010. The Wisconsin Phosphorus Index, http://wpindex.soils.wisc.edu/

Table 2. Soil test P at which land-grant universities recommend no additional P be applied.

State	Method	Soil sampling depth	Soil test P where no additional P recommended	References
		inches	mg kg^{-1}	
AK	Mehlich-3	Plow depth to a maximum of 6 inches	15-66 Starter P typically recommended	USDA NRCS Alaska Technical Note 16 - Making Fertilizer Recommendations from Soil Test Reports-October 2008.
AR	Mehlich-3	4 (pastures) or 6 (row crops)	36-50	Espinosa, L., N. Slaton, and M. Mozaffari. 2006. The soil test report. University of Arkansas Division of Agriculture, Cooperative Extension Service Fact Sheet FSA2153. http://www.uark.edu/depts/soiltest/NewSoilTest/pdf_files/FSA-2153.pdf
CO	AB-DPTA Olsen	Plow depth or 4 inches	8-11 15-22 P always recommended for potatoes	Davis, J.G. and D.G. Westfall, Fertilizing corn. CSU Ext. Pub. No. 0.538. Oct.. 2009. Davis, J.G. and D.G. Westfall, Fertilizing sugar beets. CSU Ext. Pub. No. 0.542. Apr. 2009. Davis, J.G., R.D. Davidson and S.Y.C. Essah. Fertilizing potatoes. CSU Ext. Pub. No. 0.541. May 2009.
CT	Modified Morgan	6-8	10	University of Connecticut Soil Nutrient Analysis Laboratory Recommendations for Agronomic Growers
DE	Mehlich-3	4 pastures 8 row crops	100 †	Sims, J. T. A. B. Leytem, and K. L. Gartley. 2002. Interpreting soil phosphorus tests. Nutrient Management Fact Sheet No. 4. University of Delaware College of Agriculture and Natural Resources, Newark, DE 19717-2303. Sims, J. T., and K. L Gartley. 1996. Nutrient management handbook for Delaware. Coop. Bull. 59. Univ. Delaware, Newark, DE.
GA	Mehlich-1	4 (pastures) 6 (row crops vegetables)	14-70	Kissel, D.E. and L.S. Sonon. 2008. Soil Test Handbook for Georgia. http://aesl.ces.uga.edu/publications/soil/STHandbook.pdf

IN	Bray 1	8	40-50	Vitosh, M.L., J.W. Johnson, and D.B. Mengel. 1996. Tri-state Fertilizer Recommendations for Corn, Soybeans, Wheat and Alfalfa. Ohio State Univ. Bulletin E-2567
KY	Mehlich 3	3-4 (consv till) 6-7 (conv till)	30-40	Murdock, L. and G. Schwab. 2010. Lime and Fertilizer Recommendations. University of Kentucky Extension Publication AGR-1
MI	Bray 1	8	40-50	Vitosh, M.L., J.W. Johnson, and D.B. Mengel. 1996. Tri-state Fertilizer Recommendations for Corn, Soybeans, Wheat and Alfalfa. Ohio State Univ. Bulletin E-2567
MD	Mehlich-3	8	50	McGrath, J. 2010. Agronomic crop nutrient recommendations based on soil tests and yield goals. Soil Fertility Management Series, SFM-1. Maryland Cooperative Extension. http://www.anmp.umd.edu/files/SFM-1.pdf.
ME	Morgan	6	20	Hoskins, B.R. 1997. Soil Testing Handbook. Revised 2001. Available at http://anlab.umesci.maine.edu/soillab_files/faq/handbook.pdf.
MO	Bray 1	6	35	Soil Test and Interpretations Handbook. Revised 5/2004. Available at http://aes.missouri.edu/pfcs/soiltest.pdf.
MS	Lancaster	4–6 pastures, 6 crops	36	Oldham, J.L., and K.K. Crouse. Soil test-based inorganic fertilizer nutrient recommendations for Mississippi agronomic crops. MSU Extension Service Soil Testing Laboratory.
NC	Mehlich 3	4 (consv till) or 8 (conv till)	60	Hardy, D.H., M.R. Tucker, C.E. Stokes. 2009. Crop Fertilization Based on Soil Test Report. http://www.ncagr.gov/agronomi/pdffiles/obook.pdf. NCDA&CS, Raleigh, NC
NY	Morgan	6-8	20	Ketterings, Q.M., K.J. Czymmek and S.D. Klausner (2003). Phosphorus guidelines for Field Crops in New York. Second Release. Department of Crop and Soil Sciences Extension Series E03-15. Cornell University, Ithaca NY. 35 pages.
OH	Bray 1	8	40-50	Vitosh, M.L., J.W. Johnson, and D.B. Mengel. 1996. Tri-state Fertilizer Recommendations for Corn, Soybeans, Wheat and Alfalfa. Ohio State Univ. Bulletin E-2567

OK	Mehlich 3	6	41 ¶	Zhang, H. and B. Raun. 2006. Oklahoma Soil Fertility Handbook. 6th Edition. OSU Extension Publication.
PA	Mehlich 3	8	50	AASL.psu.edu Penn State Soil Fertility Handbook
SC	Mehlich 1	6 (crops) 3 (pasture)	27.5 - 40	
TN	Mehlich 1	6	>15	http://soilplantandpest.utk.edu/pdffiles/soiltestandfertrecom/chap2-agronomic_mar2009.pdf
TX	Mehlich 3	6	50	Provin, Tony. 2010. Soil, Water and Forage Testing Laboratory Methods and Recommendations. http://soiltesting.tamu.edu .
UT	Olsen P	12 ‡	15	Cardon, G.E., J. Kotuby-Amacher, P, Hole, R. Koenig. 2008. Understanding Your Soil Test Report. Utah State Cooperative Extension Service AG/Soils/2008-01pr. http://extension.usu.edu/files/publications/publication/AG_Soils_2008-01pr.pdf
VA	Mehlich 1	4 no-till, 6-8 conventional till	55	Maguire, R.O., and S.E. Heckendorn. 2009. Soil test recommendations for Virginia (Update of 1994 version). Virginia Cooperative Extension.
WI	Bray 1	6-8	17-80§ P always recommended for potatoes	Laboski, C.A., J.B. Peters, L.G. Bundy. 2006. Nutrient application guidelines for field, vegetable, and fruit crops in Wisconsin. UW-Extension A2809.

† Optimum range for M3-P in Delaware is 50-100 mg kg^{-1} by Mehlich 3 P. In almost all cases, only starter P is recommended when M3-P values are > 50 mg kg^{-1}.

‡ Value is 32.5 mg kg^{-1} if P is measured colorimetrically.

¶ Recommendation is that the sample be confined to the upper foot. Most will focus on extracting from 6 to 10 inches deep.

§ Value within range depends on crop and soil type.

CHARGE 2

DEFINING AN UPPER P INDEX THRESHOLD THAT LIMITS PHOSPHORUS APPLICATION

Recommendation

All P Indices should "zero out", which means they must identify a critical risk of P loss from a field beyond which no P in any form should be applied. Each state must demonstrate that its P Index meets this criterion for combinations of parameters that influence P loss potential. The upper criteria or threshold should be determined based on local water quality criteria where available, or on a basic set of conditions that in combination lead to an unacceptable risk of P loss. The upper threshold should be used to establish the minimum standard for restricting P applications on a field and should not be used to justify raising limits on P applications in states with more restrictive P Indices.

Considerations

Possible methods for establishing an upper P Index threshold are detailed below and outlined in Table 3.

1. Define P loss limits for a field based on quantitative water quality criteria for the target water body.
    - This approach is similar to that for establishing TMDLs, and provides a quantitative measure justified directly by water quality standards for a specific region. Essentially, the following are estimated: (a) how much total P a specific water body can assimilate without adverse water quality impacts; (b) how much of that total acceptable P load can come from agriculture in the watershed; and (c) an allowable field scale P loss based on the total allowable agricultural P load to the water body.
    - Unfortunately, there are significant technical challenges to setting field-level P limits based on numeric water quality criteria. Currently, numeric criteria for P water quality standards only exist for a limited number of water bodies; and methods to establish field-specific limits on P loss based on numeric water quality limits are not well developed.
    - This approach requires use of a P Index that estimates field scale P loss in lb/ac so P Index results can be directly related to water quality estimates.
2. Run a range of scenarios and estimate P loss for each of them using an appropriate model. Use professional judgment to set runoff P limits that clearly limit risky management and/or prevent levels of P loss likely to degrade water quality.
    - This approach integrates professional judgment and local management into the establishment of P limits. However, subjective criteria are used to connect P loss limits with water quality criteria.

3. Run a comprehensive set of representative P runoff scenarios for a state or region using an appropriate model and set P limits to eliminate application on a specified upper percentile of the scenarios (e.g., top 20%).
   - This approach provides a limit based on local scenarios that will reliably establish and identify the worst situations. However, there is no connection between the limit and any water quality criteria. The limit could be either more restrictive or more liberal than needed.
   - To be successful, this approach requires knowing and running the full range of real field scenarios, from the lowest to the highest P loss rating.

Table 3. Potential strategies to identify field P loss limits in runoff where a P risk assessment strategy should zero out P applications.

Approach Description	Strengths	Weaknesses
Set field runoff P limits based on water quality criteria of the target watershed.	• Quantitative measure justified directly by water quality standards for a specific region. • Preferred approach in TMDL watersheds and when other water quality criteria are available.	• Requires quantitative water quality criteria to be in place and a mechanism to convert to field –level P loss limits. There is insufficient information in place to calculate such limits in many locations.
Run a range of scenarios and estimate P loss for each of them using an appropriate model. Use professional judgment to set runoff P limits that clearly limits risky management and/or prevents levels of P loss likely to degrade water quality.	• Integrates professional judgment and local management into the establishment of P limits.	• Subjective criteria used to connect P loss limit with water quality criteria.
Run a comprehensive set of representative P runoff scenarios for a state or region using an appropriate model and set P limits to eliminate application on a specified upper percentile of the scenarios (e.g., top 20%).	• Provides a limit based on local scenarios that will reliably establish and identify the worst situations.	• No connection between the limit and any water quality criteria. Limit could be either more restrictive or more liberal than needed. • Requires that the full range of real field scenarios be known and run, from the lowest to the highest loss rating, to be successful.

CHARGE 3

DEFINING THE MINIMUM REQUIREMENTS OF PHOSPHORUS INDICES

Recommendations

1. Soil test P, P additions, runoff, and erosion should be continuous variables in all P Indices.
2. The risk assigned by all Indices must increase with increasing STP, P additions, runoff, erosion, and leaching where applicable.
3. Management interpretations of P Indices should provide clear direction, and have at a minimum P-based and no P application categories. Narrative statements of management recommendations (e.g., "conservation measures should be considered to decrease the risk of P loss") have limited specificity in terms of nutrient management and implementation and, therefore, have no place in P Index interpretations.

Considerations

Differences in category boundaries and how those categories affect management are separate issues from differences in calculation. Even using similar calculation methods, there are a wide range of management interpretations for a given risk. Having different categories for management response to the same risk interpretation does not necessarily mean that one P Index is less protective of local water quality than another. Ideally for water quality protection, the interpretation of different levels of risk would not be uniform across all watersheds. Rather, the risk categories and the limits should be assigned based on water quality targets and the assimilative capacity of the receiving water body. However, some P Indices never reach a risk level assessment that restricts manure application to a field (Osmond et al., 2006), and this situation must be addressed.

Clearly, the fact that there is not a framework for establishing risk categories based on water quality is problematic. Without such a framework, the determination of "how much is too much" is generally a value judgment. At present, few states have established numeric P water quality standards. Even with numeric standards in place, it is difficult to make the connection between a field-based risk assessment and P concentrations or loads in receiving waters. We recommend that where water quality criteria are available, such as in TMDL areas, the process used in evaluating P Indices in Charge 4, also be used for setting management interpretation categories. Requirements related to each interpretation category should be clear and descriptive. As stated under Charge 2, all indices should have a no P application interpretation category.

CHARGE 4

DEFINING A PROCESS TO EVALUATE P INDICES

Recommendations

1. Ideally, local water quality data should be used to evaluate P Indices and to establish thresholds based on local water quality criteria.
2. Given that there are limited edge-of-field water quality data available, an alternative approach is to use a nonpoint source model to estimate P loss from a range of conditions consistent with P Index assessment for each state.
3. Where states have already used and validated a regionally appropriate model, that model should be used. Examples of default models are provided below.
4. Reference to any specific model to evaluate P Indices does not imply a recommendation that the model be used as an alternative risk assessment tool to the P Indexing approach.

Recommended Approach to Evaluate P Indices: Using Data and Models

Local water quality standards should be used to evaluate the P Index and to establish P application rate thresholds based directly on these water quality criteria. Unfortunately, these data are limited or unavailable in many states, particularly at scales required to validate the P Index. However, where measured data do exist (e.g., local research sites, National Resource Inventory [NRI] sites) they should be used to validate P indices; and SERA-17 should be encouraged to maintain a database of benchmark fields where water quality data are available for P Index validation (e.g., Harmel et al., 2008). As an alternative to direct evaluation with measured data, appropriate models could be used to provide information for evaluating P Indices, as long as the model selected has been validated to reliably predict field-scale P loss (e.g., Veith et al., 2005). This could also be used as the basis for justifying and documenting if P Index risk assessment does in fact limit P application at a certain specific pre-approved set of threshold conditions (see Charge 2 earlier).

We envision that in a state, or better yet a physiographic region, a model that has been evaluated for local conditions could be used to run simulations on a broad range of scenarios that would cover the expected conditions and management in that region. The P Index would then be run on the same scenarios using the same inputs that were used in the model and that apply to that particular Index. The results of model simulations and P Index evaluations would then be compared. At the present time, a nationally applicable model does not exist to use as the standard against which to compare all P Index assessments. Until a consensus driven alternative is selected, the following models are suggested as an interim option;

- Spreadsheet P runoff model of Vadas et al. (2005 and 2009) to estimate P loss in surface runoff from a range of source conditions consistent with P Index assessment for each state. This spreadsheet operates on an annual time step and is appropriate to evaluate the source components of a P Index for a user-defined set of runoff and erosion conditions. The spreadsheet does not itself predict runoff or erosion.

- Agricultural Policy Environmental eXtender (APEX; Gassman et al., 2009), which is a daily time step model that predicts runoff, erosion, and P loss for a user-defined set of field, management, and weather scenarios. APEX has been run as part of the Conservation Effects Assessment Project (CEAP). More than 22,000 sites across the nation have been modeled. The NRI sites could serve as evaluation points for the model, and where appropriate, can be used as actual data points for evaluating a P Index.
- Where locally calibrated / validated models are available, such as the quantitative P loss assessment tool for agricultural fields developed by White et al. (2010), their use would be appropriate.

This approach should be used to evaluate P Indices across the country to determine the directional and proportional integrity of P Indices with increasingly "risky" management scenarios. The model used must appropriately simulate the P loss processes under evaluation. For example, a model without a well-developed manure application or P leaching routine may not be appropriate for assessing the risk of P loss from surface applied manures or artificially drained soils, respectively. Regardless of the model used, conditions must still be defined that result in both unacceptable P loss within the model and high or very high P Index ratings that limit or preclude P applications run under the same set of conditions. Comparisons could be based on P loss estimates from the model but would not depend on any particular quantitative result for the P Index being evaluated as many P Indices are qualitative tools.

The primary criteria for comparison would be that the model and the P Index agree directionally and proportionally for an appropriate range of management, runoff, and erosion conditions. For use in regulatory programs, it is likely that more rigorous statistical criteria will need to be developed for this comparison. This evaluation approach would allow the use of existing P Indices as long as they meet the evaluation criteria. This approach can also be used to identify and support changes to existing P Indices to improve the assessment and could help in designing a new P Index. It is important to note however, that use of any model to evaluate a P Index does not imply use of the model as an alternative to existing P risk assessment tools / P Indices.

Because of the innate variability of natural systems, methods should be developed to estimate the uncertainty in predictions by P-indices and models. An example of a tool that could be used for this is @RISK commercial software which is a plug-in for Excel spreadsheets (http://www.palisade.com/decisiontools_suite/). Uncertainty in predictions should be considered when using models to test P Indices.

CHARGE 5

DEFINE LONG-TERM GOALS FOR DEVELOPMENT OF NEXT GENERATION P INDICES

Recommendations

1. Development of a National P Risk Assessment Tool should be considered. Information needed to represent all situations, soils, management, physiographic settings, etc., must be compiled. This will require a major investment of resources and infrastructure, particularity for a reliable representation of landscape hydrology, surface runoff and leaching generation, and flow pathways.
2. NRCS should use a P loss assessment approach based on physiographic regions or NRCS Major Land Resource Areas (MLRAs) rather than national or state boundaries.
3. Next generation Indices should be constructed on a GIS platform to facilitate integration of current and future information databases.
4. There needs to be a concerted training effort on how to use P Indices in the context of nutrient management planning and how to address any concerns identified by the P Index used during the plan development/implementation process.

Considerations

The initial P Index ranked transport and source factors and added them together (Lemunyon and Gilbert, 1993). Because individual states were allowed to write their own NRCS 590 standard and modify the original P Index to address local priorities and conditions, there are large structural variations in P Indices. In addition, each state's P Index was developed for a slightly different purpose, and thus variations between them are apparent. Most states have made one or more of the following changes to the original design and formula proposed by Lemunyon and Gilbert (1993): 1) source and transport factors are multiplied rather than added; 2) distance from water resources is considered; and 3) some factors, such as soil loss, STP and P application rate, are quantified continuous inputs (Sharpley et al., 2003).

Developing a National P Index

We currently do not have the science, technologies, hydrological models, political will, resources, or infrastructure to implement a single approach to P loss risk assessment that covers all situations, soils, management, and physiographic settings. It would take an effort similar to that invested in USLE to develop and implement a national P risk assessment tool. There are several important factors influencing categorization and interpretation of P Index risk assessment, which vary greatly among states. This variation influences the outcomes and management recommendations as a result of an Index assessment and many are independent of the functionality of Indices in general. These factors include the spatial and temporal resolution and representation of Indices, multiplicative versus additive approaches, and state fertilizer recommendations. While some of this variability can be addressed during the Index revision process, external factors will have to be evaluated separately.

Spatial Representation

Most P Indices are state specific. This is primarily due to the requirements of state regulations and state 590 standards. Predominant mechanisms of P loss vary widely depending on soil and climate conditions, which are certainly not uniform across the country and rarely follow state boundaries. Consequently physiographic regions would be the more logical basis for regionalization of P Indices than state boundaries.

In the Chesapeake Bay watershed for example, which only represents a small area of the country, there are five main distinctly different physiographic regions; Coastal Plain, Piedmont, Great Valley, Appalachian Mountains, and Appalachian Plateau (Figure 2). Most of the states in this watershed contain three or more of these physiographic regions. It is very difficult to develop a practical P loss assessment tool that will work equally well for all these physiographic regions. Consequently, compromises are often necessary which are usually less than ideal in any of these regions.

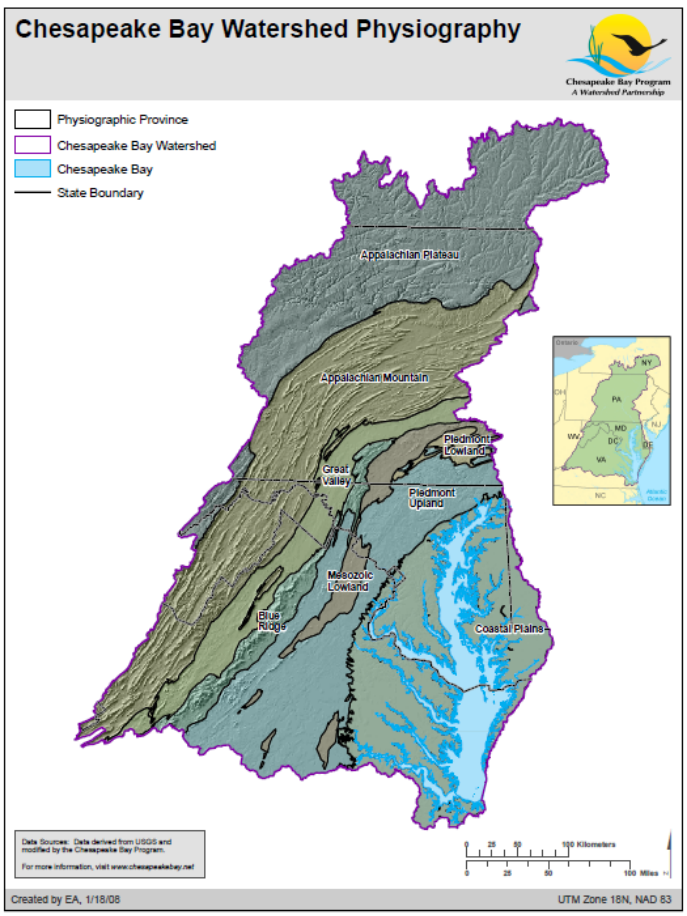


Figure 2. Physiographic regions of the Chesapeake Bay Watershed.

For example, how do you develop a P risk assessment tool that adequately addresses the predominantly leaching-driven losses of P in the Coastal Plain, where erosion is only a minor mechanism and the predominantly erosion- and runoff-driven losses in the Appalachian Mountains where leaching is much less of a factor? Indices in Maryland and Virginia attempt to do this. Because of these widely varying conditions and different relative areas of these physiographic regions in these two states, the approach to compromise varies enough that there are often significant differences in the P loss risk assessments from these states even on the same field.

Thus, in the Chesapeake Bay Watershed for example, a better approach would be to have an Index for each of the physiographic regions rather than one for each state (i.e., Delaware, Maryland, New York, Pennsylvania, and Virginia). These would be specifically tailored to the soils, climate, and management systems in these regions and be used within each physiographic region across all of the states. The challenge is to get acceptance within government programs of P Indices that cross state lines. States are generally reluctant to base regulations on something that they do not completely control.

GIS and Database Interfacing

The NRCS and EPA require the use of the Revised Universal Soil Loss Equation, Version 2 (RUSLE2) to determine soil erosion when developing nutrient management plans (NMPs). The standard approach to estimating a crop field's soil loss with RUSLE2 involves selecting a single soil type in the field. If the field has more than one soil type, the field's "dominant critical area" is supposed to be used as a “surrogate” to determine soil loss for the entire field in the conservation plan. However, the dominant critical area soil may not be the predominant soil in the field and it may not be the soil that should be used in making nutrient recommendations or in assessing the risk of nutrient and sediment loss from the field. A "spatial" approach to estimating soil loss for a field with RUSLE2 involves estimating soil loss for all digitized soil survey polygons whose boundaries overlap with the field's boundary. This would eliminate the need to select a single soil for a field to run RUSLE2, while allowing traditional conservation planning to be done on the basis of a single soil. Similarly, the P Index could be also calculated for each soil polygon in the field, using each polygon's underlying soil properties as inputs to the P Index.

Training and Support

Next generation P Index development plans need to include funding and resources to ensure effective implementation and long term support for the tool that is developed. Resource requirements for implementation are likely to be greater than those for initial development. An on-going training effort for NRCS staff, technical service providers and farmers on the use of the P Index in nutrient management planning will be needed. Planners and farmers need to understand the P Index as an indicator of P loss risk to find appropriate solutions to high P loss areas during the planning process and to be able to make appropriate adjustments when needed as the plan is implemented.

To be effective, any P loss assessment tool must be completely integrated with the nutrient management planning process. Nutrient management takes place in an agricultural landscape that is constantly changing, and ongoing funding for updates will be needed to maintain this integration. This will be especially true of assessment tools using computer software.

APPENDIX A

CURRENT STATE OF LAND-GRANT UNIVERSITY NUTRIENT RECOMMENDATIONS

Agronomic soil testing for P has been conducted for many years. These tests were initially developed to identify soils where plant-available P is insufficient to support maximum crop growth and where further addition of fertilizer was not needed. In many situations, P may not be recommended where the relative yield is >95% of the maximum yield or the likelihood of crop response to applied P is less than 5%. Soil test P where no additional P is recommended will vary with soil properties, crop type, and yield goal. Also, many states include a crop removal recommendation for STP just above this crop response critical level, as most farmers only test their soils periodically (every 2 to 5 years). This is to ensure that STP levels will not drop below the crop response critical level between soil tests. Soils are typically categorized (i.e., Very Low P, Low P, or below optimum P; Sufficient, Moderate P, or optimum P; High P, Very High P or above optimum P) based on the probability of crop response to additional P.

Soil testing to assess the potential environmental impact of P is a relatively recent development. Agronomic soil P tests were developed to assess the potential for crop response to applied P. The crop response categories / agronomic interpretations should not be equated to environmental risk interpretations. A number of tests and relationships of these P tests with runoff P have been developed for this purpose. However, there are too many other variables independent of soil P, such as P application, runoff and erosion potential, and distance to a stream or concentrated flow channel, for agronomic STP to be used as the sole indicator of the risk for P loss from a field.

Most P fertilizer recommendations for crops were established by scientists associated primarily with land-grant universities. Much of this work was done when commercial fertilizers first became widely available beginning in the 1950's. In the recent past, much less emphasis has been given to this type of research by public institutions and once-common publicly funded soil testing laboratories are now rare. This can be problematic when government programs refer to university recommendations for a standard but the land-grant university can no longer support soil test calibration research and updates. Thus, updating nutrient recommendations should be supported as new crop varieties and yield response data become available.

APPENDIX B

RELATING P LOSS IN RUNOFF TO SOIL TEST P, SOIL P SATURATION AND P INDEX RISK

There is no scientific evidence to support the use of STP or soil P saturation alone to determine the amount of P loss from a field. A wealth of scientific evidence is available documenting that STP and/or soil P saturation are one of several factors influencing the risk of P loss from a field. Use of STP or soil P saturation alone will not capture a site's risk for P loss and may be less restrictive than a well designed P Index, thereby increasing the potential for P runoff and leaching (Figure 3). The data in Figure 3 is from the FD-36 watershed on south-central Pennsylvania and is adapted from that presented in Sharpley et al. (2001). Runoff was collected from 2-m^2 plots subject to 70 mm hr^{-1} rainfall (to create 30 minutes of runoff) across the watershed and related to plot Mehlich-3 STP and soil P saturation of 0 to 5 cm samples collected after rainfall, as well as P Index ratings determined by the Pennsylvania P Index (Sharpley et al., 2001). Of the three methods, the P Index rating best represented the loss of P in runoff over the various soil, management, hydrology, and topographic conditions across the watershed (Figure 3).

More importantly, there were sites with "low" STP and soil P saturation, which had high losses of P due to a combination of factors that include high runoff volumes and / or application of fertilizer or manure. It should be noted that these "low" P sites are above the agronomic response range (i.e., >50 mg P kg^{-1} as Mehlich-3 soil P). On the other hand, there were sites with low P loss but had high STP or soil P saturation values (Figure 3). A similar lack of a strong relationship between STP and runoff P loss was demonstrated by Butler et al. (2010) for runoff from several fields in Georgia, which had received varying amounts and forms of P (Figure 4).

In summary, we recognize that the relationship between STP or P saturation and runoff dissolved P concentration is well established (e.g., Vadas et al., 2005). However, this relationship can vary as a function of soil type and land cover, and P loss is influenced by many site factors such as applied P (type, rate, method, and timing) runoff, erosion, landscape position, etc. Further, use of soil P saturation in place of STP is only suitable for noncalcareous soils where Fe and Al dominate soil P reactions. In light of these factors, it is inappropriate to use STP or soil P saturation alone to estimate P loss in runoff from a given site.

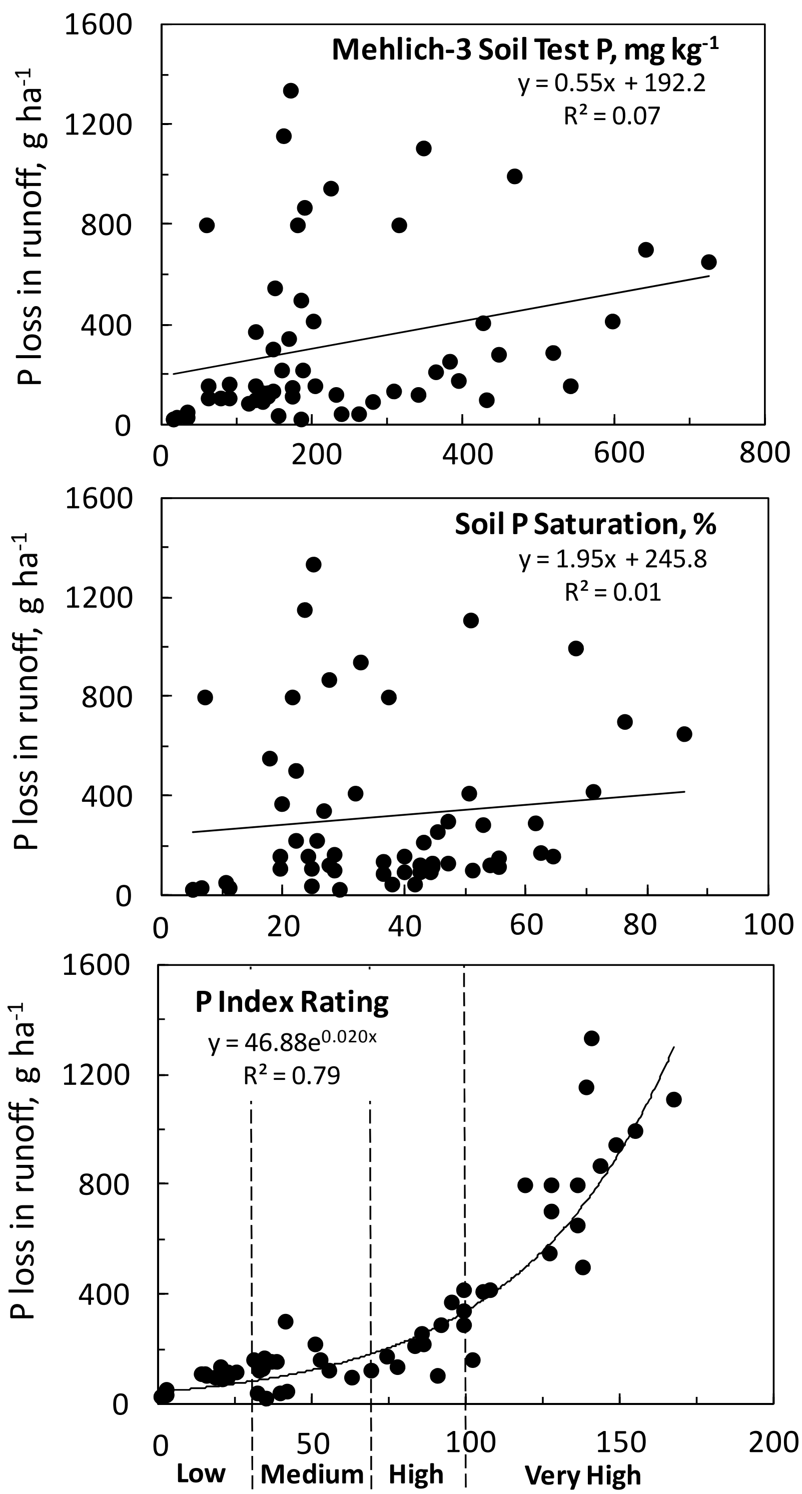


Figure 3. Relationship between the loss of total P in runoff and Mehlich-3 soil test P, soil P saturation, and the Pennsylvania P Index ratings for the plots in the FD-36 watershed, PA (adapted from Sharpley et al., 2001).

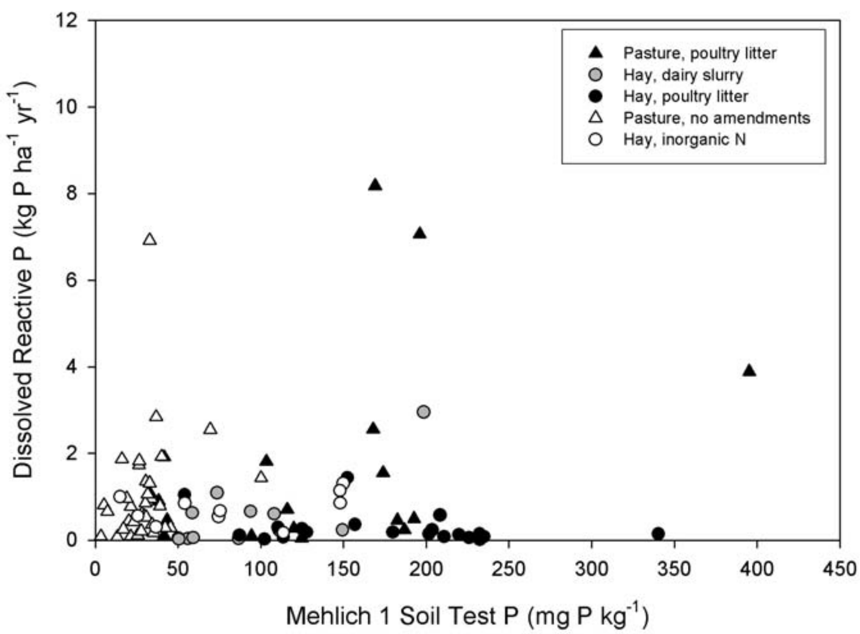


Figure 4. Relationship between Mehlich-1 soil test P and the loss of total P in runoff for several fields in Georgia (adapted from Butler et al., 2010).

REFERENCES

Butler, D.M., D.H. Franklin, M.L. Cabrera, L.M. Risse, D.E. Radcliffe, L.T. West, and J.W. Gaskin. 2010. Assessment of the Georgia Phosphorus Index on farm at the field scale for grassland management. J. Soil Water Conserv. 65:200-210.

Gassman, P.W., J.R. Williams, X. Wang, A. Saleh, E. Osei, L.M. Hauck, R.C. Izaurralde, and J.D. Flowers. 2009. The Agricultural Policy Environmental EXtender (APEX) Model: An emerging tool for landscape and watershed environmental analyses. Technical Report 09-TR 49. Center for Agricultural and Rural Development, Iowa State University, Ames, IA. http://www.card.iastate.edu/publications/DBS/PDFFiles/09tr49.pdf

Harmel, R.D., S. Qian, K. Reckhow, P. Casebolt. 2008. The MANAGE database: Nutrient load and site characteristic updates and runoff concentration data. J. Environ. Qual. 37:2403-2406.

Lemunyon, J.L., and R.G. Gilbert. 1993. Concept and need for a phosphorus assessment tool. J. Prod. Agric. 6(4):483-486.

Osmond, D.L., M.L. McFarland, R. Koenig, and D.B. Beegle. 2006. Phosphorus management within watersheds that cover multiple states. SERA-17 Phosphorus Management and Policy Workgroup: Position Papers on Key Scientific Issues. SERA-17 Organization to Minimize Phosphorus Loss. http://www.sera17.ext.vt.edu/Documents/Position_Papers_Introduction.pdf.

Sharpley, A.N., R.W. McDowell, J.L. Weld, and P.J.A. Kleinman. 2001. Assessing site vulnerability to phosphorus loss in an agricultural watershed. J. Environ. Qual. 30:2026-2036.

Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.J. Gburek, P.A. Moore, Jr., and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the United States. J. Soil Water Conserv. 58(3):137-151.

Vadas, P.A., P.J.A. Kleinman, and A.N. Sharpley. 2005. Relating soil phosphorus to dissolved phosphorus in runoff: A single extraction coefficient for water quality modeling. J. Environ. Qual. 34:572-580.

Vadas, P.A., L.W. Good, P.A. Moore, Jr., and N. Widman. 2009. Estimating phosphorus loss in runoff from manure and fertilizer for a phosphorus loss quantification tool. J. Environ. Qual. 38:1645-1653.

Vieth, T. L., A.N. Sharpley, J.L. Weld, and W.J. Gburek. 2005. Comparison of measured and simulated phosphorus losses with index recommendations. Trans. Am. Soc. Agric. Eng.. 48(2):557 565.

White, M.J., D.E. Storm, P.R. Busteed, M.D. Smolen, H. Zhang, and G.A. Fox. 2010. A quantitative phosphorus loss assessment tool for agricultural fields. Environmental Modelling and Software 25:1121-1129.