The Impact of Concentrated Animal Feeding Operations on Groundwater Quality in Private versus Public Wells [*]

Andrew Meyer [†] Zach Raff [‡]

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Abstract

While previous studies have associated concentrated animal feeding operations (CAFOs) with surface water contamination, causal links to groundwater pollution remain understudied. We address this gap in the literature, assessing the impact of CAFOs on both public and private well water quality in Wisconsin. We focus on nitrate contamination, which poses health risks when consumed in drinking water. We spatially link wells to CAFO locations and determine a possible exposure buffer of 1 mile, which we use to divide wells into treatment and control groups. Using difference-in-differences methods, we find that CAFO presence significantly increases nitrate concentrations in nearby private wells, with limited effects on public wells. We find the largest impacts on private wells that are shallow or located in areas with shallow carbonate bedrock. Our findings underscore the disproportionate burden of groundwater pollution on rural, lower-income households reliant on private wells, with implications for environmental justice and public health.

JEL Classification: D63, Q18, Q53
Keywords: groundwater pollution, concentrated animal feeding operations, environmental justice, public health, nitrates

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[†] Marquette University; andrew.g.meyer@marquette.edu

[‡] USDA Economic Research Service; zachary.raff@usda.gov

1 Introduction

Concentrated animal feeding operations (CAFOs) are intensive livestock facilities that generate large quantities of animal waste in small geographies. By definition, CAFOs have over 1,000 animal units onsite in covered facilities for at least 45 days per year (US Environmental Protection Agency 2012).[1] Previous work documents the negative impact of CAFOs on surface water quality (Heaney et al. 2015; Raff and Meyer 2022; Skidmore et al. 2023; Weldon and Hornbuckle 2006), air quality (Sneeringer 2009; Sousan et al. 2021), and property values (Herriges et al. 2005; Jowers et al. 2025; Kim and Goldsmith 2009).

CAFOs can also contribute to groundwater pollution through a variety of mechanisms, including direct manure application to fields, spills and leaks from storage facilities, and runoff from fields. There is some cross-sectional evidence of correlations between CAFOs and other animal feeding operations and groundwater quality (Lockhart et al. 2013; Zirkle et al. 2016). However, to date, no study has rigorously evaluated the causal impact of CAFOs on groundwater quality using quasi-experimental methods. Our study aims to fill this gap in the literature by estimating an ex post relationship between longitudinal CAFO exposure and groundwater quality on a large spatial scale.

Furthermore, the extant literature does not systematically analyze the differential effect of CAFOs on public and private wells. This is important because there is potential for CAFOs to affect private wells differently than public wells. Private wells are often located closer to CAFOs (in agricultural, rather than urban/suburban settings) and are not subject to the same regulatory oversight as public wells. For example, in some areas of Wisconsin near CAFOs, 30% or more of private wells are contaminated with bacteria or nitrates (Borchardt et al. 2021; Wisconsin Groundwater Coordinating Council 2024). But these same areas also have other forms of agriculture that could be at least partially responsible for groundwater contamination (e.g., row crop production), obfuscating causal attribution. In this study, we use a difference-in-differences (DID) approach to estimate the differential effect of CAFOs on public and private well water quality.

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[1] An animal unit is the equivalent of 1,000 pounds of live animal weight. One thousand animal units is roughly equal to 700 dairy cows, 2,500 swine, 125,000 broiler chickens, or 82,000 laying hens. In many states, AFOs smaller than 1,000 animal units can also be considered CAFOs, depending on their wastewater discharge or manure spreading practices.

We focus on the state of Wisconsin for our application. There are several reasons why Wisconsin provides a useful setting for addressing our research question. First, there is substantial CAFO presence in the state. And the CAFOs are spread across many regions within Wisconsin. As of 2023, there were over 300 CAFOs operating under permits from the Wisconsin Department of Natural Resources (WDNR). Second, Wisconsin has experienced substantial growth in CAFO operations over the last several decades, providing ample temporal variation to identify how changes in CAFO intensity affect changes in groundwater quality. Lastly, groundwater is an important source of drinking water in Wisconsin: nearly 67% of the state's population receives their drinking water from the ground. By comparison, this value is roughly 38% for the US as a whole. There is therefore substantial policy concern regarding potential adverse health effects.

To address our research question, we spatially link CAFO locations and monitored wells, drawing buffer zones of varying diameters around each CAFO. We then merge publicly available data from WDNR to assess groundwater quality, focusing on nitrate levels, as CAFO exposure is most likely to impact this contaminant and nitrate pollution has been shown to affect human health (Mathewson et al. 2020). We find that CAFOs substantially increase nitrate concentrations in nearby private wells but have little discernible effect on nitrate concentrations in the average public well. In addition, our analysis shows that the effects are largely concentrated on private wells within one mile of CAFOs.

We then turn to a binary DID analysis where we split wells into treatment and control groups based on the one-mile data-driven exposure buffer. Across several specifications, we find that a CAFO locating within 1 mile of a private well increases average nitrate concentrations by 50-56%. At baseline nitrate concentrations, this implies that a nearby CAFO could plausibly raise many wells from under the WDNR advisory level to above the advisory level. The advisory level is set according to scientific evidence of adverse health consequences, so the magnitude of our findings have public health implications. Additionally, for shallow private wells, baseline nitrate concentrations are higher and estimated effects of CAFOs are larger than those we find for the average well.

We present a series of results to further enhance the credibility of our private well findings. First, an event study for private wells shows no differential pretrends for treatment wells versus control wells. Second, we show that the private well results are robust to the heterogeneous treatment effect estimators of Wooldridge (2021) and Borusyak et al. (2024). Third, we rule out several potential concerns related to nitrate test selection in the private well sample. Fourth, we implement a placebo outcome test on a groundwater contaminant that should not be affected by CAFOs and a placebo treatment test using a treatment that should not affect nitrate concentrations (CAFOs beyond 10-20 miles away). Finally, we show that treatment effects are substantially larger in areas where we would expect groundwater to be more susceptible to contamination based on geology (aquifers in areas with shallow carbonate bedrock).

There are environmental justice implications from our finding that CAFOs affect private wells more than public wells. To further explore differences between private and public wells, we link census tract characteristics to each well, using geospatial data about community water systems for municipal wells. We find that, compared to public wells, private wells are located in areas that are more rural. Among private wells, we show that shallower private wells (which are much cheaper than deeper private wells) tend to be in areas that are more rural, with lower median income, lower education, and a lower minority percentage than the average private well.

As noted by Cain et al. (2024), most of the environmental justice literature focuses on air pollution, with substantially less research on water pollution. “To our knowledge, the impact of water pollution on environmental justice remains a gap in the literature” (Cain et al. 2024). In this research, we take a first step towards quantifying systematic differences in exposure to groundwater pollution. Policymakers and regulators should recognize that economic damages of groundwater contamination will not be shared equally across residents.

2 Background

In this section, we cover important background information for our study. We first explain how water becomes groundwater and how groundwater can become contaminated. Next, we provide background information on CAFOs, highlighting their growth and the potential associated environmental concerns. We then explain how well depth potentially relates to groundwater quality. Finally, we discuss nitrates, providing rationale for the public health importance of these contaminants, and describing the relevant regulatory environment.

2.1 Groundwater and sources of contamination

Groundwater begins with precipitation that soaks into pervious ground. Gravity then pulls the water down through the unsaturated upper layer of soil, past the water table and into the saturated zone, where water fills all pores between soil particles. At this point, in the saturated zone, water is known as groundwater. The body of rock or sediment that holds groundwater is called an aquifer. To access groundwater, one drills a well into the aquifer.

Groundwater contamination occurs through various human activities and natural processes. As water travels from the surface to the saturated zone, it can pick up contaminants from the surface or within the unsaturated zone. Point sources of contamination include leaking underground storage tanks, septic tanks, industrial facilities, landfills, and agricultural activity such as barnyards or feeding operations. Additionally, agricultural practices such as the spreading of fertilizers and pesticides contribute to nonpoint source contamination when these chemicals leach through soil into the groundwater. Other nonpoint sources of contamination include urban runoff from roads and developed land. Lastly, natural processes such as erosion or the dissolution of minerals from geological formations can introduce contaminants into groundwater. Depending on geology and soil characteristics, the soil can filter out some contaminants before they reach the saturated zone.

Once water reaches the saturated zone, it will often move laterally in addition to moving deeper into the aquifer. Groundwater will generally flow from upland areas to lower elevation areas and eventually seeps into surface water such as rivers, lakes, and streams. The speed that groundwater moves can vary widely depending on several factors, including soil characteristics, the hydraulic gradient, and the presence of fractures or conduits in the sediment or rock. In general, groundwater moves much slower than surface water, typically ranging from inches (in material like clay) to feet (in porous materials like sand or gravel) per day (Mechenich and Shaw 1996). Further filtration of contaminants typically occurs as water moves deeper within the saturated zone.

2.2 Concentrated Animal Feeding Operations

In recent decades, there has been a shift in animal distribution away from traditional, small-scale operations to larger, more concentrated AFOs. As a result, the number of livestock farms has decreased despite relatively consistent livestock inventories (Coplean 2010; Sneeringer 2009). This transition has spurred a notable rise in CAFO numbers across the US and within Wisconsin. Figure 1 shows estimated historical numbers of CAFOs across the entire US[2] and Figure 2 depicts Wisconsin's CAFO growth over time. Although CAFOs represent a minority of total AFOs in Wisconsin, they house an increasing percentage of livestock. In 2019, CAFOs accommodated nearly 25% of Wisconsin's dairy cows despite accounting for only 3.5% of the state's dairy operations (Cushman 2019).

Wisconsin's CAFOs are dispersed throughout the state, although some counties and regions have higher CAFO intensity than others. Figure 3 shows the geographic distribution of CAFOs in Wisconsin in 2023. Several counties in the northeast corner of the state bordering Green Bay and Lake Michigan have the highest concentrations of CAFOs. These counties include Brown, Kewaunee, Manitowoc, and Outagamie counties. In general, there are not as many CAFOs in the immediate vicinity of larger cities such as Milwaukee, Green Bay, and Madison.

The expansion of CAFOs poses risks to both human health and the environment due to the concentrated nature of the waste that they produce. For instance, a dairy farm with 1,200 cows can produce over 30,000 tons of manure per year, equivalent to the annual sanitary waste of a US city with 46,000 residents (GAO 2008). Many Wisconsin CAFOs have significantly more than 1,200 dairy cows. In the US, CAFOs contribute over 4,000,000 metric tons of nitrogen and 1,400,000 metric tons of phosphorus to agricultural lands (Glibert 2020).

Contaminants stemming from CAFO manure can reach groundwater through various channels. CAFOs typically manage manure in two stages. First, they store manure onsite. Dairy and swine operations commonly store liquid or slurry manure in surface lagoons, pits

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[2] It is difficult to determine the exact number of CAFOs for the entire US because many states have not historically tracked the numbers, many states do not issue NPDES permits for large numbers of CAFOs, and EPA information is incomplete. We combine data from GAO and EPA to create the estimates for this figure.

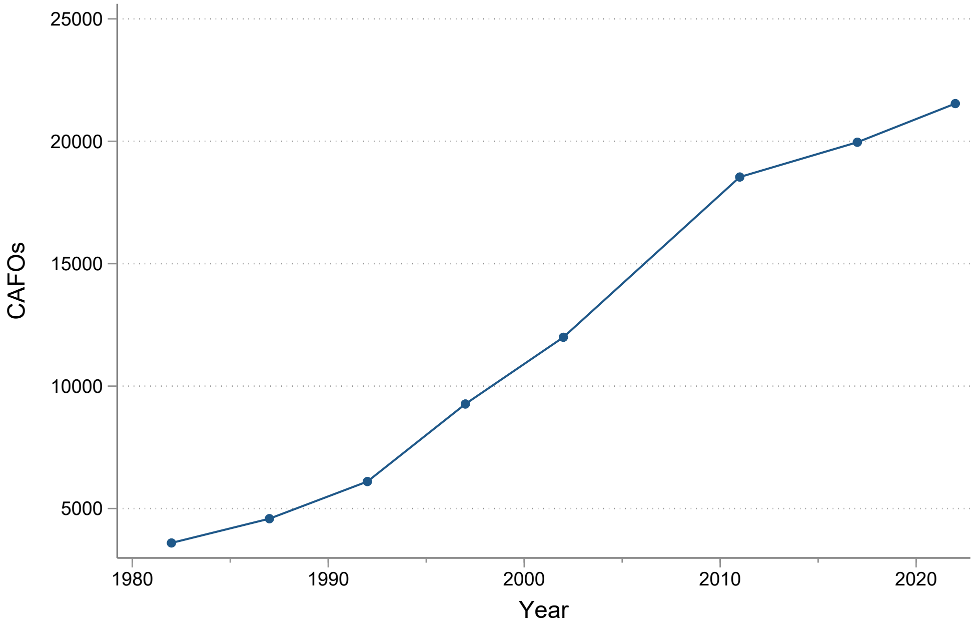

Figure 1: Estimated CAFOs in the U.S.

Notes: This figure shows the estimated number of CAFOs across the United States and is based on authors' calculations using data from the US GAO and EPA.

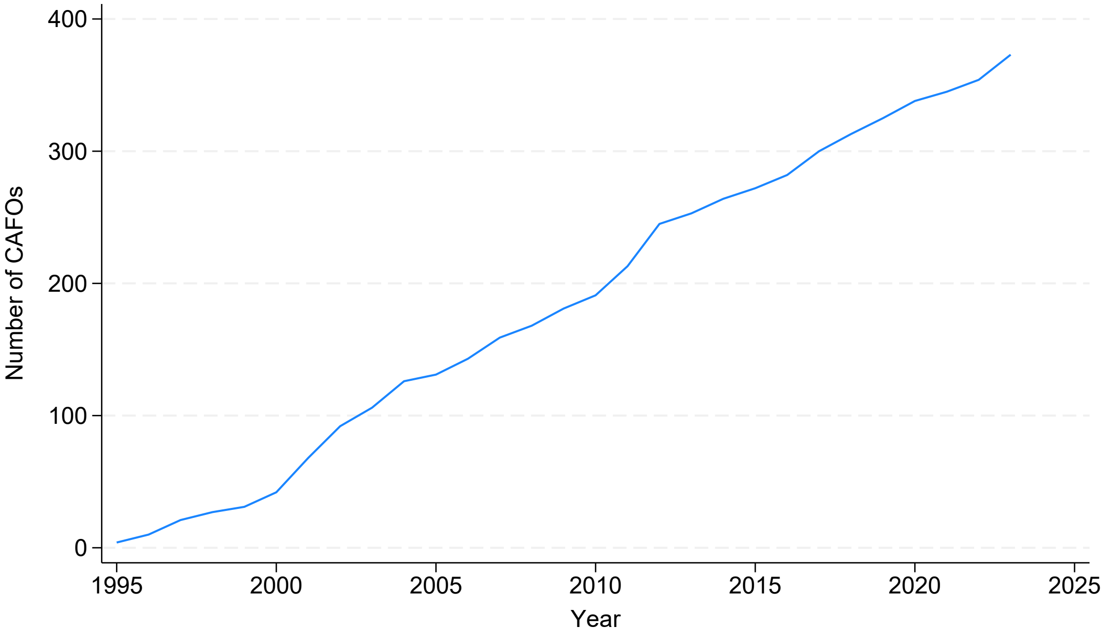

Figure 2: Wisconsin's CAFO expansion

Notes: This figure shows the number of permitted CAFOs in Wisconsin and is based on authors' calculations using data from Wisconsin DNR.

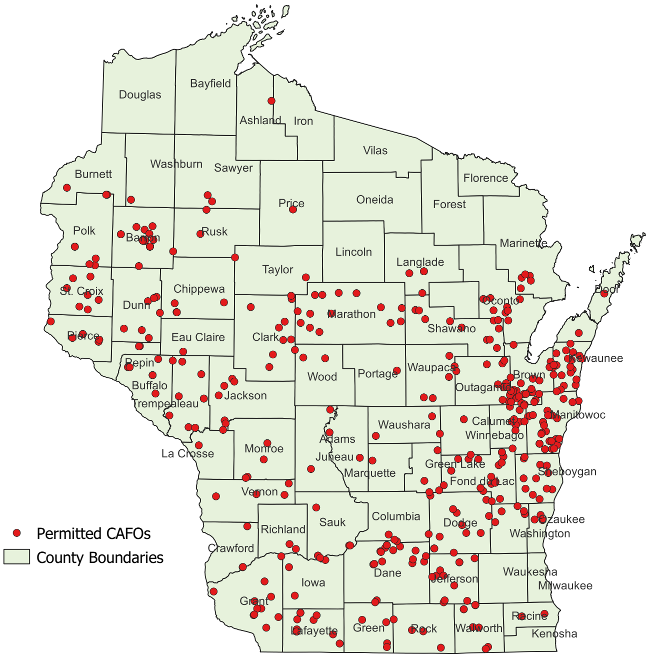

Figure 3: Wisconsin's CAFOs

Notes: This figure shows the location of permitted CAFOs in Wisconsin at the end of our sample period (2023) and is based on data from Wisconsin DNR.

(under or outside the barns), or slurry tanks. Poultry operations store dry litter outdoors under tarps or in buildings. Second, after storage, manure is most often spread onto nearby farmland or incorporated into the soil, sometimes at inappropriate rates or times (Osterberg and Wallinga 2004; Meyer et al. 2024). Groundwater contamination can occur during either stage. In the first stage, manure lagoons and pits are often insecure and lack linings or retaining walls, leading to runoff during precipitation events or leaching into groundwater (Hribar 2010; Waller et al. 2021). In the second stage, if manure is spread onto farmland at inappropriate rates or times (e.g., on frozen ground, without plant cover), contaminants could leach through the soil and into groundwater. Compared to land application, incorporating manure into the soil allows the soil to better retain manure nutrients (Bierer et al. 2021) and reduces the risk of runoff during precipitation events (Saha et al. 2023), but incorporation is more likely to lead to nutrients leaching into groundwater (Dell et al. 2012).

The application of manure and other fertilizers presents specific hazards in regions characterized by sandy, highly permeable soils (i.e., karst geology), where surface water rapidly infiltrates to impact groundwater. Many areas with these characteristics have seen significant increases in the contamination of wells used for drinking water (Borchardt et al. 2021; Erb et al. 2015; Nicole 2021). Even in regions with deeper soils, excessive manure and fertilizer usage pose a significant threat to the quality of both surface and groundwater (Craswell 2021).

The Clean Water Act statutorily regulates CAFOs as point sources of water pollution. The EPA updated the National Pollutant Discharge Elimination System (NPDES) in 2003, requiring CAFOs to obtain permits and develop nutrient management plans (Sneeringer and Key 2011; Chen et al. 2019). However, NPDES permits do not directly regulate the amount of animal waste spread onto fields, and nutrient management plans often prove inadequate to control excess manure nutrients from CAFOs (Chen et al. 2019). Manure in slurry form is heavy, so it is expensive to transport and distribute. Therefore, manure spreading typically occurs within a relatively small radius (typically <1-3 miles in Wisconsin).[3] Moreover, there is significant variation in state-level regulation, leading to unpermitted CAFOs in certain states, exacerbating pollution risks (GAO 2003; Meyer et al. 2024).

2.3 Well depth

In general, shallower wells draw from a smaller, more proximate land area and deeper wells draw from a larger, potentially more remote area. However, even for the deepest wells, there is a limit to the distance that groundwater will flow. In Wisconsin, most groundwater from private wells originates from within a few miles (Mechenich and Shaw 1996). “Shallow” and “deep” are relative qualifiers that can vary based on the context, especially the depth of the water table. Typically, shallow wells are more susceptible to contamination (Nolan and Hitt 2006).

Another important factor to consider is the well casing, which is the pipe that lines the well hole. The depth of a well casing can vary substantially. In sand and gravel aquifers,

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[3] A 2014 survey of Minnesota farmers reports that they transfer liquid manure an average of 0.82-1.63 miles from the barn to the field, depending on the region of the state (Minnesota Department of Agriculture 2017). Minnesota is adjacent to Wisconsin and has similar agricultural practices. Nearly all CAFOs in Wisconsin are dairy, which produce and store predominantly liquid manure.

the casing is typically nearly as deep as the well hole to prevent sediment from collapsing back and closing the hole. In granite or sandstone aquifiers, wells may only have a casing a portion of the depth of the hole because it is unlikely that the rock collapses and closes the hole. In this case, the well casing depth may be more relevant than the overall well depth in determining the groundwater source of a well.

2.4 Nitrate pollution

Excessive nitrate levels in drinking water can pose serious public health risks, particularly for infants, pregnant women, and individuals with certain medical conditions. High nitrate intake can lead to methemoglobinemia, commonly known as “blue baby syndrome”, a condition where the oxygen-carrying capacity of blood is reduced. This turns the skin a blue color and may lead to a rapid heartbeat, shortness of breath, lethargy, seizures, or even death in severe cases (Wisconsin DHS 2023). Additionally, nitrates can react with organic matter to produce compounds linked to various cancers, including stomach and colorectal cancers. The direct medical costs from nitrate-attributable adverse health effects are substantial; Mathewson et al. (2020) estimate an annual cost of $23-$80 million for Wisconsin residents.

The EPA has set enforceable standards for public drinking water supplies. The maximum contaminant level (MCL) for nitrates is 10 mg/L (roughly 10 parts per million). The EPA believes that exposure below this level is safe for everyone, including infants and other sensitive populations. In contrast, there are no national enforceable standards for nitrates in private wells. The WDNR provides recommended guidelines for nitrates in private wells, but there does not exist a specific regulation or enforceable MCL set by the state. This advisory level for nitrates is set at 10 mg/L.

3 Data

This study uses data from several sources, all originating from the state of Wisconsin. First, we obtain via open records requests multiple files containing permit information on CAFOs. Permit information includes the CAFO name, address, number and type of permitted animals, and relevant permit dates. With these files, we construct a panel of the historical universe of permitted CAFOS in the state. This panel updates Raff and Meyer (2022) with permits through 2023.

Next, we obtain publicly available groundwater quality data from Wisconsin's Groundwater Retrieval Network (GRN) and Wisconsin's Public Drinking Water System Portal. GRN contains well characteristics for public drinking water supplies and for private drinking water wells. GRN also consolidates water quality testing results from various sources for private wells. The Public Drinking Water System Portal provides water quality testing results for public water supplies. For public wells, we merge the well characteristics from GRN with contaminant results from the Public Drinking Water System Portal. We limit the public well sample to those classified as "municipal community" wells.[4]

Both groundwater quality sources provide geographic information in the form of Public Land Survey System (PLSS) descriptions. In this system, a township and range combination identifies a 36-square mile parcel. The township number indicates the number of cells north of the Wisconsin-Illinois border. The range number identifies the number of cells east or west of the principal meridian. These parcels are then further divided into 36 sections, each approximately one square mile in area (not exact due to the curvature of the Earth). Thus, a township-range-range direction-section combination uniquely identifies a one square mile area of Wisconsin; the Wisconsin groundwater quality sources provide the township, range, range direction, and section.

We obtain geocoded files for Wisconsin's PLSS and plot this grid in a geographic information system (GIS). Figure 4 shows this grid. We then match each private and public well to its corresponding township-range-range direction-section. For illustration, Figure 5 shows the sections that have at least one nitrate sample at a private well. Next, we geocode each permitted CAFO in our panel. Again, for reference, Figure 6 layers the CAFOs that operate at least one year during our sample period.

Conceptually, a CAFO could plausibly affect groundwater quality several miles away since CAFOs typically spread manure within a several mile radius from the animal barns.

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[4] These are wells from a water system owned by a municipality which serves year-round residents, and hence best represents wells used for primary public drinking water supplies.

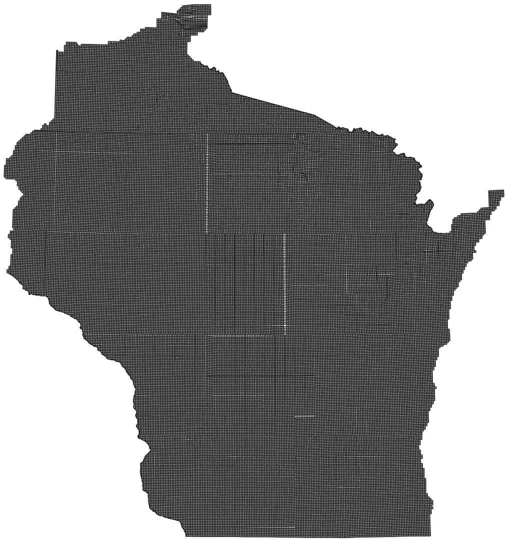

Figure 4: PLSS grid

Notes: This figure shows Wisconsin's PLSS grid. Each square on the grid is approximately 1 square mile.

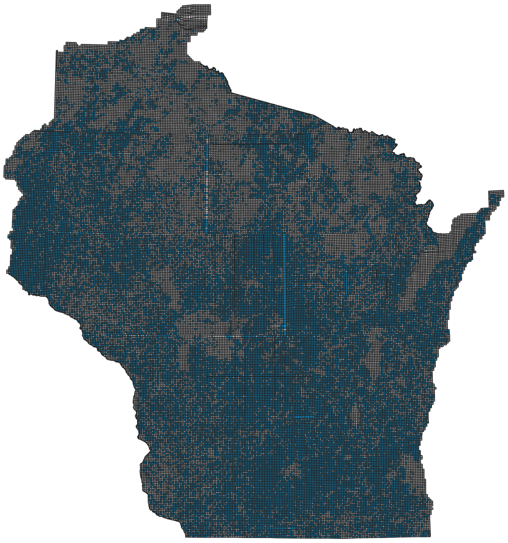

Figure 5: Nitrate private well sample on the PLSS grid

Notes: This figure shows Wisconsin's PLSS grid, highlighting (in blue) the sections with at least one nitrate observation from a private well.

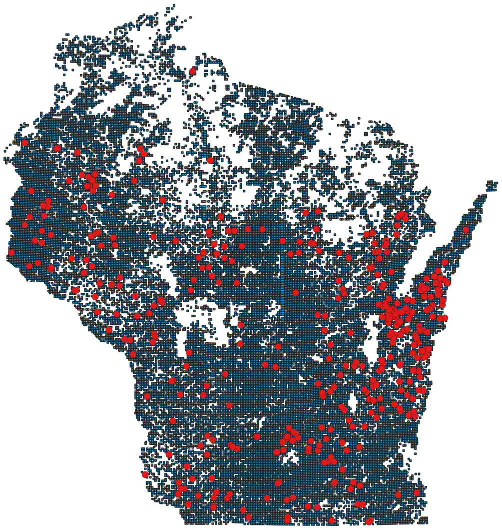

Figure 6: Nitrate private well sample with CAFOs

Notes: Shaded areas represent sections with at least one nitrate observation from a private well. Red dots represent permitted CAFO locations in 2023.

However, it is also possible that any groundwater impacts could be stronger in the immediate vicinity of a CAFO due to runoff from barns or leakage from manure holding structures. It is ultimately an empirical question, so we construct buffers of varying radii around each CAFO (1 mile, 3 mile, 5 mile, and 10 mile radii). Figure 7 shows a zoomed in view of this spatial matching. In this figure, each red circle represents a 1 mile buffer around a CAFO (diameter of 2 miles) and each square represents a PLSS section. Blue squares have at least one nitrate reading at a private well whereas white squares do not have any nitrate readings at private wells. As seen in Figure 7, CAFOs clearly match to multiple PLSS sections and a given PLSS section may match with multiple CAFOs.

After making the spatial linkages between wells and CAFO locations, we merge our CAFO panel to create time-varying CAFO "treatment" measures. We also merge in other potentially important time-varying land use controls that could be correlated with CAFO arrivals and/or expansions. First, we use Schlenker and Roberts (2024) for precipitation data. The authors use PRISM climate data and weather monitoring stations to create daily precipitation data for 2.5- by 2.5-mile grids throughout the conterminous US. Next, we use

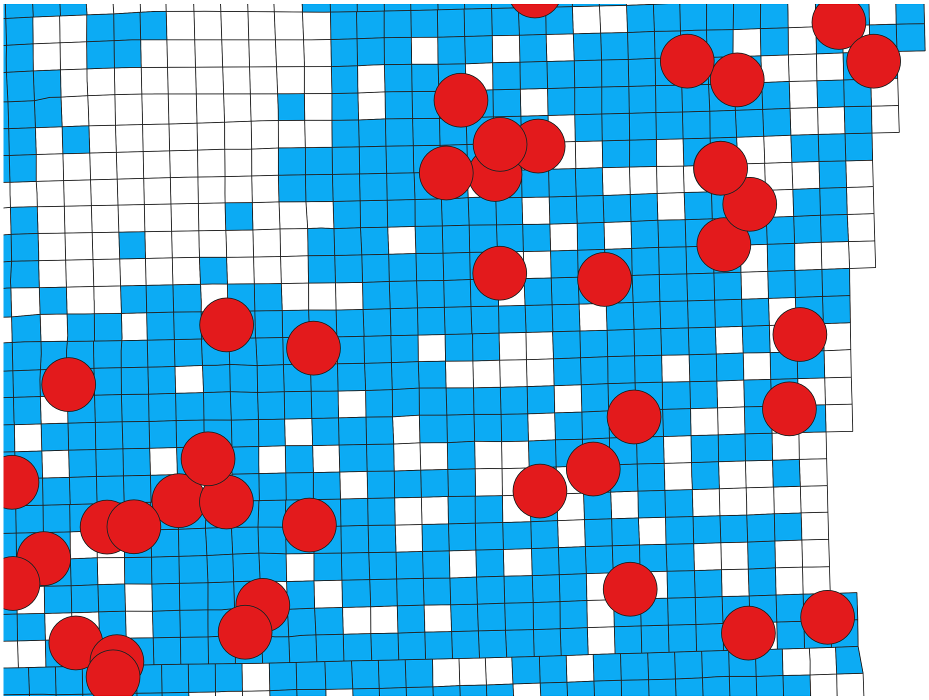

Figure 7: Nitrate private well sample with 1 mile CAFO buffers

Notes: This is a zoomed in view of a portion of Wisconsin's PLSS grid. Shaded areas represent sections with at least one nitrate observation from a private well. Red dots represent 1 mile radius buffers around permitted CAFO locations in 2023.

the USDA Census of Agriculture for our measure of non-CAFO animal count. We then gather fertilizer application data from Falcone (2020), who produces measures of on-farm and off-farm fertilizer application amounts for nitrogen and phosphorus fertilizer from the Census of Agriculture. Non-CAFO animal counts and fertilizer application data are at the county level and recorded every 5 years, so we convert these measures to the PLSS level and interpolate to get yearly measures. Finally, we leverage the National Land Cover Database (NLCD), which provides the most comprehensive and most frequently revised land cover maps for the US. The NLCD classifies land cover at 30m resolution for the years of 2001, 2003, 2006, 2008, 2011, 2013, and 2016, 2019, and 2021. For each of these years, we overlay the NLCD maps on PLSS sections and calculate the percentage of each PLSS section that is covered by each NLCD classification.[5]

Table 1 presents summary statistics for our nitrate sample. Panel A shows characteristics

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[5] We group NLCD classifications into the mutually exclusive categories of water, developed, barren, forest, shrubland, grassland, pasture/hay, cultivated crops, and wetlands.

of the private well sample and Panel B shows characteristics of the public well sample. Each observation represents a groundwater quality test result, which comes from a particular well in a given month.[6] The CAFO measures indicate how many permitted CAFOs are within 1 mile, between 1 and 3 miles, between 3 and 5 miles, or between 5 and 10 miles of the corresponding well on the month of the groundwater sample collection. As seen in Table 1, mean and median nitrate concentrations are higher in private well samples than in public well samples. In general, public wells tend to be deeper than private wells. Both private and public wells have substantial and similar exposure to CAFOs. The median well-month observation does not have a CAFO within 5 miles, but some wells are exposed to as many as 5 CAFOs within one mile.

Table 1: Summary statistics for the primary well sample

Variable	Mean	Med	Min	Max	Obs
Panel A. Private Wells					
Nitrate concentration (mg/L)	6.35	2.75	0.0015	87	10,322
Well Bottom (ft)	174.61	160	15	760	9,751
Well Casing Bottom (ft)	107.12	86	6	479	9,700
CAFOs (within 1 mile)	0.0639	0	0	5	10,322
CAFOs (1 to 3 miles)	0.237	0	0	7	10,322
CAFOs (3 to 5 miles)	0.405	0	0	9	10,322
CAFOs (5 to 10 miles)	2.108	1	0	26	10,322
Panel B. Public Wells					
Nitrate concentration (mg/L)	2.27	1.10	0.0015	9.5	21,704
Well Bottom (ft)	410.77	340	28	1865	21,527
Well Casing Bottom (ft)	180.46	130	20	950	21,472
CAFOs (within 1 mile)	0.0575	0	0	4	21,704
CAFOs (1 to 3 miles)	0.219	0	0	6	21,704
CAFOs (3 to 5 miles)	0.385	0	0	9	21,704
CAFOs (5 to 10 miles)	1.93	1	0	24	21,704

Notes: Summary statistics are at the well-month level and represent observations in the final analysis samples.

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[6] Because GRN aggregates data from several sources, it occasionally reports multiple identical nitrate test results within a span of several days. We average the nitrate results to the monthly level to avoid overweighting these results.

4 Empirical approach and results

This section lays out the empirical foundation of our study, which aims to identify the effects of CAFOs on groundwater quality. Our approach leverages the staggered timing of CAFO arrivals at varying proximity to public and private wells. Geographically linking potential groundwater contamination sources with groundwater quality sampling at a well is crucial to this exercise. There are two relevant spatial scales for our estimation: the scale that groundwater contamination occurs and the scale at which CAFOs operate.

First, as discussed in section 2, most groundwater from private wells originates from within a few miles in Wisconsin. For shallower wells, any effects would likely be concentrated within a mile of the contamination source. Contaminants from more distant sources could potentially contaminate deeper (typically public) wells. However, the deeper the well, the less likely it is that contaminants could make their way from the surface to the well. Also detailed in section 2, CAFOs could potentially affect groundwater through two main mechanisms. Barn runoff or manure storage pits could leach contaminants to groundwater directly below the CAFO. Of the two mechanisms, this has potential for being the more concentrated contamination source. Next, potential groundwater contamination from manure application on farm fields could occur within the spreading zone; this is likely within a few miles and rarely more than 5 miles away from animal barns. Because the manure is spread, this mechanism would likely have more diffuse effects.

In summary, there are several potential spatial scales of interest and it is ex ante uncertain how far away CAFOs could potentially affect wells. In general, the most concentrated effects would likely be within a mile or two of the animal barns and manure storage, and in relatively shallower wells. There is potential for diffuse effects up to the distance of manure spreading, which is typically under 5 miles.[7] It is therefore unlikely that CAFOs beyond 5 miles away would substantially impact a well's water quality.

In light of these spatial considerations, our empirical approach proceeds as follows. First, we define our estimation sample to include all wells that are within 10 miles of an operating

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[7] Some anecdotes suggest larger Wisconsin CAFOs may rarely spread manure between 5 and 10 miles away.

CAFO at any time during the sample period.[8] We then estimate a preliminary specification to identify the appropriate exposure buffer. This allows us to separate our wells into treatment and control groups. Finally, we use these treatment and control groups to estimate binary DID and event studies.

4.1 Identifying the exposure buffer

We first compare changes in nitrate concentrations for wells near CAFOs (in response to a first or additional CAFO) with changes in concentrations for wells that are farther away but still within 10 miles of CAFOs. Because the exposure buffer is ex ante uncertain, we bin CAFOs into rings around each well, creating a series of treatment variables.[9] In our baseline specification, we allow for continuous treatment intensity, where each treatment variable represents the number of CAFOs within a given buffer around the well.[10] We regress the natural log of nitrate concentrations (mg/L) at well location i in PLSS section j in month m[11] of year t, on the count of operating CAFOs within several distance bins:

$$\begin{aligned} ln(ntrt)_{ijmt} = \beta_1 CAFOs(<1mi)_{jmt} + \beta_2 CAFOs(1-3mi)_{jmt} + \beta_3 CAFOs(3-5mi)_{jmt} \\ + \beta_4 CAFOs(5-10mi)_{jmt} + \beta_5 \mathbf{X}_{jmt} + \gamma_i + \psi_m + \lambda_t + \epsilon_{ijmt}, \end{aligned} \tag{1}$$

where $CAFOs(<1mi)_{jmt}$ is the count of operating CAFOs within 1 mile of a well, $CAFOs(1-3mi)_{jmt}$ is the count of operating CAFOs between 1 and 3 miles of a well, $CAFOs(3-5mi)_{jmt}$ is the count of operating CAFOs between 3 and 5 miles of a well, and $CAFOs(5-10mi)_{jmt}$ is the count of operating CAFOs between 5 and 10 miles of a well. $\mathbf{X}_{jmt}$ are time-varying PLSS section level controls (non-CAFO animals, 30-day precipitation, commercial fertilizer, % forested, % planted, % developed), γ_i are well fixed effects, ψ_m are month fixed effects, λ_t

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[8] We show results using alternative sample cutoffs of 5 miles and 20 miles from an operating CAFO in the online appendix.

[9] This approach is similar to those used in other contexts to estimate the causal effect of a possible source of contamination on well water quality. Examples include Hill and Ma (2022) and Hill and Ma (2017), who investigate the effects of shale gas drilling on well water quality.

[10] In subsection 4.2, we investigate DID specifications with dichotomous treatment indicators.

[11] In the online appendix, we also show results where we collapse our data to the well-year level. These results are similar to those from our baseline well-month analysis.

are year fixed effects, and ϵ_{ijmt} is the exogenous error term.

Any time-invariant differences in average nitrate levels across wells will be absorbed by the well fixed effects, γ_i. Thus, factors such as local geology, soil characteristics, and well-depth will not bias our estimates. However, we are interested in treatment effect heterogeneity along some of these dimensions, and we investigate this after presenting our baseline results.

Our baseline two-way-fixed-effect (TWFE) strategy is within the general class of difference-in-differences designs with continuous treatments. The identifying assumption is that the water quality impacts at wells further away from CAFOs capture the counterfactual changes in water quality that would have occurred at wells more proximate to CAFOs in the absence of increasing proximate CAFO intensity.

Table 2 presents results for our estimation of equation 1. Panel A shows results for the private well sample and Panel B shows results for the public well sample. We estimate equation 1 alternatively using all wells (columns 1 and 2) or limiting the sample to only comparatively shallow wells (columns 3 and 4). Here, we define shallow wells as any well with a well casing bottom depth that is shallower than the 25th percentile well in its respective sample (56 feet for private wells and 90 feet for public wells). Columns 1 and 3 show results when we include only CAFO treatment variables and fixed effects (well, month, and year) whereas columns 2 and 4 show results when we add time-varying controls (non-CAFO animals, commercial nitrogen fertilizer, land cover, and precipitation).

Concentrating first on Panel A, we see that an additional CAFO locating within 1 mile of a private well significantly increases nitrate concentrations. The effect size here is rather large; one additional CAFO within a mile of a private well increases nitrate concentrations by exp(0.392)-1=50%. The marginal effect grows to 64.9% for an additional CAFO within a mile of a shallow private well. In contrast, we do not find evidence that CAFOs further than 1 mile away significantly affect nitrate concentrations. For the full sample of all private wells within 10 miles of a CAFO, coefficients on the more distant CAFO treatment variables (1-3 miles, 3-5 miles, and 5-10 miles) are generally small in magnitude and have comparatively large standard errors. For shallow wells, there is some weak evidence that moderately distant CAFOs (1-3 miles) increase nitrate concentrations, but the estimates are not significant at

Table 2: Regression results to establish treatment buffer: Effect of CAFOs within treatment rings on nitrate concentrations

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFOs within 1 mile	0.390***	0.392***	0.546**	0.500**
	(0.130)	(0.130)	(0.248)	(0.254)
CAFOs between 1 and 3 miles away	-0.0508	-0.0298	0.375	0.376
	(0.116)	(0.115)	(0.238)	(0.238)
CAFOs between 3 and 5 miles away	0.116	0.118	0.00161	0.00349
	(0.0876)	(0.0867)	(0.223)	(0.221)
CAFOs between 5 and 10 miles away	0.0304	0.0170	0.0185	0.00154
	(0.0304)	(0.0322)	(0.0826)	(0.0892)
Sample	All	All	Shallow	Shallow
Observations	10,322	10,319	1,924	1,924
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFOs within 1 mile	-0.0638	-0.0651	-0.161	-0.124
	(0.107)	(0.105)	(0.194)	(0.195)
CAFOs between 1 and 3 miles away	-0.0174	-0.0290	0.0109	0.0247
	(0.0461)	(0.0471)	(0.102)	(0.101)
CAFOs between 3 and 5 miles away	0.0827	0.0798	0.0612	0.0602
	(0.0526)	(0.0516)	(0.0890)	(0.0893)
CAFOs between 5 and 10 miles away	0.0160	0.0168	0.0204	0.0281
	(0.0156)	(0.0177)	(0.0391)	(0.0377)
Sample	All	All	Shallow	Shallow
Observations	21,704	21,704	6,403	6,403
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 1. Standard errors clustered by well are shown in parentheses. *** $p<0.01$, ** $p<0.05$, * $p<0.1$

conventional levels.[12]

Next, examining the results in Panel B of Table 2, we find that a marginal CAFO does

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[12] Results using an alternative sample of control wells (wells ever within 20 miles of an operating CAFO) are similar and shown in Panel A of Online Appendix Table A5.

not have a statistically significant effect on nitrate concentrations for the overall sample of public wells within 10 miles of a CAFO. This is true across the various treatment variables (within one mile, 1-3 miles, 3-5 miles, and 5-10 miles). Moreover, the point estimates on the CAFO treatment variables are comparatively small, do not have consistent signs, and are imprecisely estimated.[13]

4.2 DID and event studies

We next proceed to a binary DID analysis where we divide wells into treatment and control groups based on the spatial results from the previous subsection. As a baseline, we consider wells ever within one mile of a CAFO to be in the treatment group and wells between 1 and 10 miles away to be in the control group.[14] We therefore compare changes in nitrate concentrations at wells that experience their first exposure to a CAFO within 1 mile to changes in nitrate concentrations at wells that are between 1 and 10 miles from a CAFO, before and after the arrival of the CAFO. Our two-way fixed-effects (TWFE) specification is

$$ln(ntrt)_{ijmt} = \beta_1 CAFO(< 1mi)_{jmt} + \beta_2 \mathbf{X}_{jmt} + \gamma_i + \psi_m + \lambda_t + \epsilon_{ijmt}, \tag{2}$$

where $CAFO(< 1mi)_{jmt}$ is the binary treatment indicator for at least one operating CAFO within 1 mile of a well, and all other notation is equivalent to that in equation 1.

Table 3 shows results for this binary DID TWFE specification. For the full sample of private wells (panel A, columns 1 and 2), there is a large, positive average effect on nitrate concentrations of having at least one CAFO within 1 mile of a well. CAFO treatment leads to a statistically significant, 56% increase in nitrate concentrations. For the subsample of shallow private wells (panel A, columns 3 and 4), the marginal effect grows to 65 percent. In contrast, in Panel B of Table 3, we do not see any significant effect of CAFO treatment

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[13] We find similar null results using an alternative sample of control wells (wells ever within 20 miles of an operating CAFO). These results are shown in Panel B of Online Appendix Table A5.

[14] We examine the variation in treatment and outcome variables in Online Appendix A.2. There, we show substantial cross-sectional and within-well variation in treatment and outcome variables.

Table 3: DID 2WFE regression results: effect of a CAFO within 1 mile on nitrate concentrations

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFO within 1 mile	0.445***	0.446***	0.575**	0.502**
	(0.152)	(0.154)	(0.245)	(0.252)
Sample	All	All	Shallow	Shallow
Observations	10,322	10,319	1,924	1,924
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFO within 1 mile	-0.0154	-0.0274	-0.137	-0.0994
	(0.133)	(0.133)	(0.186)	(0.188)
Sample	All	All	Shallow	Shallow
Observations	21,704	21,704	6,403	6,403
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 2. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

on public well nitrate concentrations.[15] In Section 5, we further explore the implications of this divergence in results from private and public wells.

We also estimate event study specifications to test for pretrends in nitrate concentrations prior to a CAFO's arrival and to assess how treatment effects evolve after a CAFO's arrival. Specifically, we take the standard approach and modify equation 2 to replace the binary treatment indicator with a series of event time indicators. We estimate

$$ln(ntrt)_{ijmt} = \sum_{k \in -l,\ldots,0,\ldots,n\}} \delta_k * C_{ijm,t-k} + \beta \mathbf{X}_{jmt} + \gamma_i + \psi_m + \lambda_t + \epsilon_{ijmt}, \tag{3}$$

where $C_{ijm,t-k}$ are the event time indicators for at least one operating CAFO within 1 mile of a well, and all other notation is equivalent to that in equations 1 and 2. In our event study specification, we bin event time years into three year periods for the 18 years before

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[15] DID results using alternative sample selection criteria (wells ever within 20 miles of an operating CAFO or wells ever within 5 miles of an operating CAFO) are similar and shown in Online Appendix Tables A2 and A3.

and after the arrival of the first CAFO and include end-cap indicators for more than 18 years before or after the event.[16]

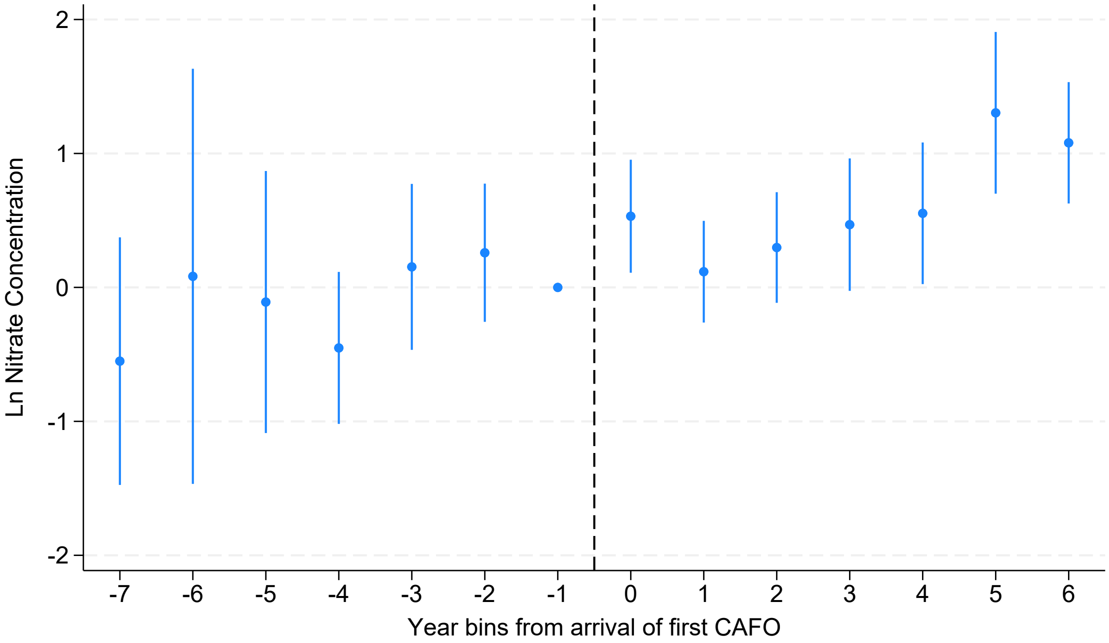

Figure 8: Private well event study

Notes: The figure shows point estimates from the estimation of the event study specification given in equation 3, along with 95% confidence intervals. Standard errors are clustered by well.

As shown in Figure 8, we do not see differential pretrends for treatment wells (wells within 1 mile of a CAFO) versus control wells (wells between 1 and 10 miles of a CAFO). Pre-treatment event time coefficients are sometimes negative, sometimes positive, and never statistically different from zero.

Once a CAFO arrives within 1 mile of a private well, we see a positive and significant effect on log nitrate concentrations within the first three years. Coefficients remain positive and fairly stable over the first fifteen years post-CAFO arrival. The event study suggests that effects may become larger beyond that point. One explanation could be that nitrogen concentrates in the soil over time and eventually reaches the aquifer in higher concentrations. However, these estimates should be interpreted cautiously because there are relatively few wells in our sample that we observe beyond fifteen years post-CAFO arrival.

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[16] We use the standard normalization of setting δ_{-1} equal to zero.

4.3 Robustness to heterogenous DID estimators

Recent research shows that standard TWFE regressions may fail to recover causal parameters when there are multiple time periods and staggered treatment timing (De Chaisemartin and d'Haultfoeuille 2020; Borusyak et al. 2024; Callaway and Sant'Anna 2021; Goodman-Bacon 2021; Sun and Abraham 2021). This literature on heterogenous DID points out potential problems when treatment effects are heterogenous across time and cohorts. In a comprehensive review of this literature, Roth et al. (2023) covers several classes of estimators that are robust to the aggregation of heterogeneous treatment effects in settings with staggered treatment timing. The approaches are often classified as: 1) the regression-based approach, 2) the imputation approach, and 3) the group-time estimator approach. The regression-based approach and the imputation approach are particularly well suited to our application, where we have an unbalanced panel of monthly observations on our dependent variable and treatment is an absorbing state. In this subsection, we show estimates of the average treatment effects on the treated (ATTs) from these heterogeneous treatment effect robust approaches.

First, in a regression-based approach, Wooldridge (2021) extends the standard TWFE estimator to incorporate interactions between treatment, cohort, and post-treatment periods. This estimator is often termed "extended two-way fixed effects (ETWFE)." In this approach, one obtains a simple ATT by aggregating event-time coefficients with regression-based weights. Second, for an imputation approach, Borusyak et al. (2024) fit a TWFE regression using not-yet-treated observations to impute counterfactual estimates for each treated unit in the absence of treatment. The method then averages estimated treatment effects across treated units and time periods to form the ATT. Both Wooldridge (2021) and Borusyak et al. (2024) avoid the problems of forbidden controls/comparisons and negative weights that can be problematic in the estimation of DID models using traditional TWFE.

We present ETWFE estimates in column 1 and Borusyak et al. (2024) imputation estimates in column 2 of Table 4. Comparing these estimates with those in columns 1 and 2 of Table 3 Panel A, we see that the results are robust to these alternative heterogeneous treatment effect DID estimators. Across TWFE, ETWFE, and the Borusyak et al. (2024) imputation estimator, the arrival of a CAFO within 1 mile of a private well increases nitrate concentrations by 50-56%.

Table 4: DID regression results: heterogeneous treatment effect estimators

Variable	(1)	(2)
CAFO within 1 mile	0.422***	0.408***
	(0.0724)	(0.0793)
Estimator	ETWFE	Borusyak et al.

Notes: Each column represents a separate ATT from the corresponding estimator, estimated on the baseline sample of private wells. ETWFE=Wooldridge (2021), Borusyak et al.=Borusyak et al. (2024). Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

4.4 Private well nitrate sampling

In the private well sample, we have an unbalanced panel of nitrate observations at the well-month level. This could create two concerns related to sample selection. We address these concerns in this subsection.

First, suppose individuals who were most concerned about CAFOs locating nearby were most likely to test and report nitrate levels to the Wisconsin DNR. Suppose further that these same CAFOs were systematically different from other CAFOs and negatively impacted groundwater quality more than the average CAFO. Then, the treated observations most likely to end up in our sample could be from the places where CAFOs were causing the most damage. This could lead to biased results, overestimating the true impact of CAFOs on nitrate concentrations.

To address this first concern, we use all wells in our final analysis sample (those with two or more nitrate observations) and create balanced panels of well-month observations and well-year observations. We then create a dichotomous dependent variable, where 1 indicates the presence of a nitrate test in our analysis sample and 0 indicates no nitrate test for that well-month/well-year in our analysis sample. Finally, we estimate our DID TWFE specification (analog of equation 2) to test for systematic differences in the probability of a nitrate sample based on CAFO treatment. We also estimate a TWFE specification using the continuous count of CAFO animal units within 1 mile as the treatment variable to test whether the probability of a nitrate sample differs by the intensity of nearby CAFO operations. As shown in Appendix Table A7 columns 1 and 3, we find no difference in the probability of a nitrate sample based on CAFO treatment. Likewise, in columns 2 and 4, we find no difference in the probability of a nitrate sample based on the number of CAFO animal units within 1 mile of a well.

A second potential concern in the private well sample relates to more nitrate observations from some wells than from others. Wells with more observations implicitly receive more weight in the regressions. If these wells with more nitrate observations were systematically different from wells with fewer nitrate observations, we could either overestimate or underestimate the true impact of CAFOs on nitrate concentrations.

To address this second concern, we retain only the first and last observations for each well in our final analysis sample. We then re-estimate equation 2 on this subsample of observations. As seen in Appendix Table A8, results on this subsample of two observations per well are consistent with our main results in Table 3. Here, we find that the location of a CAFO within one mile of a private well increases nitrate concentrations by appoximately 68%.

4.5 Placebo tests

The most likely threat to our identification of causal effects is other non-CAFO agriculture. Most notably, Wisconsin CAFOs are typically located in areas that have intensive row crop agriculture, which also uses commercial fertilizers. If changes in commercial fertilizer correlate with changes in proximate CAFO treatment, we could misattribute the effects of commercial fertilizer to CAFOs. The restriction of our control group to wells within 10 miles of CAFOs and our inclusion of time varying land use and fertilizer controls in our regression specifications partially assuage this potential confounder, but concerns could remain. In this section, we conduct placebo tests to further increase the credibility of our research design. In the typology of Eggers et al. (2024), we conduct a placebo outcome test on a groundwater contaminant that is a function of row crop agriculture but not affected by CAFOs, and a placebo treatment test using CAFOs beyond the plausible manure spreading range.

Wells with high levels of nitrates often also test positive for pesticides, commonly attributed to pesticides spread onto row crops. CAFO manure should not contain pesticides and therefore should not affect pesticide readings in groundwater. We create a dichotomous outcome variable to indicate whether a well sample tests positive for pesticides and then re-estimate equations 1 and 2 using this dichotomous placebo outcome.

Table 5 presents the results for these pesticide placebo outcome regressions. In columns 1 and 2, we find no effect of CAFOs on the probability of a private well testing positive for pesticides, across all distances within 10 miles. The coefficients are very small in magnitude and never statistically different from 0 at conventional levels. Likewise, in the DID estimates of columns 3 and 4, we find no effect of a nearby CAFO (within 1 mile) on the probability of a private well testing positive for pesticides. These null placebo outcome results add further credibility to the claim that our research design identifies the effect of CAFOs on well water quality.

Table 5: Private wells placebo outcome regression results: pesticides

Variable	(1)	(2)	(3)	(4)
CAFOs within 1 mile	0.00242	0.00270		
	(0.00311)	(0.00318)		
CAFOs between 1 and 3 miles away	0.0000316	0.000103		
	(0.00289)	(0.00264)		
CAFOs between 3 and 5 miles away	-0.00320	-0.00344		
	(0.00281)	(0.00285)		
CAFOs between 5 and 10 miles away	0.000139	-0.0000441		
	(0.00164)	(0.00144)		
CAFO within 1 mile (DID)			0.000886	0.00166
			(0.00341)	(0.00356)
Observations	39,922	39,922	39,922	39,922
Time Varying Controls	No	Yes	No	Yes

Notes: Columns 1 and 2 represent separate estimations of equation 1, using pesticide detection as the dichotomous dependent variable. Columns 3 and 4 represent separate estimations of equation 2, using pesticide detection as the dichotomous dependent variable. Standard errors clustered by well are shown in parentheses. *** $p<0.01$, ** $p<0.05$, * $p<0.1$

Next, for a placebo treatment test, we estimate a modified version of equation 1, using the count of operating CAFOs between 10 and 20 miles from a well:

$$ln(ntrt)_{ijmt} = \beta_1 CAFOs(10-20mi)_{jmt} + \beta_2 \mathbf{X}_{jmt} + \gamma_i + \psi_m + \lambda_t + \epsilon_{ijmt}. \tag{4}$$

The motivation for this test is that we might not perfectly measure changes in commercial fertilizer/row crop agriculture near the well, and these changes in commercial fertilizer/row crop agriculture may be correlated with changes in proximate CAFOs. The distance of 10-20 miles away from a well is farther than the spread of manure, and hence these more distant CAFOs should not affect nitrate levels. However, fertilizer/row crop agriculture 10-20 miles away is likely similar to fertilizer/row crop agriculture near the well. Thus, a positive and significant effect of CAFOs 10-20 miles away could signal problems with our identification strategy.

Table 6 shows results for private wells. We do not find any evidence that CAFOs 10-20 miles away affect nitrate concentrations. Point estimates on the coefficients across the four columns of Table 6 are small and not statistically different from 0. These null effects further enhance the credibility of our research design.

Table 6: Private wells placebo treatment regression results: CAFOs 10-20 miles away

Variable	(1)	(2)	(3)	(4)
CAFOs between 10 and 20 miles away	0.00808	0.00554	0.0274	0.0194
	(0.0157)	(0.0182)	(0.0458)	(0.0522)
Sample	All	All	Shallow	Shallow
Observations	10,322	10,319	1,924	1,924
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 4. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

4.6 Karst and shallow carbonate landscapes

Carbonate bedrock underlies a substantial portion of Wisconsin. Carbonate rocks, such as dolomite and limestone, are soluble and commonly fractured. Because these rocks are soluble, seeping water often widens fractures, creating underground conduits and caves.

Additionally, sinkholes may form at the surface. These features are typical of a “karst” landscape (Bradbury 2009).

Weary and Doctor (2014) describe USGS digital maps that delineate areas of karst or the potential for development of karst. These maps categorize formations into 18 karst types for the conterminous US. Of these, three types are found in Wisconsin: carbonate rocks at or near the land surface, carbonate rocks buried beneath ≤ 50 ft of glacially derived insoluble sediments, and carbonate rocks buried beneath ≥ 50 ft of glacially derived insoluble sediments. Figure 9 shows the geographic distribution of carbonate bedrock at these three exposure levels. We overlay these USGS karst maps onto our maps of well locations to identify all wells located in sections with shallow bedrock.

Aquifers in areas with shallow carbonate bedrock are particularly vulnerable to contamination. First, carbonate rocks provide poor filtration. Thus, contaminants that enter the ground can penetrate all the way to the aquifer. Second, groundwater flows can be much more rapid in karst systems, up to hundreds of feet per day. Therefore, we create two indicator variables, one for wells in PLSS sections with carbonate bedrock at or near the surface and one for carbonate bedrock at the surface or less than 50 feet from the surface.

Table 7 presents results for specifications that interact the DID indicator in equation 2 with indicators for shallow carbonate well locations. In columns 1 and 2 of Table 7, we define only wells on carbonate bedrock at or near the surface to be shallow carbonate bedrock. In columns 3 and 4, we add wells located on carbonate bedrock not at the surface but within 50 feet of the surface to our definition of shallow bedrock. Once again, Panel A shows results for private wells whereas Panel B shows results for public wells.

As seen in columns 1 and 2 of Table 7, there is a large and positive estimated differential treatment effect for wells located on carbonate bedrock at or near the surface. Interestingly, the estimated magnitude of the differential effect is quite similar for public and private wells, although the differential effect is not statistically significant at conventional levels for public wells. Concentrating on private wells, the average treatment effect of a CAFO within one mile increases from 54% for non-shallow carbonate bedrock wells to 165% for shallow carbonate bedrock wells. The differential effects in columns 3 and 4 of Table 7 are not as precisely estimated, and are of comparatively smaller magnitudes. Together, these results

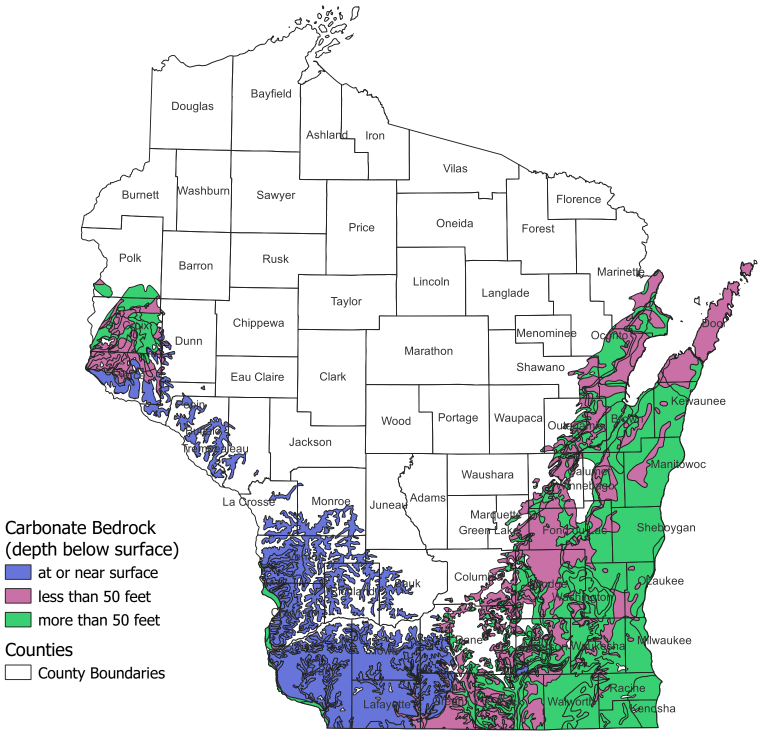

Figure 9: Karst in Wisconsin

Notes: This figure shows the location of shallow carbonate bedrock in Wisconsin, according to GIS data from USGS.

suggest that groundwater quality in wells located on shallow carbonate bedrock near the surface may be especially susceptible to contamination from CAFOs.

4.7 Discussion of results

In summary, the private well results imply that an average private well is mostly impacted by very proximate CAFOs, a shallow private well is more heavily impacted than a deeper well, and a well located in an area with shallow carbonate bedrock is substantially more impacted than wells in other areas. Taken together, this informs on the potential mechanisms of contamination. Activities at the CAFO itself appear to drive the strongest effects; possibilities include manure storage, barn runoff, or very concentrated manure spreading near the CAFO. However, since CAFOs several miles away may also affect shallow wells, manure spreading on nearby agricultural lands remains a plausible explanation for this contamination.

These marginal effects imply policy relevant increases in average nitrate concentrations at

Table 7: Shallow carbonate bedrock heterogeneity: effect of a CAFO within 1 mile on nitrate concentrations

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFO within 1 mile	0.433***	0.434***	0.390**	0.394**
	(0.155)	(0.156)	(0.176)	(0.177)
CAFO within 1 mile × *I(SCB)*	0.593***	0.541**	0.295	0.277
	(0.224)	(0.232)	(0.295)	(0.303)
SCB	Surface	Surface	≤ 50 ft	≤ 50 ft
Observations	10,322	10,319	10,322	10,319
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFO within 1 mile	-0.0618	-0.0759	0.0599	0.0462
	(0.138)	(0.138)	(0.149)	(0.149)
CAFO within 1 mile × *I(SCB)*	0.503	0.526	-0.253	-0.247
	(0.438)	(0.437)	(0.295)	(0.293)
Carbonate bedrock	Surface	Surface	≤ 50 ft	≤ 50 ft
Observations	21,704	21,704	21,704	21,704
Time Varying Controls	No	Yes	No	Yes

Notes: SCB=shallow carbonate bedrock. Each column represents a separate estimation of equation 2, with the DID CAFO indicator interacted with SCB. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

private wells. At the sample mean of 6.35 mg/L for all private wells, the results from Table 3 imply that the arrival of a CAFO within 1 mile would increase the nitrate concentration by approximately 3.56 mg/L. Thus, an additional CAFO within 1 mile of a private well could plausibly push many wells from under the WDNR advisory level for nitrates of 10 mg/L to above the advisory level. Additionally, the nitrate sample mean for the subsample of shallow private wells (6.42 mg/L) is similar to the nitrate sample mean for the overall private well. Since the estimated marginal effect for shallow private wells is higher than the estimated marginal effect for the overall private well sample, the potential health effects could be even more pronounced in shallow private wells.

Our findings highlighting differential effects of CAFOs on private versus public groundwater quality agree with previous research concerning other environmental hazards. For example, Muehlenbachs et al. (2015) find that nearby shale gas construction substantially harms property values for homes on private wells but impose much smaller costs property values for homes on community water supplies. In their analysis, property owners on community water supplies may even receive net benefits from shale gas construction, after considering royalty payments.

5 Environmental justice of private vs public wells

Motivated by our econometric results showing differential effects for private versus public wells and shallow versus average-depth wells, we next explore the implications for different groups. We first investigate systematic differences in nitrate concentrations between private and public wells that correlate with well depth. We then examine how well casing depth correlates with different demographic characteristics.

As described in section 2.3, deeper wells tend to be more protected from contamination. We quantify this relationship in our private and public well samples by regressing the natural log of nitrate concentration on well casing depth.[17] As seen in column 1 of Panel A in Table 8, each additional foot of private well casing depth corresponds with a 0.75% reduction in nitrate concentration. This relationship is similar in public wells; Panel B shows each additional foot of public well casing depth corresponds with a 0.8% reduction in nitrate concentrations. Recall that the mean public well has a casing bottom that is 73 feet deeper than the mean private well. This "intercept shift" in public wells may be enough to mitigate nitrate contamination for the vast majority of public wells. Moreover, well casing depth can differ by hundreds of feet, so this relationship with average nitrate concentration can be meaningful.

We next link demographic data at the census tract level to each PLSS section. Figure 10 shows a map of Wisconsin PLSS sections with layered census tracts. This is straightforward for private wells, where we match census tract characteristics to each PLSS section, weighting by percentage of the section covered by a census tract. For public wells, individuals

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[17] We use the same controls as in equation 1: non-CAFO animals, 30-day precipitation, commercial fertilizer, % forested, % planted, % developed), month fixed effects, and year fixed effects.

Table 8: Descriptive differences in wells related to depth and census characteristics

Dependent Variable	(1) ln(nitrate)	(2) ln(well depth)
Panel A. Private Wells		
Well depth (ft)	-0.00754***	
	(0.00127)	
Percentage rural		-0.000912*
		(0.000497)
Median income ($10k)		0.0427**
		(0.0174)
Percentage with no HS diploma		-0.0139***
		(0.00583)
Percentage minority		0.00580**
		(0.00257)
Observations	9,700	9,700
Panel B. Public Wells		
Well depth (ft)	-0.00800***	
	(0.00105)	
Percentage rural		-0.00152
		(0.00126)
Median income ($10k)		0.0380
		(0.0258)
Percentage with no HS diploma		-0.0468***
		(0.0108)
Percentage minority		0.00407
		(0.00709)
Observations	21,704	21,704

Notes: Each column represents a separate OLS regression. Standard errors clustered by PLSS section are shown in parentheses. *** $p<0.01$, ** $p<0.05$, * $p<0.1$

consuming water from a well are typically spread over a larger geographic area. We use the EPA Community Water System Area Boundaries geospatial data to link census tracts to service areas of public wells.[18]

There are several demographic characteristics that could correlate with well casing depth. First, impervious surfaces prevent water from soaking into the ground, which could lower the water table. Therefore, it is possible that urban areas may need deeper wells to reach an aquifer, whereas rural areas could be more likely to reach an aquifer with a shallower well. Aside from the depth of the water table, a deeper well is ultimately a form of drinking water pollution mitigation. It is generally more expensive to drill deeper wells, so areas with higher income may be more likely to have deeper wells. The environmental justice literature suggests two other categories that could be important in explaining levels of environmental pollution mitigation: minority status and low educational attainment. Following the CDC Environmental Justice Index (EJI), we measure minority status with the percent of the census tract population that is a racial/ethnic minority (all persons except white, non-Hispanic) and we measure educational attainment with the percent of the census tract population (age 25+) with no high school diploma.

Table 9 presents summary statistics for linked census characteristics in the private and public well samples. In general, mean sample demographics are similar between the two samples. Compared to public wells, private wells are located in areas that are more rural, have a slightly smaller percentage of racial/ethnic minorities, have a slightly smaller percentage of individuals without a high school diploma, and have slightly higher income.

We regress the natural log of well casing depth on these census tract demographics to test for systematic differences in well depth. Column 2 of Table 8 shows these results. In Panel A, for private wells, each linked census tract demographic characteristic is statistically related to well depth. Census tracts with a higher percentage of rural population, lower median income, a higher percentage of individuals without a HS diploma, and a higher percentage of white individuals have shallower wells.

In Panel B, for public wells, we see the same directional associations, but only percentage without a HS diploma is significant at conventional levels. One interpretation is that resources and demographics still play a limited role in the depth of public wells, but the

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[18] Information on EPA's methodology for the community water system service area boundaries is available at https://www.epa.gov/ground-water-and-drinking-water/community-water-system-service-area-boundaries.

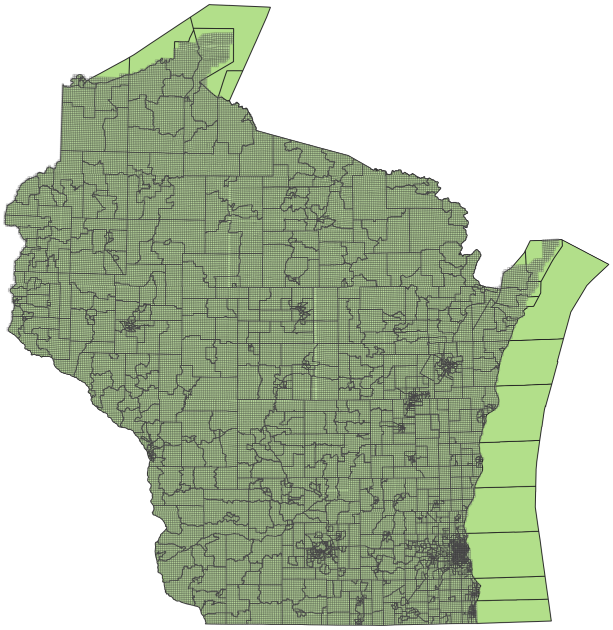

Figure 10: Wisconsin's PLSS sections with layered census tracts

Table 9: Summary statistics for linked census characteristics

Variable	Mean	Med	SD	Obs
Panel A. Private Wells				
Percentage rural	73.70	91.10	31.69	10,322
Median income ($10k)	8.15	7.92	2.09	10,322
Percentage with no HS diploma	6.43	5.7	3.82	10,322
Percentage minority	7.59	6.20	5.65	10,322
Panel B. Public Wells				
Percentage rural	59.17	62.53	40.07	21,704
Median income ($10k)	6.93	6.71	1.49	21,704
Percentage with no HS diploma	7.85	7.40	3.75	21,704
Percentage minority	8.80	6.40	7.23	21,704

Notes: Summary statistics are at the well-month level and represent observations in the final analysis samples.

relationship is muted because drinking water quality in public wells is regulated. Municipalities must drill wells deep enough to supply an ample quantity of water that meets drinking water standards.

6 Conclusion

In this study, we investigate the impact of nearby CAFOs on groundwater quality. Leveraging spatial and temporal variation in the proximity of CAFOs to wells in Wisconsin, we show that CAFOs increase nitrate concentrations in nearby private wells. These effects are especially concentrated at private wells within 1 mile of a CAFO. In contrast, we find null effects of CAFOs on nitrate levels in the average public well, even when public wells are within a mile of CAFOs.

We find the strongest CAFO impacts in shallow private wells and in private wells located in areas with shallow carbonate bedrock. In addition, we find that shallower wells tend to have higher baseline levels of nitrate. Among private wells, shallower wells tend to be in areas that are more rural, have lower median income, have lower levels of education, and have a whiter population. These results underscore the importance of considering environmental justice concerns in the regulation and monitoring of intensive livestock operations, as CAFOs disproportionately affect individuals residing in rural areas with private wells.

Finally, this study motivates future work in several areas. First, because nitrates are known to adversely affect maternal and fetal health, a logical extension is to test for impacts of CAFO exposure on local incidence of birth defects and miscarriages. Relatedly, nitrate exposure is linked to multiple cancers, so another line of inquiry could investigate whether proximate CAFO exposure affects cancer rates. This would require longer term health data that could be matched to CAFO exposure with a temporal lag. From a policy perspective, we could use more work evaluating existing policies and regulations aimed at mitigating groundwater pollution from CAFOs. A policy analysis leveraging variation in multiple states' regulatory frameworks and enforcement mechanisms could help inform future mitigation strategies.

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A Appendices (for online publication only)

A.1 Alternative Sample Criteria

Table A1: Regression results to establish treatment buffer: Effect of CAFOs within treatment rings on nitrate concentrations. Sample includes wells ever within 20 miles of an operating CAFO.

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFOs within 1 mile	0.410***	0.407***	0.530**	0.483**
	(0.129)	(0.128)	(0.228)	(0.225)
CAFOs between 1 and 3 miles away	-0.0365	-0.0317	0.353	0.361
	(0.116)	(0.114)	(0.232)	(0.235)
CAFOs between 3 and 5 miles away	0.120	0.122	0.0267	0.0417
	(0.0875)	(0.0869)	(0.223)	(0.223)
CAFOs between 5 and 10 miles away	0.0460	0.0327	0.00115	-0.00462
	(0.0287)	(0.0303)	(0.0783)	(0.0820)
Sample	All	All	Shallow	Shallow
Observations	14,956	14,953	3,253	3,253
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFOs within 1 mile	-0.0475	-0.0544	-0.134	-0.107
	(0.108)	(0.106)	(0.190)	(0.189)
CAFOs between 1 and 3 miles away	-0.0163	-0.0213	0.00577	0.0101
	(0.0465)	(0.0471)	(0.102)	(0.0997)
CAFOs between 3 and 5 miles away	0.0806	0.0819	0.0577	0.0455
	(0.0535)	(0.0523)	(0.0888)	(0.0901)
CAFOs between 5 and 10 miles away	0.0221	0.0188	0.0238	0.0269
	(0.0150)	(0.0166)	(0.0388)	(0.0369)
Sample	All	All	Shallow	Shallow
Observations	28,174	28,174	8,645	8,645
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 1. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

Table A2: DID 2WFE regression results: effect of a CAFO within 1 mile on nitrate concentrations. Sample includes wells ever within 20 miles of an operating CAFO.

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFO within 1 mile	0.517***	0.504***	0.566***	0.508**
	(0.149)	(0.151)	(0.207)	(0.210)
Sample	All	All	Shallow	Shallow
Observations	14,956	14,953	3,253	3,253
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFO within 1 mile	0.0246	0.00229	-0.102	-0.0820
	(0.132)	(0.133)	(0.182)	(0.184)
Sample	All	All	Shallow	Shallow
Observations	28,174	28,174	8,645	8,645
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 2. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

Table A3: DID 2WFE regression results: effect of a CAFO within 1 mile on nitrate concentrations. Sample includes wells ever within 5 miles of an operating CAFO.

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFO within 1 mile	0.472***	0.458***	0.559**	0.322
	(0.163)	(0.168)	(0.283)	(0.338)
Sample	All	All	Shallow	Shallow
Observations	5,263	5,263	873	873
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFO within 1 mile	-0.0655	-0.0803	-0.147	-0.0587
	(0.135)	(0.136)	(0.193)	(0.207)
Sample	All	All	Shallow	Shallow
Observations	11,510	11,510	3,227	3,227
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 2. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

A.2 Variation in treatment and outcome for DID estimation

In this online appendix, we describe the variation in our treatment and outcome variables for the DID estimation. Appendix Table A4 provides statistical summaries for the final DID analysis samples. Panel A shows summaries for the private well sample and Panel B shows summaries for the public well sample. Approximately 6 % of private well observations and 5% of public well observations receive treatment (CAFO within one mile). Therefore, the cross-sectional coefficients of variation for our treatment measure are 4.09 (private wells) and 4.42 (public wells).

Our DID identification relies on changes within wells. Thus, we next examine the within-well variation in our outcome and treatment variable. We calculate the standard deviation of each well's (ln) nitrate concentrations over the sample period and generate summary statistics for this measure. We then do the analogous calculations for each well's treatment status over the sample period.

Beginning with the private well sample, the well-specific standard deviation of (ln) nitrate concentrations has a mean of 1.12 and a standard deviation of 1.60. The well-specific standard deviation of CAFO treatment has a mean of 0.0121 and a standard deviation of 0.089. Thus, the private well sample ample within-well variation in both outcome and treatment variables.

For the public well sample, well-specific standard deviation of (ln) nitrate concentrations has a mean of 1.09 and a standard deviation of 0.88. The well-specific standard deviation of CAFO treatment has a mean of 0.0283 and a standard deviation of 0.11. As with the private well sample, we see that the public well sample has substantial within-well variation in both outcome and treatment variables.

Table A4: Summary statistics for DID estimation

Variable	Mean	SD	Min	Max
Panel A. Private Wells				
Nitrate concentration (mg/L)	6.35	12.22	0.0015	87
CAFO within 1 mile	0.0562	0.230	0	1
Observations		10,322		
Panel B. Public Wells				
Nitrate concentration (mg/L)	2.27	2.66	0.0015	9.5
CAFO within 1 mile	0.0487	0.215	0	1
Observations		21,704		

Notes: Summary statistics are at the well-month level and represent observations in the final DID samples.

A.3 Analysis at the annual level

Table A5: Regression results to establish treatment buffer: Effect of CAFOs within treatment rings on nitrate concentrations. Observations collapsed to the annual level.

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFOs within 1 mile	0.404***	0.392***	0.605***	0.579**
	(0.131)	(0.131)	(0.228)	(0.234)
CAFOs between 1 and 3 miles away	-0.0437	-0.0172	0.379	0.385
	(0.118)	(0.116)	(0.234)	(0.234)
CAFOs between 3 and 5 miles away	0.101	0.106	0.0183	0.0267
	(0.0886)	(0.0875)	(0.221)	(0.219)
CAFOs between 5 and 10 miles away	0.0374	0.0265	0.0239	0.0110
	(0.0306)	(0.0322)	(0.0821)	(0.0873)
Sample	All	All	Shallow	Shallow
Observations	8,670	8,670	1,548	1,548
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFOs within 1 mile	-0.0576	-0.0585	-0.169	-0.140
	(0.109)	(0.108)	(0.209)	(0.210)
CAFOs between 1 and 3 miles away	-0.0158	-0.0274	0.00589	0.0141
	(0.0459)	(0.0471)	(0.104)	(0.104)
CAFOs between 3 and 5 miles away	0.0813	0.0794	0.0570	0.0610
	(0.0551)	(0.0542)	(0.0888)	(0.0891)
CAFOs between 5 and 10 miles away	0.0131	0.0122	0.0242	0.0284
	(0.0167)	(0.0189)	(0.0400)	(0.0385)
Sample	All	All	Shallow	Shallow
Observations	16,667	16,667	4,294	4,294
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 1. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

Table A6: DID 2WFE regression results: effect of a CAFO within 1 mile on nitrate concentrations. Observations collapsed to the annual level.

Variable	(1)	(2)	(3)	(4)
Panel A. Private Wells				
CAFO within 1 mile	0.386**	0.393**	0.605***	0.579**
	(0.153)	(0.154)	(0.228)	(0.234)
Sample	All	All	Shallow	Shallow
Observations	8,670	8,670	1,548	1,548
Time Varying Controls	No	Yes	No	Yes
Panel B. Public Wells				
CAFO within 1 mile	-0.0108	-0.0212	-0.145	-0.117
	(0.136)	(0.136)	(0.201)	(0.204)
Sample	All	All	Shallow	Shallow
Observations	16,667	16,667	4,294	4,294
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 2. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1

A.4 Results for private well nitrate sampling

Table A7: Private wells nitrate sampling probability

Variable	(1)	(2)	(3)	(4)
CAFO within 1 mile (DID)	-0.000276		-0.00220	
	(0.000734)		(0.00892)	
CAFO AU within 1 mile (1000s)		-0.00011		-0.00661
		(0.000128)		(0.00154)
Level of analysis	Well-month	Well-month	Well-year	Well-year

Notes: We form balanced panels of well-month observations (columns 1 and 2) and well-year observations (columns 3 and 4) for these regressions. The dependent variable is a dichotomous indicator for the presence of a nitrate test in our main analysis sample for a given well and month/year. We estimate separate specifications analogous to equation 2. AU=animal units. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1 *** p<0.01, ** p<0.05, * p<0.1

Table A8: DID 2WFE regression results: retaining only first and last observations for each private well

Variable	(1)	(2)	(3)	(4)
CAFO within 1 mile	0.517***	0.514***	0.667*	0.639*
	(0.183)	(0.185)	(0.349)	(0.335)
Sample	All	All	Shallow	Shallow
Time Varying Controls	No	Yes	No	Yes

Notes: Each column represents a separate estimation of equation 2. Standard errors clustered by well are shown in parentheses. *** p<0.01, ** p<0.05, * p<0.1