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Abstract

Fertilizers and manure applied to cropland to increase yields are often lost via surface erosion, soil leaching, and runoff, increasing nutrient loads in surface and sub-surface waters, degrading water quality, and worsening the ‘dead zone’ in the Gulf of Mexico. We leverage spatial and temporal variation in agricultural practices and precipitation events to examine how these factors affect stream total phosphorus (TP) concentrations and loads in the Sugar River (Wisconsin), recently listed as impaired. To perform our analysis, we first collected water quality data from 1995 to 2017 from 40 sites along the Sugar River and its tributaries. Starting in 2004, three dairy farms expanded to become concentrated animal feeding operations (CAFOs) in this watershed. We then estimated how time of year, stream position, discharge volume, and proximity to the newly expanded CAFOs affected TP concentrations and loads. Total P concentrations, which ranged from 0.02 to 1.4 mg/L and often exceeded the EPA surface water standard of 0.1 mg/L, increased with increases in stream discharge and proximity to dairy operations, peaking in early spring to mid-summer coincident with extreme precipitation events. Our empirical analysis also shows that TP concentrations downstream from the newly permitted CAFOs increased by 19% relative to upstream concentrations. When examining total daily phosphorus loads (concentration × discharge) from this 780 km^2 watershed, we found that loads ranged from 5.88 to 4801 kg. Compared to upstream TP loads, those downstream from the CAFOs increased by 91% after the expansions – over four times that of concentration increases – implying that the rate of downstream phosphorus transfer has increased due to CAFO expansion. Our results argue for standards that focus on loads rather than concentrations and monitoring that includes peak events. As agriculture intensifies and extreme rainfall events become more frequent, it becomes increasingly important to limit soil and TP runoff from manure and fertilizer. Siting CAFOs carefully, limiting their size, and improving farming practices in proximity to CAFOs in spring and early summer could considerably reduce nutrient loads.

Keywords

Nutrient loads, Total phosphorus, CAFO, Surface water impairment, Watershed, Water quality

1. Introduction

Nutrients applied to increase agricultural production become pollutants when they run off cropland, are transferred into groundwater, and delivered to streams. Rivers in many agricultural areas of the Midwest are experiencing increases in phosphorus (P) and nitrogen (N) nutrient loads, impairing surface and groundwater quality, and eutrophying local streams and lakes (Cooke et al., 1993; WI DNR 2016, 2017). These nutrient loads generate harmful cyanobacterial blooms that deplete oxygen levels, kill fish, impair the growth of aquatic vegetation, and reduce biodiversity (Carpenter et al., 1998; McDowell et al., 2004). Since the Clean Water Act – which strictly regulates point sources of pollution – was passed, non-point sources have expanded in both relative and absolute terms to the point that they now represent the largest source of nutrient runoff, impairing water quality in many watersheds (Olmstead 2009). The intensification of agricultural practices, particularly in the form of large-scale concentrated animal feeding operations (CAFOs), has accelerated these trends. The number of CAFOs in the U.S. has increased almost 10% just since 2012, contributing to total manure inputs to agriculture of over 4,000,000 MT as N and 1,400,000 MT as P (Gilbert 2020). The manure from dairy operations held in open lagoons also increases aerial emissions of ammonia (>4,500,000 MT in 2019 from animal waste) and greenhouse gases like methane, where emissions from dairy increased 134% between 1990 and 2017 (Gilbert 2020). The U.S. Environmental Protection Agency (EPA) (2020) estimates that in 2007, farms in Wisconsin generated 191,761 MT of manure containing 42,098 MT of P, or 684 kg/km^2.

Under the Clean Water Act, CAFOs are considered point sources of pollution and are strongly regulated in most U.S. states. In contrast, less intensive forms of agriculture are considered non-point sources subject to far less regulation. This division is artificial, however, in that CAFOs are directly linked to nearby agricultural lands by their need to spread manure locally. CAFO permits do not directly regulate the amounts of CAFO waste spread onto fields, which tend to cause diffuse P losses (see Note 1, Supplementary Information). Because manure spreading typically occurs within an economic hauling radius (<3–10 miles under most Wisconsin permits), CAFOs have an incentive to overapply manure to save on transport and distribution costs. Non-point source pollution is then likely to occur when manure is used as a fertilizer as it oversupplies P relative to N, generating no agronomic benefit from the excess P when manure is applied to raise N (Wortmann et al., 2005). Both Sharpley et al. (1994) and Kleinman et al. (2011) consider the over-application of P relative to crop requirements in areas with intensive crop and livestock production to be the most important, and recalcitrant, cause of increasing soil P levels and consequent P losses. Nutrient inputs, especially P, have led to many harmful algal blooms, including those in Lake Erie where an estimated 88%–93% of P loading has been attributed to nonpoint sources, particularly agriculture (Wilson et al., 2019).

Agricultural pollution remains a difficult policy issue for several reasons. Modes of agricultural production continue to intensify with farmers now routinely applying high loads of fertilizer to increase yields. Livestock operations continue to grow in size and involve more feedlot production (Hufane 2015). Smaller farms with pasture-grazed dairy cows continue to decline or transition to larger operations, including CAFOs. Manure, long considered an asset on small family farms, becomes harder to handle and potentially a disposal problem for CAFOs, which must capture the manure in large pits or tanks to store until it can be can transported and applied as fertilizer to cropland. Large manure lagoons must be emptied or managed regularly, creating incentives to spread manure at times and at levels that may not be optimal for crop production (e.g., on frozen fields in winter, contributing to spring runoff). Using large tractors to directly inject liquid manure into farmlands may overload soils locally. Leakages also occur, both in fields and on roads from trucks distributing manure, increasing nutrient losses. This spreading of manure and other fertilizers poses particular risks in areas with sandy, highly permeable soils or karst geology where surface water quickly percolates down to affect groundwater. Such areas have seen significant increases in the contamination of wells used for drinking water (Berquist 2018; Dukehart 2017; Wang et al., 2017). In reviewing CAFO impacts on water quality, Burkholder et al. (2007) concluded that accepted livestock manure management practices fail to adequately protect waters from excessive nutrients, microbial pathogens, and added pharmaceuticals. Heavier applications of manure and fertilizers, particularly at times when soils cannot readily absorb them, combined with increases in the frequency and intensity of storm events now threaten the quality of both surface and groundwater even in areas with deeper soils (Gerbs and Smith, 2004; Hufane 2015; Wang et al., 2017).

Multiple mitigation options exist to improve water quality by reducing nutrient and sediment loads from point and non-point sources (Schoumans et al., 2014). Many watershed management programs have begun to implement agricultural Best Management Practices (BMPs - Bishop et al., 2005; Smith and Porter 2010). Some regions and federal Farm Bill programs now require Nutrient Management Plans for particular types of farming (e.g., grain, dairy). However, there is a need to better understand the effectiveness of these programs and thus their potential to reduce the ecological impacts of agricultural practices. Countering these advances are statewide "right to farm" laws dating to the 1980s which increasingly include provisions designed to protect industrial farms from private and public nuisance claims associated with environmental degradation (Schultz and Jacobs 2017).

Specific water quality regulations exist in Wisconsin to limit P and other nutrient losses from non-point source agricultural operations, particularly regarding discharges from CAFOs associated with manure treatment. The regulations rely on modeling the assimilative capacity of the soils in farm fields where manure spreading occurs. However, absent extensive careful monitoring and consistent enforcement, enforcement actions have usually been triggered by citizen science findings or conspicuous fish kills. Regular interval sampling is required under Wisconsin's Quality Standards for Surface Water. These standards stipulate that streams in the Sugar River Watershed should have TP concentrations of 75 µg/L (0.075 mg/L) or less based on the median of 6+ samples taken monthly from May–October. Streams that exceed 0.075 mg/L are considered impaired by the Wisconsin Department of Natural Resources (WI DNR).

In this study, we evaluated the effects of agricultural practices on surface water quality in the Sugar River watershed, located in south-central Wisconsin (Fig. 1). We specifically explored how changes in stream flow and the scale of dairy operations have affected stream P concentrations and loads. We focus on P as it often limits productivity (and thus biotic impacts) in freshwater environments (Correll 1999). The Sugar River has ecological significance as recognized by its designation as an "Exceptional Resource Water" warranting special protection (WI DNR 2016). Despite this designation, the Sugar River is now listed among Wisconsin's impaired waters as total phosphorus (TP) levels regularly exceed EPA levels considered safe for fish and aquatic life (0.1 mg/L, WI DNR 2017). Dodds and Welch (2000) and McDowell et al. (2004) recommend using TP for assessing the nutrient status of lakes and rivers given that appreciable particulate P may be present.

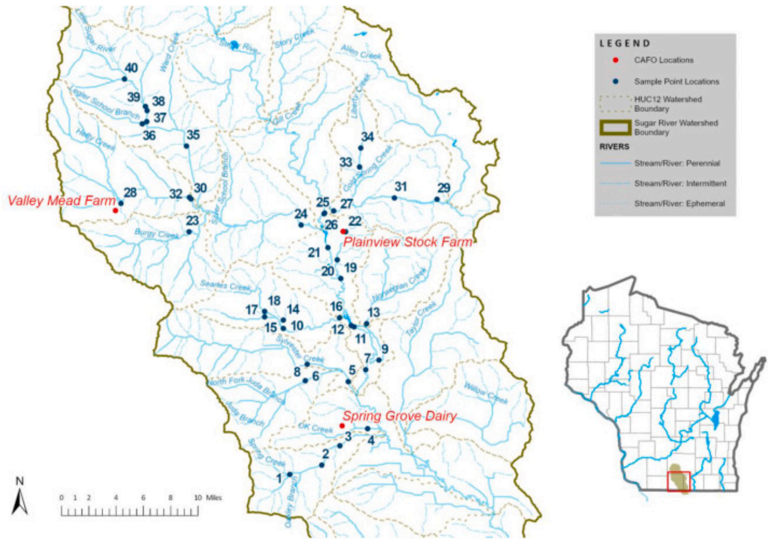

Fig. 1. Location of the Sugar and Little Sugar River watersheds in relation to the surrounding region in south central Wisconsin, USA. Also shown are the three CAFOs and the locations where water quality was monitored.

We first document long-term trends in discharge for the Sugar River (at Brodhead Station). These trends in discharge reflect increases in precipitation, boosting surface water nutrient loads both from increases in volume and from the positive correlation reported between nutrient concentrations and precipitation (Raff and Meyer 2019). We then analyzed how TP concentrations and loads in the river and its tributaries between 1995 and 2017 covary with stream discharge and distance to dairy operations over this period. In our sample area, three dairy farms adjacent to the Sugar and Little Sugar Rivers have gained CAFO permits since 2004 (Fig. 1). We use these expansions and water quality data, which derive from a program that employed citizen-scientists to collect water samples that were then analyzed by the Wisconsin Department of Natural Resources (WDNR), to estimate the effects of CAFO proximity and stream position on TP concentrations and loads. Our results suggest that after CAFO expansion, downstream TP concentrations and loads increased by 19% and 91%, respectively, relative to upstream monitoring locations. The larger treatment effects for loads than concentrations imply that the rate of downstream P transfer has increased due to CAFO expansion. Finally, we discuss what these results imply for monitoring and managing TP in this and similar watersheds.

2. Background and data

2.1. Study area

The Sugar River watershed occupies 780 km^2 in the Southeast Glacial Plains of Wisconsin, a region dominated by glacial till and moraines deposited during the Wisconsin Ice Age. The underlying shallow bedrock is highly fractured Silurian age limestone with moderate to high groundwater exfiltration rates, resulting in moderate to high base flow contributions to total stream flows. Soil types range from clay and silt loams over shallow dolomite bedrock on elevated and rolling uplands to sandy loams in river bottoms. Further downstream, some interbedded peat soils are present (WI DNR 2005). This topography makes the river susceptible to fast runoff and flash floods as steep gradient feeder streams rapidly deliver runoff to the river (WDNR, 2005). The river flows 149 km southeast through Green and Rock Counties from headwaters in Dane County, draining an upper watershed of 554 km^2 (Amrhein 2015). Near Shirland, Illinois, the Sugar River joins the Pecatonica River then the Rock River before joining the Mississippi River which flows to the Gulf of Mexico (Fig. 1).

Historically, hardwood forests, prairies, savannas, and wetlands occupied the Sugar River Watershed. Today, the river flows through productive agricultural land (Fig. 1). Most tillable land is intensively farmed for dairy and cash-crop grains and vegetables, with roughly 60% of the watershed in row crops, 20% in pastures, and approximately 6% in forest (WDNR, 2005). Green County covers 1510 km^2 in southeastern Wisconsin and represents the epicenter for dairy within the "Dairy State." In 2010, the county supported a human population of 36,842 and a dairy cow population of about 31,000 (20.5 cows/km^2) (Green County 2012). The watershed supports some of the largest floodplain forest, wet prairie, and oak savanna remnants left in the region including the Avon Bottoms State Wildlife and Natural Area (Amrhein 2015). Although many wetlands have been drained, others remain along stream and river margins providing key resources to retain nutrients, reducing loads downstream (Chen et al., 2020). Remaining natural habitats support rare terrestrial and aquatic plants, insects, and grassland birds, justifying the Sugar River's designation as an "Exceptional Resource Water" by the WDNR.

2.2. Discharge, phosphorus, and precipitation data

We accessed stream discharge data for U.S. Geological Survey (USGS) gage station #05436500 on the Sugar River in Brodhead, WI. We obtained data on daily mean discharge levels (m^3/sec) from the USGS Water Science Center, National Water Information System Web Interface (USGS 2017). Stream discharge is defined as the volume of water passing a gage station in a river channel of known (surveyed) cross-sectional area. Stream discharge reflects inputs from precipitation, both as direct runoff and as base flow from ground water (Leopold et al., 1964). Direct runoff responds quickly to storm events (Bras 1990). Groundwater sources provide most of the baseline flow.

USGS gage station #05436500 is the sole historical source of daily discharge data in the Sugar River Watershed, so we use the drainage area ratio method to estimate discharge at all ungaged water quality monitoring locations in our sample. The drainage area ratio method is commonly used to estimate discharge for ungaged locations using streamflow data from nearby stations (e.g., Asquith et al., 2006) and has been shown to accurately predict daily streamflow, especially when estimating within a watershed for areas of similar soil and land use (Gianfagna et al., 2015). We apply standard weighting equations to USGS drainage area data for Wisconsin streams (Henrich and Daniel 1983).

Next, we accessed historical daily precipitation data for Brodhead Station from the PRISM Climate Group (2020). Because precipitation is highly correlated with discharge and likely impacts surface water TP concentrations and loads, we control for daily precipitation and the sum of precipitation over the seven preceding days in our multivariate analyses. We note that results are not sensitive to the inclusion of these precipitation controls but previous studies (e.g., Burkholder et al., 2007; Raff and Meyer 2019) show that precipitation affects TP concentrations, so their inclusion can help reduce residual variation and improve precision.

Stream discharge, surface water TP concentration, and their product (total daily load) all vary widely in the Sugar River (Table 1). Within our sample and across all monitoring locations, the average daily discharge is 5.88 m^3/s (minimum: 0.0345). Daily discharge at Brodhead Station peaked at 122.3 m^3/s on May 25, 2004, reflecting a heavy precipitation event (when peak P loads also occurred). Historically, mean and minimum stream discharge levels on the Sugar River increased over the last century, reflecting changes in both climate (WICCI 2011 Chap. 3) and land use (Fig. 2).

Table 1. Summary statistics for the variables used in the analyses. There are 667 observations (monitoring location-days). "Downstream" refers to the monitoring locations downstream from the nearest CAFO (vs. upstream). "Treatment" refers to monitoring location-days downstream of a CAFO after its expansion (Downstreamij×Postijmt in the empirical model specification).

Variable	Mean	SD	Min	Max
Total phosphorus concentration (mg/L)	0.116	0.097	0.020	1.41
Discharge (m^3/second)	5.88	9.02	0.0345	115.5
Phosphorus load (concentration × discharge) [kg/day]	83.18	256.96	0.148	4801
Distance to CAFO (miles)	3.644	1.449	0.143	7.102
Downstream	0.207	0.405	0	1
Treatment	0.169	0.375	0	1
Precip. (cm, daily)	0.24	0.76	0	8.15
Precip. (cm, preceding 7 days)	1.94	2.20	0	14.0

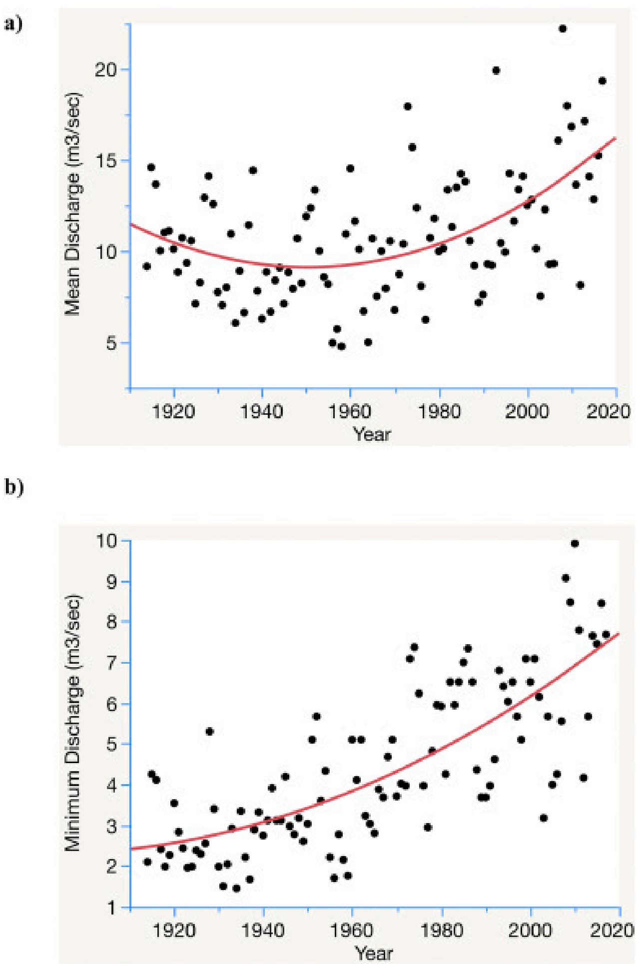

Fig. 2. Increases in discharge volume on the mainstem of the Sugar River over the past century from USGS data. a) Quadratic regression of mean discharge on year (r^2 = 0.27, both terms p < 0.001), b) Quadratic regression of minimum discharge on year (r^2 = 0.58, linear term p < 0.001, quadratic term p = 0.036).

Our final dataset includes information on TP concentrations from 667 water quality samples collected from 40 monitoring locations in the Sugar River Watershed between 1995 and 2017. The dataset represents an unbalanced panel, because the water quality samples were taken at irregular intervals. The median number of TP measurements per monitoring location is 12 (mean: 16.7). Earlier data (before 2012) mostly reflect samples collected by WDNR and USGS. Other pre-2012 TP samples and all samples since 2012 derive from USEPA's STORET database collected by the USGS using both regular interval and storm-event sampling. To estimate daily loads, we multiplied discharge (historical data for Brodhead Station and drainage area ratio estimates for all other monitoring locations) by measured P concentrations that day using the USGS FLUX procedure (USGS 2005).

Throughout our sample period, TP concentrations and loads in the watershed fluctuated widely. Total P concentrations between 1995 and 2017 averaged 0.116 mg/L (Table 1). More than 50% of the observations exceed the “impaired” limit (WI DNR, 2010; Fig. 3a). Total daily P loads increase more than proportionally to stream discharge, as TP concentrations increase with discharge ($r = 0.329$, $p < 0.001$). Total P loads ranged from 0.148 to 4801 kg/day (mean: 83 kg/day). The lowest TP load occurred on June 28, 2017, but most low loads occurred in winter when manure spreading is rare and frozen ground and low precipitation limit erosion and runoff. Peak TP loads coincided with extreme rainfall events, usually between mid-May and mid-July. The highest daily load coincided with the day of peak discharge (May 25, 2004) and was more than double the amounts estimated for any other day.

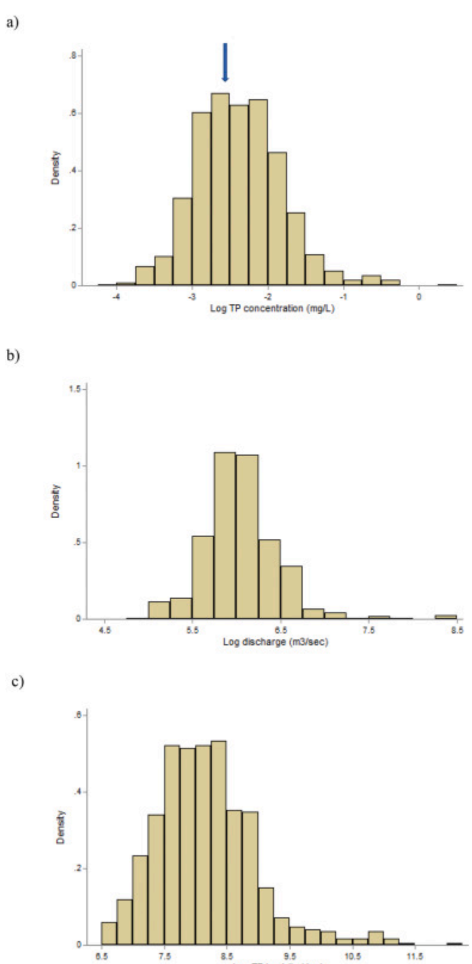

Fig. 3. Distributions of total phosphorus (TP) concentrations (a), stream discharge volumes (b), and phosphorus loads (c) on the mainstem of the Sugar River observed between 1995 and 2017. All original distributions were

highly skewed and are thus shown on a log scale. The vertical arrow in a) shows the established standard for maximum allowable median TP concentration (0.075 mg/L).

2.3. Farm expansions

Over the period of this study, dairy operations and their numbers of animals increased considerably in the watershed. Three dairy farms became CAFOs in Green County during our study period. We obtained sizes and locations of these operations from the WDNR database of CAFO Wisconsin Pollutant Discharge Elimination System (WPDES) permits (Bauman 2017). CAFO sizes are measured in animal units (AU), which estimate the potential impacts (in terms of grazing and manure production) for different species of livestock. One AU is defined as a 1000 pound beef cow with a daily dry matter forage requirement of 26 pounds (Minnesota Department of Agriculture 2017). According to WPDES permits, three large dairies operated in our study area. Plainview Stock Farm had approximately 1000 AU from August 2008 to May 2013 before expanding to 1600 AU (June 2013 to present). Spring Grove Farm managed 1000 AU (July 2004 to February 2013) and 2574 AU (March 2013 to present). Valley Mead Farm managed 1758 (March 2011 to present). We estimate that these operations supported a mean of 350 AU before the earliest WPDES permit date. Permits are only required for operations at or above 1000 AU and must be renewed every five years. We used ArcGIS software to classify the monitoring stations as up- or downstream of the dairies. As shown in Table 1, nearly 21% of monitoring location-days indicate readings downstream from these three dairy operations, regardless of their CAFO status. To measure distances between each CAFO and monitoring location, we used ArcGIS to calculate the geodetic distance between the two points. The monitoring location to dairy operation distances ranged between 0.14 and 7.1 miles, with a mean of 3.6 miles. Tracking trends in livestock numbers across the watershed was impossible as neither the country nor the state collect and track such data. Because CAFOs are distributed throughout this region of Wisconsin (Raff and Meyer 2019), we could not identify a "control" watershed lacking CAFOs with similar characteristics, e.g., soils, land use, depth to bedrock. We therefore applied a before-after, control-impact (BACI) design to leverage our knowledge of when CAFO operations expanded and the up- vs. downstream locations of monitoring locations (Underwood 1991). That is, we use the period before these CAFOs were established as a baseline (or control) to evaluate their effects (impacts).

3. Methods

To estimate how precipitation and proximity to dairies affect TP outcomes (concentrations and loads) in the Sugar River Watershed, we modeled TP outcomes as a function of stream discharge and the distance between sampling locations and particular dairies. Cross-sectional variation allows us to estimate how time invariant factors (e.g., distances from dairy operation) affect TP levels. We also leverage time series variation and the BACI design to identify the causal relationship between CAFO presence and TP outcomes.

3.1. Multivariate analyses

To characterize the relationship of TP to discharge and dairy proximity, we separately estimated three regression models for both concentration and load using ordinary least squares (OLS). The first (TP-D) examines the effects of distance from each monitoring location to the nearest dairy. This specifies TP outcomes as:

(1)

$$ln(TP)_{ijdmt} = \beta_0 + \beta_1 Distance_{ij} + \beta_2 lnDischarge_{dmt} + \psi_m + \lambda_t + \varepsilon_{ijdmt},$$

where TP_{ijdmt} represents the TP concentration (in mg/L) or load (kg) for monitoring location i near dairy operation *j* for day d in month m of year t. β_0 is the intercept term and ε_{ijdmt} is the error. We log-transform TP outcomes and discharge as the distributions of these variables are highly skewed (Fig. 3). We include indicator variables ψ_m and λ_t to account for seasonal effects (month) and longer-term trends (years) common to all monitoring locations. This controls for any factors (e.g., shifts in policy, land use changes) that vary systematically over time. Other factors control for precipitation and drainage basin (when load is the outcome) as these factors covary with discharge and could affect outcomes. Controlling for these factors reduces bias in our estimates. We clustered standard errors at monitoring locations to account for potential serial autocorrelation.

Our second model (TP-L) examines how monitoring location relative to the nearest CAFO dairy operation affects TP outcomes. We again use OLS to estimate:

(2)

$$ln(TP)_{ijdmt} = \beta_0 + \beta_1 Downstream_{ij} + \beta_2 lnDischarge_{dmt} + \psi_m + \lambda_t + \varepsilon_{ijdmt},$$

where $Downstream_{ij}$ is a binary variable indicating whether monitoring location i is downstream of the nearest dairy operation *j*. Instead of using a "control" watershed with no CAFOs or dairy farms, we use upstream sites as controls by using $Downstream_{ij}$ to identify monitoring locations "treated" by dairy exposure (other notation and controls as in equation (1)).

Our third model (TP-L + CAFO) extends the analysis from equation (2) to examine how results differ among CAFOs. This provides information about which CAFOs most affect surface water quality. Again, we use OLS to estimate:

(3)

$$\begin{aligned} ln(TP)_{icdmt} = \beta_0 + \beta_1 Downstream_{ij} + \boldsymbol{\beta}_2 Dairy_j + \boldsymbol{\beta}_3 (Downstream_{ij} \times Dairy_j) + \beta_4 lnDischarge_{dmt} \\ + \psi_m + \lambda_t + \varepsilon_{ijdmt}, \end{aligned}$$

where $Dairy_j$ represents each of the three dairy operations. Here, the $Downstream_{ij} \times Dairy_j$ interaction serves as a dummy variable to indicate whether the TP outcome comes from a monitoring location downstream of the closest dairy. The β_3 coefficient thus reflects the differential between up- and downstream TP concentrations and loads relative to the reference dairy (other notation and controls as in equation (1)).

3.2. Quasi-experimental estimation

Coefficients for discharge and dairy proximity in the models above estimate how strongly thee predictors affect TP outcomes. However, these models could omit certain factors, potentially biasing these estimates. The up- vs. downstream analysis identifies effects using only cross-sectional differences in discharge and dairy proximity, which could be biased if other factors consistently differ between up- and downstream monitoring locations, e.g., nearby slope or cover. As one example, if the upstream location happened to occur in a shallow, fast-moving portion of the stream while the downstream location occurred in a deep, slow-moving part, an increase in TP concentrations or loads might be attributed to the CAFO rather than to stream morphometry. To address such potential omitted variables, we analyzed a difference-in-differences model (Card and Krueger 1994), termed DD-L, to represent our quasi-experimental (BACI) design:

(4)

$$\begin{aligned} ln(TP)_{ijdmt} = \beta_1 Downstream_{ij} + \beta_2 Post_{ijmt} + \beta_3 (Downstream_{ij} \times Post_{ijmt}) + \beta_4 Discharge_{dmt} + \delta_i \\ + \psi_m + \lambda_t + \varepsilon_{ijdmt}, \end{aligned}$$

Here $Post_{ijmt}$ indicates the period after dairy operation *j* associated with monitoring location *i* expanded to become a permitted CAFO. The $Downstream_{ij} \times Post_{ijmt}$ DID interaction term reflects the quasi-experimental treatment, making β_3 the coefficient of interest. We include the δ_i term to represent the fixed effects of monitoring location. These location effects control for time-invariant differences among monitoring locations (e.g., topography) to reduce error variance and increase the precision of our estimates (other notation and controls as in equation (1)).

Our DID model imitates an experimental design for observational data, estimating the differential effect on a treatment group relative to a control group. It compares the change in TP outcomes in locations downstream of a CAFO to the change in TP outcomes in locations upstream of a CAFO before and after the CAFO was permitted. This approach allows us to plausibly identify causal effects of CAFO expansion on TP outcomes by "netting out" permanent differences and common trends across up- and downstream monitoring locations. We present results from both this model and an alternative model that replaced monitoring location effects with dairy operation effects, γ_j (DD-CAFO), again clustering standard errors at the monitoring location level.

4. Results

4.1. Precipitation and CAFO proximity affect phosphorus concentrations and loads

Total P concentrations and loads in the Sugar River Watershed vary seasonally, peaking in spring or mid-summer (SI - Table A1). Increases in TP concentrations are strongly linked to stream discharge in all three statistical models (Table 2, Panel A). Next, TP concentrations and loads declined with distance from each monitoring location to its nearest large dairy operation (Model TP-D, Table 2, Panels A and B). Each additional mile from the nearest dairy dropped average TP concentrations by 6.5% and average TP loads by 6.9% (for regressions involving log-transformed dependent variables, we interpret a one-unit change in each coefficient as an e^{β}-1% change in the outcome).

Table 2. Results from the three statistical models analyzing variation in (log) TP concentrations (Panel A) and loads (Panel B). Values show coefficients for each predictor variable and robust standard errors (in parentheses). Precipitation, long-term and seasonal trends (year and month indicator variables), and drainage area (Panel B only) were accounted for in these models but are not shown here. Plainview Stock Farm serves as the reference farm (hence is omitted here). N = 667. Significance levels: ***p < 0.001, **p < 0.01, *p < 0.05.

Variable	**TP-D**	**TP-L**	**TP-L + CAFO**
Panel A: Concentrations			
Discharge (logged) [m^3/second]	0.104*** (0.0242)	0.129*** (0.0211)	0.137*** (0.0214)
Distance to CAFO (miles)	−0.0679* (0.0320)		
Downstream		0.353* (0.180)	−0.0914 (0.0843)
Spring Grove Dairy			−0.0352 (0.0468)
Valley Mead Farm			−0.0619 (0.136)
Downstream × Spring Grove Dairy			0.829*** (0.109)
Downstream × Valley Mead Farm			0.564* (0.237)
R-squared	0.459	0.463	0.491
Panel B: Loads			
Distance to CAFO (miles)	−0.0719* (0.0327)		

Downstream		0.393* (0.185)	−0.0367 (0.0969)
Spring Grove Dairy			0.0312 (0.0563)
Valley Mead Farm			−0.0148 (0.128)
Downstream × Spring Grove Dairy			0.833*** (0.107)
Downstream × Valley Mead Farm			0.495 0.273
R-squared	0.937	0.938	0.940

Monitoring locations downstream of large dairy operations have higher TP concentrations and loads than those upstream (Model TP-L, Table 2, panels A and B). On average, TP concentrations and loads increased by 42% and 48%, respectively, downstream of CAFOs (ignoring distance to CAFOs in this model). Model TP-L + CAFO explored the effects of individual CAFOs in more detail. In this model, Plainview Stock Farm is the omitted category (serving as the reference category). The insignificant coefficients on "downstream" indicate that TP concentrations and loads above and below that farm did not differ (Table 2, panels A and B). Total P concentrations and loads upstream of the other two dairies (Spring Grove Diary and Valley Mead Farm) do not statistically differ from those upstream of Plainview Stock Farm. However, TP concentrations (loads) downstream of Spring Grove Dairy were 129% (130%) above those immediately upstream while those downstream of Valley Mead Farm were 76% (64%) above its reference conditions. These CAFOs thus appear to increase downstream nutrient loads appreciably.

4.2. DID model results

Both DID models showed increases in TP concentrations downstream of CAFOs that have expanded operations (Table 3, columns 1 and 2). In this quasi-experiment, DID Model DD-L found that downstream TP concentrations increased by an average of roughly 19% relative to those upstream once dairy farms expanded to become permitted CAFOs. Systematic changes in other factors affecting overall TP concentrations (e.g., changes in pollution from other agricultural or residential sources upstream of the study area) are controlled for in the month and year fixed effects here to prevent bias. We can thus attribute increases in downstream TP concentrations to dairy operations expanding to become CAFOs. Note that results for the two concentration DID models agree qualitatively but that model DID-L (which included the effect of monitoring location) has more power, producing more precise estimates than the DID-CAFO model (that instead included effects of individual CAFOs). The DID load models also showed TP loads increasing downstream of CAFOs (Table 3, columns 3 and 4). The estimated effects of CAFO expansion are large with TP loads increasing by 91% in model DD-L and by 54% in model DD-CAFO.

Table 3. Results from the DID models (equation (4)) analyzing (log) TP concentrations and loads across all observed monitoring locations. Values show coefficients for each explanatory variable, significance levels, and robust standard errors (in parentheses). "Treatment" refers to monitoring locations downstream of dairies after they expanded to become permitted CAFOs. Models DD-L and DD-CAFO both treat the regressors shown as fixed effects and include (log) discharge, precipitation, and long-term and seasonal trends (year and month) as covariates. Drainage area controls were included where load is the outcome. N = 667. Robust standard errors appear in parentheses. Significance levels: ***p < 0.001, **p < 0.01, *p < 0.05.

Variable	**DD-L**	**DD-CAFO**	**DD-L**	**DD-CAFO**
	Conc.	**Conc.**	**Load**	**Load**

Treatment	0.178*** (0.0656)	0.374* (0.192)	0.648*** (0.0977)	0.434* (0.194)
Year included	Yes	Yes	Yes	Yes
Month included	Yes	Yes	Yes	Yes
Monitoring location included	Yes	No	Yes	No
CAFO included	No	Yes	No	Yes
R-squared:	0.614	0.467	0.949	0.938

5. Discussion

5.1. Effects of discharge and dairy expansion of TP concentrations and loads

The changes we document here in the Sugar River watershed may reflect changes broadly across Midwest. If so, the region has experienced substantial increases in stream discharge, nutrient concentrations, and especially nutrient loads in recent decades (Fig. 2, Fig. 3). These reflect shifts in land use, agricultural practices, climate, and hydrology. Throughout the Midwest, increases in tile drainage, expanded annual row crops, and losses of wetlands have increased mean and low-flow discharge in rivers and watersheds (Apfelbaum 1993; Apfelbaum et al., 2012). Precipitation has also increased over the last half-century, increasing discharge from the Sugar River and in many other southern Wisconsin watersheds (WICCI 2011; Mallakpour and Villarini 2015).

The increases in TP concentrations and loads observed in the Sugar River reflect, in part, increases in minimum and peak discharge. They also coincide with increases in animal production and manure spreading in the watershed. In this study, TP concentrations and loads increased downstream of large dairy operations. Although differences among monitoring locations complicate efforts to link these increases to particular sources, our DID analyses teased out the clear signal that TP concentrations and loads have increased substantially downstream of dairy farms that recently expanded to become CAFOs. Because our analysis controlled for permanent differences among monitoring locations as well as systematic trends across the watershed, we can credibly infer that these CAFOs substantially increased TP concentrations and loads in the Sugar River Watershed.

In the Sugar River and its tributaries, TP concentrations and loads now consistently exceed Wisconsin standards. Concomitant increases in discharges have accelerated these increases in TP loads. This means that even if median concentrations meet Wisconsin's stream standards, the watershed will continue to release increasing loads of P. We also confirmed that much of the annual P loads to this river occur during a few high-flow events (as noted by Carpenter et al., 1998 for other systems). Because the highest (>90th percentile) discharges account for >80% of TP export, reducing in-stream nutrient loads even by 50% at low discharge would do little to reduce annual nutrient exports (Royer et al., 2006).

Habitats downstream of the dairies are threatened in many ways by increased nutrient loads. Most conspicuously, high P loads trigger eutrophication and harmful algal blooms in surface waterbodies, threatening fish, wildlife, and drinking water supplies. These effects extend to the mouth of the Mississippi River, where elevated nutrient inputs support giant algal blooms whose decay deoxygenates surface waters, creating the notorious “dead zone.” In addition, manure applications degrade stream quality by introducing pathogens and antibiotics that impair fisheries, drinking water quality, and human health (Burkholder et al., 2007). Several introduced plant species also thrive when nutrient levels increase (e.g., reed canary grass, Apfelbaum and Sams 1987).

Future increases in the frequency and intensity of extreme rain events further threaten the quality of surface water in the studied region. Major rain and snowmelt events amplify P loads by increasing both TP concentrations and discharge. Historically, such peak events only occurred about once per year in the Sugar

River Watershed. In 2018 and 2019, however, the Sugar River experienced four bank-full discharges, confirming that these events are growing more frequent.

5.2. Implications for policy and management

Within the U.S., legal pressures are growing to adopt more strenuous water quality improvement efforts based on Total Maximum Daily Loads (Henry 2020). Wisconsin water quality standards, however, are based on median TP concentrations, ignoring the effects of increasing discharge on nutrient loads. Focusing on concentrations (or failing to measure these during peak events) lead us to inappropriately focus on improving practices that actually do little to reduce total nutrient loads which primarily occur during peak-flow events.

Issues surrounding manure applications and CAFOs continue to attract considerable legal and policy attention in the U.S. and Wisconsin. A recent lawsuit failed to resolve issues related to how manure applications impair waters of the state (Wisconsin Dairy Business Association vs. DNR, settled Jan. 11, 2017). The land needed to support local manure spreading without increasing nutrient runoff was 37% of all cropland and 75% of all tilled croplands in Wisconsin even before recent expansions in permitted CAFOs (Saam et al., 2005). Additional CAFOs, including a new 6000-cow dairy (more than double the size of the current largest dairy), further threaten water quality in the Sugar River and downstream ecosystems.

Best management practices devised to reduce TP concentrations and loads include no-till cropping, over-crop and after-harvest cover crops, maintaining high (>90%) crop residue levels, and planting riparian buffer zones. Such BMPs can effectively reduce runoff and soil erosion during peak events. Converting even 10% of a field to diverse, native perennial vegetation, for example, can reduce sediment movement by 95% and runoff of TP and N by 90% and 85%, respectively (Schulte et al., 2017). BMPs also seek to limit how much manure is applied per acre and limit manure spreading on frozen ground and during snowmelt. Applying stabilized P fertilizers that do not break down under anaerobic or acidifying soil conditions would also help reduce dissolved reactive P loads during low streamflows when eutrophic conditions prevail.

Although these BMPs show promise, many are recommended rather than required. We thus need to improve both how widely they are adopted and our ability to evaluate their effectiveness. In reviewing practices for reducing P inputs to Lake Erie, Wilson et al. (2019) found that although most farmers are willing to adopt recommended practices, they have few incentives to do so. The lack of cost-benefit information and technical assistance (including site-specific decision support tools) appear to prevent wider adoption.

As better practices begin to be adopted, we will also need to monitor how effective particular BMPs are under various conditions and in different agricultural contexts. Applying particular BMPs as experiments and comparing these across watersheds would allow us to gauge their effects under real-world conditions. As outcomes from particular BMPs become known, we should design appropriate incentives to promote their adoption. The success of these efforts should also be monitored, allowing us to progressively adjust how we implement BMPs through cycles of adaptive management.

Regulatory agencies play a key role in monitoring and controlling nutrient loads. We must regularly measure discharge and TP concentrations during peak events to accurately estimate total loads. These varied by more than 200-fold in the Sugar River over the 23-year period we studied. Inexpensive automated samplers or sensors installed widely within and among watersheds could prove quite useful. Deploying edge-of-field sensors (e.g., up- and downstream of CAFOs) would improve our ability to assess local and seasonal effects of different agricultural practices. We should also exploit the experience and perspective that farmers have in designing improved BMPs and monitoring programs. This would increase their interest and support for BMPs, increasing the likelihood they will succeed.

Current regulations have yet to effectively control TP concentrations and loads in the Midwest. This Sugar River case study demonstrates how important it is to evaluate in tandem the many factors affecting water quality. Limits on the amounts and timing of when manure is spread, cover crops, and other BMPs clearly have merit, but volunteer adoption of BMPs has proved inadequate to prevent substantial declines in water quality. We must therefore contemplate just what regulations are needed, particularly for CAFOs as they continue to proliferate and grow in size. If state regulations prove too weak to reduce nutrient runoff from farms, federal regulation may be needed.

Novelty and relevance statement

The paper presents an unusually long-term (23-year) longitudinal case study of how phosphorus concentrations and loads in a Midwestern agricultural watershed have increased in response to both recorded increases in extreme rain events and intensified land use (dairies expanding to become CAFOs). Water quality monitoring stations above and below these large-scale dairy operations and our quasi-experimental Before-After, Control-Impact design allowed us to isolate and quantify these effects. The increase in phosphorus concentrations with river discharge strongly amplify variation in total daily phosphorus loads (total variation: 240-fold). These findings confirm the importance of adequate monitoring and the importance of regulating nutrient runoff from agricultural fields, especially during peak rainfall events. Limiting CAFO expansions and enforcing the wider use of key Best Management Practices on surrounding agricultural lands could greatly enhance nutrient retention and prevent further declines in water quality. This detailed, long-term case study of one agricultural watershed likely reflects findings general to temperate-zone agricultural lands around the world.

Credit statement

Donald M. Waller: Conceptualization, Methodology, Formal analysis, Writing – original draft. Andrew G. Meyer: Data curation, Formal analysis, Methodology, Writing – review & editing. Zach Raff: Data curation, Formal analysis, Methodology, Writing – review & editing. Steven I. Apfelbaum: Conceptualization, Methodology, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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