Phosphorus Flows and Balances for the Lake Mendota and Yahara River Watersheds: 1992-2017

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Cover photo: View of Yahara River estuary on northern end of Lake Mendota

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Executive Summary

This study presents an analysis of watershed-level phosphorus (P) mass balances from 1992 to 2017 for the Lake Mendota Watershed (LMW) and Yahara River Watershed (YRW) in south-central Wisconsin. P accumulation rates declined substantially in both watersheds over the time period (from 0.83 million kg-P/year to 0.13 million kg-P/year – an 85% decline – in the LMW and from 1.2 million kg-P/year to 0.15 million kg-P/year – an 88% decline – in the YRW). While declines in agricultural fertilizer P imports were the largest drivers of accumulation rates declining, increases in exports of livestock and manure digester products were also important in the LMW and increases in net crop export and exports of livestock products were also important in the YRW.

Introduction

Management of water quality in the Yahara Lakes and Yahara River Watershed in Dane County has been an intensive activity in the region for well over a century [*Gillon et al.*, 2016; *Lathrop*, 2007]. In the late 1800s to early 1900s, untreated human sewage was the primary source of water quality impairment in the lakes as the Madison metropolitan area increased in population without any treatment of human waste. To alleviate the public health crisis, considerable public resources were expended in the middle of the 20th century to both build cutting-edge wastewater treatment facilities and divert treated wastewater downstream of the lakes [*Mollenhoff*, 2003]. However, as this initial source of water quality impairment was being brought under control (at least from the perspective of the lakes), non-point sources of nutrients from agricultural lands were increasing with the intensification of crop and dairy production in the watershed following World War II [*Gillon et al.*, 2016; *Lathrop*, 2007]. In addition, non-point sources of sediment and nutrients were also increasing during this time as urban development and residential construction expanded with the growth of the Madison metropolitan area.

Today, despite renewed and highly collaborative efforts that are leading to positive changes in both agricultural and urban land management practices, the region is still challenged by the goal of substantially reducing nutrients - primarily phosphorus (P) as the driver of lake eutrophication [*Carpenter and Lathrop*, 2014] - in the lakes and river [*Wardropper et al.*, 2018]. This wicked problem is especially difficult in the face of counteracting drivers such as intense precipitation events increasing in frequency [*Carpenter et al.*, 2018] and more intensification of dairy production in the Upper Yahara Watershed [*Gillon et al.*, 2016; *Larson and Sharara*, 2016].

One critical component of water quality management is monitoring and assessment of the drivers of water quality [*Rissman and Carpenter*, 2015]. In the realm of non-point source P pollution, monitoring and assessment of these drivers is typically focused on practices that reduce the transport of P from land to water. These transport-focused practices include use of cover crops or no-till on cropland (which reduces soil erosion) or riparian buffers (which reduces connectivity between agricultural land and streams). Monitoring and assessment of these transport-focused practices can be painstaking but is fairly straightforward as it is dependent on land management monitoring (aided by increasing submission of nutrient management plans (NMP) and recently through advancements in remote sensing) and familiar, albeit uncertain, scientific modeling tools that estimate P loss in surface runoff (e.g., SnapPlus).

However, less attention has been paid to monitoring and assessment of supply-focused practices that aim to reduce the supply of P available for transport in the first place. While certain strategies like NMPs will implicitly account for the supply of P available in a given cropping system, a more holistic focus on reducing P supply requires an integrated investigation of the P mass balance in a certain region of interest (typically a watershed). Use of the mass balance approach is particularly important as more research highlights the substantial role of legacy P – that accumulates when the mass balance is positive – in water quality impairment [*Motew et al.*, 2017].

Many regions across the United States and beyond [e.g., *Peterson et al.*, 2017; *Wironen et al.*, 2018] have begun to add P mass balance monitoring and assessment to their water quality management portfolio as a complement to other transport-focused efforts. While most of these studies are primarily focused on agricultural sources of P, new investigations have highlighted the important role of urban and human sources of P [e.g., *Sabo et al.*, 2021]. While P mass balances are not able to pinpoint acute locations of P transport risk like other transport-focused tools, they can be very effective at highlighting opportunities to reduce long-term imbalances of P entering and leaving the watershed connected to agricultural and urban activities.

Two previous P mass balance investigations have been conducted in the Lake Mendota Watershed. Bennett [1999] looked at the balance for a single point in time (1995) and estimated a large P accumulation rate. Kara et al. [2012] used similar methods to look at an updated snapshot in time (2007) and estimated that the P accumulation rate was lower than that estimated by Bennett [1999] but was still greater than zero. This study also explored various scenarios of changes in P flows to assess their impact on the overall balance. While similar methods were used between the two studies, it is still challenging to assess changes through time in the watershed mass balance. In addition, the livestock inventories used in the previous studies have been found to be underestimates [*Booth and Kucharik*, 2021].

In this study, a monitoring and assessment framework was developed that uses up-to-date, publicly available, and consistent-through-time datasets and models to create a P mass balance inventory for both the Lake Mendota and Yahara River watersheds through a recent 25-year period from 1992 to 2017 (in 5-year increments). This novel investigation of changes in the watershed P balance through time is designed so that it can be updated periodically (every 5 years) into the future as a water quality management tracking and assessment tool that complements other existing tools in the region.

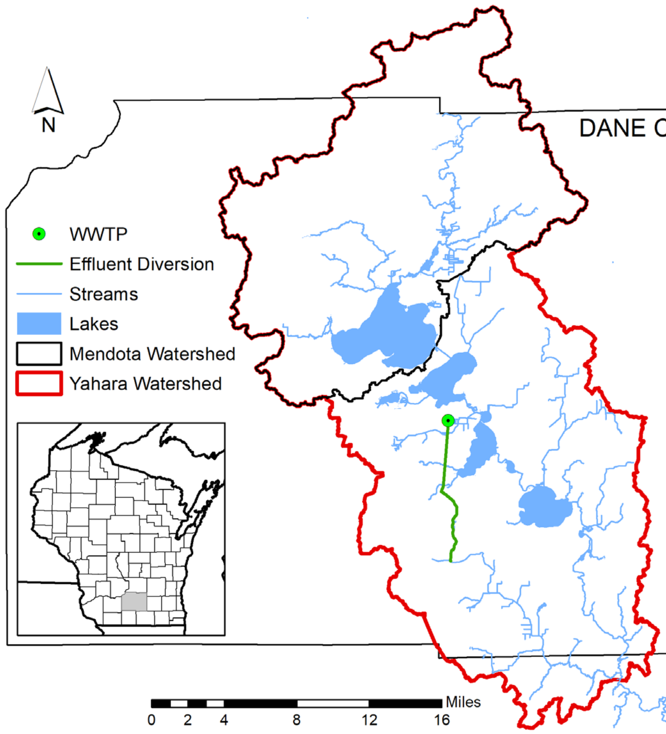

Figure 1. Map of Lake Mendota and Yahara River Watersheds situated within Dane County and the state of Wisconsin (inset map). The Nine Springs WWTP is also shown with the effluent diversion path to Badfish Creek.

Objectives

This report presents all data and methods used to construct a P mass balance for the Lake Mendota and Yahara River watersheds from 1992 to 2017. The developed methodology and framework are intended to represent an ongoing water quality assessment tool that indicates potential risk and improvement in long-term water quality outcomes connected to the supply of P available for transport within the two watersheds. This assessment tool is not meant to replace any other existing tool, but instead is a valuable addition and complement to the existing set of tools being implemented by various actors and organizations within the region. Specific objectives outlined by the county are below:

1. Identify the sources and amounts of P being imported and exported from the Yahara Watershed on an annual basis including developing methodologies and procedures for the quantification of identified sources
2. Develop a model that captures the movement of P into and out of the watershed to determine if an imbalance exists between imports and exports as well as the corresponding amount

Methods

Study Area

The Yahara River Watershed (YRW) is an urbanizing agricultural watershed that spans the central majority of Dane County. The northern part of the watershed drains to Lake Mendota and is dominated by dairy production with associated land uses of corn, alfalfa, and soy and manure application. The central part of the watershed is dominated by urban land uses and represents the core of the Madison metropolitan area that includes the cities of Madison, Middleton, Fitchburg, and Monona. The southern part is dominated by commodity grain (corn and soy) agriculture with substantially less livestock.

The Lake Mendota Watershed (LMW) - nested fully within the upper part of the YRW - was the subject of the two previous P mass balance studies and is typically highlighted in water quality management due to its disproportionate contribution of phosphorus relative to the drainage areas to the other Yahara Lakes (Lathrop and Carpenter). The Yahara River Watershed is also the subject of considerable interest due to an ongoing TMDL project that aims to reduce P loading at its confluence with the Rock River downstream of the lakes. Both watershed P mass balances represent important assessment scales in that the LMW represents a dominant driver of Yahara Lakes water quality and the YRW represents the integration of both agricultural and urban P flows with implications for water quality in the downstream portions of the Yahara River and Rock River.

Imports of agricultural P to the LMW and YRW considered in this analysis of changes from 1992 to 2017 (in 5-year increments) include Net Feed Demand (which represents the difference between total livestock feed demand and crop production within the watershed), Agricultural Fertilizer, and Agricultural Pesticides. Crop production is accounted for within the Net Feed Demand component because a substantial share of the crops grown in the watersheds are fed to livestock within the watershed and never cross the watershed boundary. Exports of agricultural P include Net Crop Export, Livestock Products (meat and milk), Manure Export (through conventional land spreading), and Digester Export (manure digestate products). Imports of non-agricultural P include Urban Fertilizer, Food/Household Demand, Biosolids, Pet Feed Demand, and Atmospheric Deposition. And finally, the export of non-agricultural P considered is Stream Export. The following sections describe each of these flows and the methodology for their quantification in greater detail.

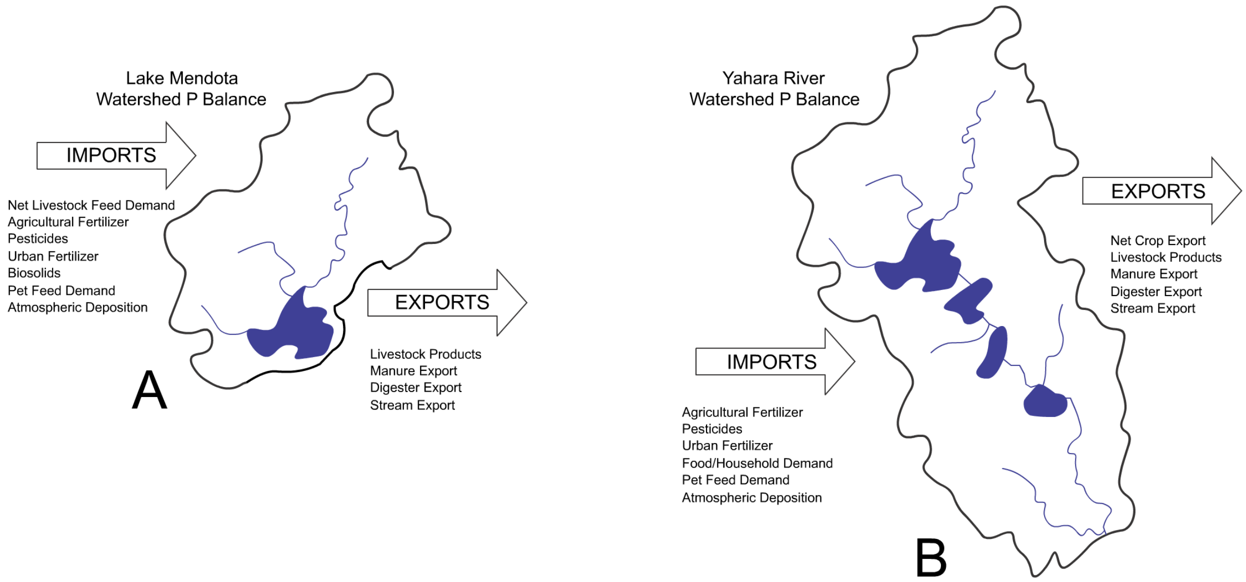

Figure 2. Conceptual diagram showing all P imports and exports considered in the P mass balance for the Lake Mendota Watershed (A) and the Yahara River Watershed (B).

Agricultural P Imports

Net Livestock Feed Demand

Livestock within the LMW and YRW demand P in their feed for maintenance, lactation, growth, and reproduction. The source of this feed comes primarily from crops grown within the watershed but a smaller proportion is sourced from feed and supplements imported into the watershed. While crop production is typically treated as an export of P from the watershed, it is represented within the Net Feed Demand component to represent the flow of material more accurately across the watershed boundary. Therefore, a positive Net Feed Demand means that there is more feed P being imported than crop P being produced.

Feed demand is calculated based on the following variables: livestock population by distinct animal types, dry matter intake (DMI) for each animal type, and concentration of P in feed for each animal type. Livestock populations by distinct animal types were estimated from an analysis described in Booth and Kucharik [2021] that relies on a special tabulation data request from the USDA Census of Agriculture for the zip codes that best match the outline of the YRW. DMI for lactating cows was estimated based on a regression model using body weight and milk production as predictors [*Roseler et al.*, 1997]. For dairy heifers, a regression model using body weight as the sole predictor [*Hoffman et al.*, 2008] was used to estimate DMI. A DMI value of 1.16 kg/day for calves was used based on survey data from Thoma et al. [2013]. For all other animal types (dry cows, beef cows, beef heifers, steers, bulls, and hogs), DMI was calculated based on the percentage of the body weight based on data from Peters et al. [2014]. Animal weights were estimated through time based on data from the USDA-ERS [2020].

Feed P concentration changed through time for lactating cows and was held constant for all other animal types. The feed P concentration for lactating cows was assumed to be 0.55% for

1992-1997 and 0.35% for 2002-2017 based on expert guidance [pers. comm., Randy Shaver, UW-Madison Dairy Science]. High P concentration diets were normal in the 1990s and were believed to positively impact reproductive success [*Howard and Shaver*, 1992] until research determined otherwise in the mid-1990s. A popular dairy trade magazine featured an article advocating for reduced mineral P feed supplementation [*Shaver and Howard*, 1995] and this led to a rapid change in feeding strategies throughout the dairy sector because of the associated cost reductions. Dry cow and dairy heifer feed P concentration was determined based on local feed dealer expertise [pers. comm., Bob Hagenow, Vita Plus Corp.]. Feed P concentrations were determined based on NRC [2000] for all beef cattle and NRC [2012] for all swine.

Crop production P was determined by estimating acreage and yield of major crop types for each year and multiplying together with standard P removal rates [*Laboski and Peters*, 2012]. Crop type area determination began with a 2012 dataset created for the YRW by Booth et al. [2016] that used county land use data to map crop types more accurately including corn, soy, alfalfa, wheat, hay, and pasture. Hay and pasture, in particular, suffer from poor accuracy in most remotely-sensed land cover products [*Lark et al.*, 2017] with many urban lawn landscapes classified as hay. The 2012 dataset was then used to adjust cropland area determined for 1992-2017 using the LCMAP land cover product [*Brown et al.*, 2020; *USGS*, 2020]. Crop type proportions – determined using the 2012 dataset – were assumed to be constant through time and corn acres were assumed to be 70% for grain and 30% for silage based on local nutrient management plan data [pers. comm., Matt Diebel, Dane County].

Crop yield estimates were obtained from USDA-NASS [2020a] and subsequently adjusted based on spatially-explicit (30-m) estimated crop yields for corn, soy, and wheat from Lark et al. [2020] who used soil properties as predictors.

Agricultural Fertilizer

P is an essential nutrient for crop growth and is very often imported to croplands from international mineral sources [*Cordell et al.*, 2009]. Agricultural fertilizer P imported to the LMW and YRW were determined using a spatially-explicit manure and fertilizer application model described in Motew et al. [2017] that uses point-scale livestock operation data to first spread manure on cropland within an estimated hauling radius and then apply fertilizer at different rates depending on crop type (demand) and whether manure was previously applied. Non-manured fertilizer P rates were adjusted through time based on county-scale farm fertilizer use data [*Falcone*, 2021]. For comparison with the P application model, average non-manured fertilizer P rates were determined for Dane County based on fertilizer data from Falcone [2021] and Census of Agriculture data [*USDA-NASS*, 2020b] on manured cropland area (for 2007) and total cropland and pastureland (for 1992-2017).

Agricultural Pesticides

While often considerably less than fertilizer P imports, pesticides are also an important – and often overlooked – source of P in agricultural regions [*Hebert et al.*, 2019]. We estimated pesticide P imports using county scale glyphosate application data for 1992-2017 from the USGS [*Baker and Stone*, 2015; *Stone*, 2013; *Wieben*, 2019]. P concentration in glyphosate was estimated to be 18.3% [*Hebert et al.*, 2019].

Non-Agricultural P Imports

Urban Fertilizer

Urban fertilizer is often used to enhance the growth of turfgrass and can represent a large flux of P to developed areas. While Dane County implemented a ban on fertilizer P used on established lawns (unless a soil test shows a need) in 2005 (Dane County Code of Ordinances Chapter 80), fertilizer P is likely still used in establishing turfgrass and other non-ideal situations. Urban fertilizer P imports were estimated using county scale non-farm fertilizer use data from Falcone [2021]. The county scale P mass values were then scaled proportionately based on developed area within Dane County and the two watersheds from 1992-2017 using LCMAP land cover data [*Brown et al.*, 2020; *USGS*, 2020]. The P mass was further scaled using estimated pervious cover within developed areas using NLCD land cover data from 2001-2016 [*MRLC*, 2021; *Wickham et al.*, 2017].

Food/Household Demand

Human diets represent an important direct driver of P flows across watershed boundaries in the form of imported food [*Metson et al.*, 2012]. In addition, P flows toward populated areas in the form of household products such as detergents – which have specifically been regulated in recent decades with considerable success [*Keiser*, 2020]. We estimate this food/household demand by using influent P data from the Nine Springs Wastewater Treatment Plant (WWTP) operated by the Madison Metropolitan Sewage District (MMSD) (pers. comm., Dave Taylor and Kim Meyer, MMSD). This influent P represents only the food/household P demand that enters the sanitary sewer system and not the proportion that is sent to the landfill (although regional landfills send their leachate to be treated at the WWTP). Influent P mass data was converted to influent P per capita based on estimated populations of the service area for 1998 and 2018 from MMSD. These per capita rates were then multiplied by the watershed populations from 1992-2017 based on block level data from the U.S. Census [*Manson et al.*, 2020] to get total influent P for the watershed. Food/household demand was only considered to be a P import for the YRW. The flux of P in food/household products for the LMW is largely short-circuited through the watershed as the WWTP effluent is discharged much further downstream in the YRW (Figure 1).

Biosolids

MMSD has a long history of applying biosolids from the wastewater treatment process to agricultural land in the Madison area. We considered Biosolids to be a P import for the LMW only because biosolids were assumed to be applied only within the YRW and represent an internal flux as food/household demand is separately accounted for in the YRW mass balance. Data was obtained from MMSD for biosolids P applied within the LMW for the years 2008-2016. In addition, the annual biosolids production volume was estimated for 2018 based on biosolids P concentration (available for 1996-2019) and P load data from MMSD. This annual volume was then assumed to be constant for the 1992-2017 time period. Biosolids P mass was then estimated for 1992-2017 assuming that biosolids P concentration for 1992-1995 was identical to 1996 and the fraction of biosolids applied in the LMW for 1992-2007 was equal to the average for 2008-2010.

Pet Feed Demand

Pet feed and waste is another often overlooked flux of P that can be a dominant import in urban areas and contribute to water quality impairment [*Hobbie et al.*, 2017]. Dogs were the only pets considered in this analysis as their waste is often not sent to the landfill or sanitary sewer system unlike cats and other indoor pets. Dogs per capita – assumed to be constant through the entire time period – was first estimated based on state-level estimates of the dog population for 2016 [*AVMA*, 2018] and the Wisconsin human population from the U.S. Census [*Manson et al.*, 2020]. The dog population of the LMW and YRW were then estimated by multiplying the dogs per capita rate by the watershed population based on the U.S. Census [*Manson et al.*, 2020] (linear interpolation was used in between the decennial years). Dog feed P demand was then calculated following Baker et al. [2007] given a P content of 0.8% and the average weight of a dog from Greer et al. [2007]. Dog feed demand was assumed equal to dog waste production assuming a constant body mass of the dog population. Dog waste was then adjusted to remove the proportion that is picked up and sent to the landfill or sanitary sewer system based on survey data from Swann [1999].

Atmospheric Deposition

P from the atmosphere – derived from sources such as fine soil particles, pollen, and fossil fuels – represents another import of P to watersheds. Wet and dry P deposition rates (0.3 kg P/ha/year total) were used from a study across Iowa [*Anderson and Downing*, 2006] and assumed to be constant through time and uniform across the entire watershed area.

Agricultural P Exports

Net Crop Export

Identical methods as described in the Net Livestock Feed Demand section were used to quantify livestock feed P demand and crop P production with net crop P export defined as the difference between the two. A positive net crop P export indicates that crop P export is greater than net livestock feed P demand.

Livestock Products

Livestock products are exported from both watersheds in the form of milk and meat and can represent a substantial flux of P. Animal inventories were previously described in the Net Livestock Feed section. Annual carcass weight equivalents per breeding female were used from a national herd level analysis by Peters et al. [2014] to calculate animal mass exports for beef, dairy beef, and swine. Milk production was estimated by multiplying the lactating cow population by the county level milk yield estimates [*USDA-NASS*, 2020a]. The P content for milk was determined from USDA-ARS [2020] and for beef cattle, dairy cattle, and swine from Antikainen et al. [2005].

Manure Export

Some portion of manure produced at livestock operations near the boundaries of the watersheds will be spread on land outside of the watershed due to manure hauling distances. This process is accounted for in the manure and fertilizer application model described in the Agricultural Fertilizer section. Any manure P that is applied outside the watersheds is treated as a unique export term.

Digester Export

Beginning in 2012 with the construction of a facility in Waunakee, manure digesters have provided an opportunity for manure P to be concentrated in a form that can be exported from the watershed as part of the digestate stream. Research continues to aid in the strategic development of manure collection systems in the LMW [*Sharara et al.*, 2018; *Sharara et al.*, 2017]. Partial data on P exports from two digesters in the LMW was obtained and applied to the year 2017. Clean Fuel Partners who owns and operates the Waunakee digester estimated their annual P export to be 76,000 lb coming from the manure of 2,250 animal units. We then applied that P export to animal unit ratio to the Middleton digester to calculate P export based on an estimated 3,780 animal units that feed into that system.

Non-Agricultural P Exports

Stream Export

Intensive daily P load monitoring has taken place on streams in the LMW and YRW since the 1970s and has captured a lack of any trend in P loading to the Yahara Lakes since then [*Gillon et al.*, 2016]. Additional monitoring stations have come on-line since 2012 and now provide a comprehensive assessment of 88% of the direct drainage area to Lake Mendota. We use regression methods to fill in data gaps to construct a 1990-2019 annual P load dataset for all of the gages in the monitoring network (Figure 3). For the 12% that of the LMW that is ungauged, the proportion of that area was determined that is in urban/developed use and agriculture/other uses. P yields for representative watersheds (average of Yahara River at Windsor, Dorn Creek, and Sixmile Creek for agriculture/other category and Spring Harbor for urban category) were then applied to those area proportions to get an estimate of the ungauged area's P load contribution. The Yahara River at Fulton gage (with data gaps filled in using regression with the Yahara River at Windsor gage) was used to determine stream P export from the YRW. To account for the fact that the MMSD WWTP effluent P load was much higher in the 1990s (prior to the regression time period), the excess effluent P load (above the 2014-2019 average) was added to the original estimated P load at the Fulton gage based solely on the regression model. This method assumes that no P uptake occurred within the stream network from the Badfish Creek effluent discharge location to the Yahara River at Fulton gage. Five year moving averages were used to obtain the estimates from 1992 to 2017 in 5-year increments.

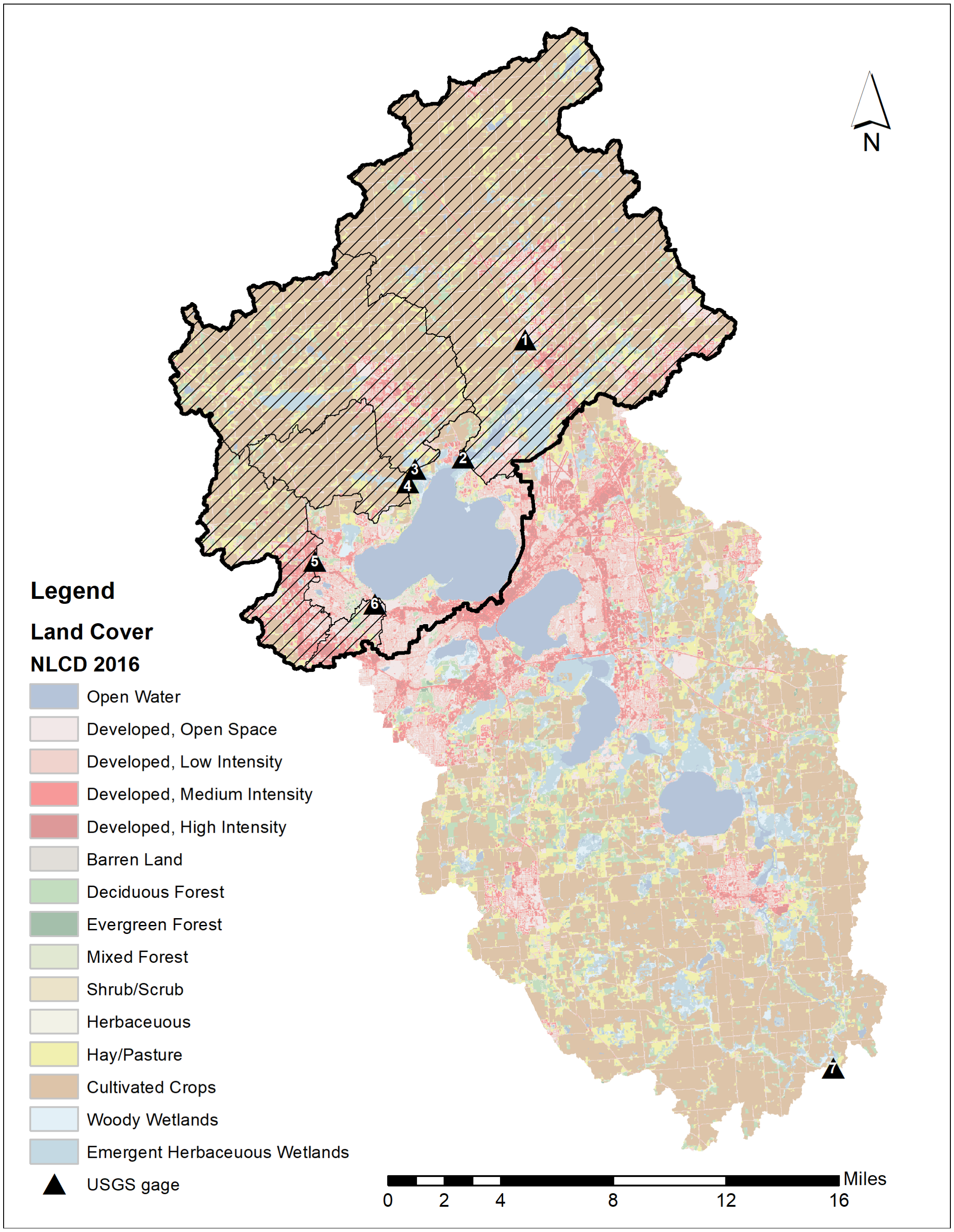

Figure 3. Yahara River Watershed with inset Lake Mendota Watershed (thick black line), all USGS P load gages with associated watersheds (hatched areas), and 2016 land cover. USGS gages are 1) Yahara River at Windsor, 2) Yahara River at Hwy 113, 3) Sixmile Creek at Hwy M, 4) Dorn Creek at Hwy M, 5) Pheasant Branch at Middleton, 6) Spring Harbor Storm Sewer, and 7) Yahara River at Fulton.

Results

Agricultural P Imports

Net livestock feed P demand to the LMW was positive for all years and indicates that feed P was imported to satisfy livestock demand even if all crops produced within the watershed were fed to livestock. The demand declines from 1992 to 2002 as crop production increases and feed P concentration for lactating cows declines but steadily increases from 2002 to 2017 as the livestock population expands and milk yield increases are not outweighed by increases in crop yields (Figure 4). The net feed P demand was negative for all years in the YRW and is presented as an export in the Net Crop Export section below.

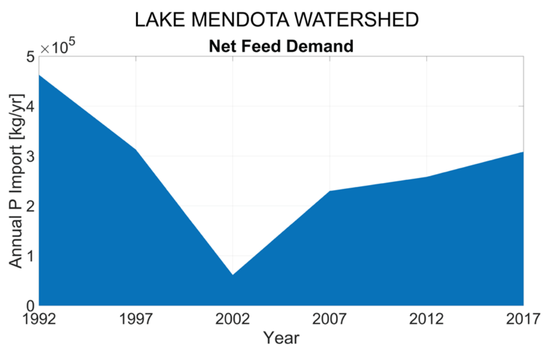

Figure 4. Net livestock feed demand for the LMW from 1992 to 2017

Agricultural fertilizer P imports represent a large import of P to both watersheds but relatively larger for the YRW because of the lower proportion of manured cropland. In both watersheds, agricultural fertilizer P imports were nearly halved from 1992 to 2002 (Figure 5) as part of a larger state- and national-level declining trend in P fertilizer use [*Falcone*, 2021] that was likely also driven by nonpoint water quality awareness and education. However, national trends have flattened (or increased in some areas) since 2002. In both the LMW and YRW, agricultural fertilizer P imports held nearly constant from 2002 to 2017.

To gain confidence in these estimates of fertilizer P imports, average fertilizer P application rates were calculated using the nutrient application model and compared with those determined by using county-scale fertilizer use data [*Falcone*, 2021] and Census of Agriculture data [*USDA-NASS*, 2020b] on the non-manured portion of cropland. The average fertilizer P application rate from the model is 18 kg-P/ha/year and the county-scale estimate is 14 kg-P/ha/year.

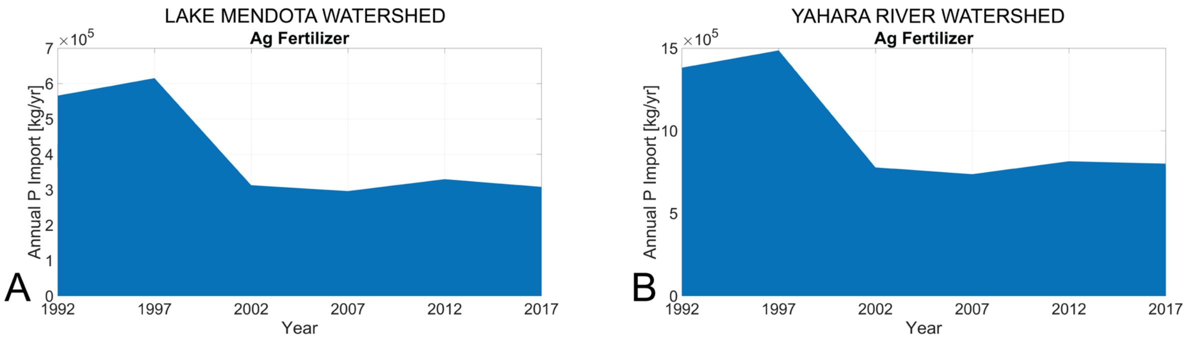

Figure 5. Agricultural fertilizer P imports to the LMW (A) and YRW (B) from 1992 to 2017

As glyphosate use became commonplace in the production of corn and soy in the 1990s and 2000s, pesticide P imports rose dramatically in both watersheds, peaking in 2012 and declining somewhat in 2017 (Figure 6). Again, pesticide P imports are larger in the YRW due to the larger cropland area. While more than an order of magnitude less than agricultural fertilizer P imports, pesticide P imports still represent an important flux of P to the watershed that could undermine other changes elsewhere in the P balance.

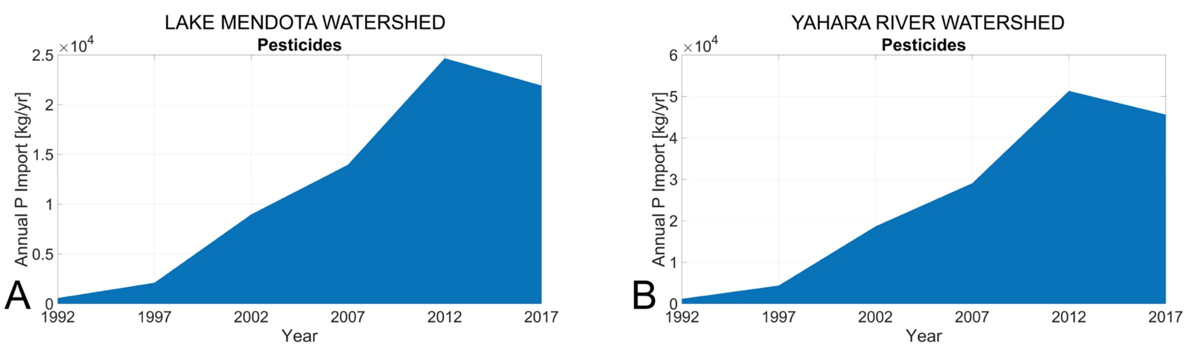

Figure 6. Agricultural pesticide P imports to the LMW (A) and YRW (B) from 1992 to 2017

Non-Agricultural P Imports

While non-agricultural P imports represent an order of magnitude less than agricultural ones, they still represent an important flux and opportunity for integration into P management activities. Urban fertilizer P imports for both the LMW and YRW showed a general decline over the 1992-2017 time period with most of that occurring in the earlier portion (Figure 7). More urban fertilizer P was imported to the YRW compared to the LMW due to more developed area. The broader declining trend is similar to the agricultural fertilizer P imports, which is partially an artifact of the original data source from Falcone et al. [2021] that is based on total county-level fertilizer use that is then apportioned into farm and non-farm uses based on land cover and human population. This represents a limitation in the method because it does not account for changes in use that are more specific to non-farm uses such as the Dane County urban fertilizer P ban (Dane County Code of Ordinances Chapter 80).

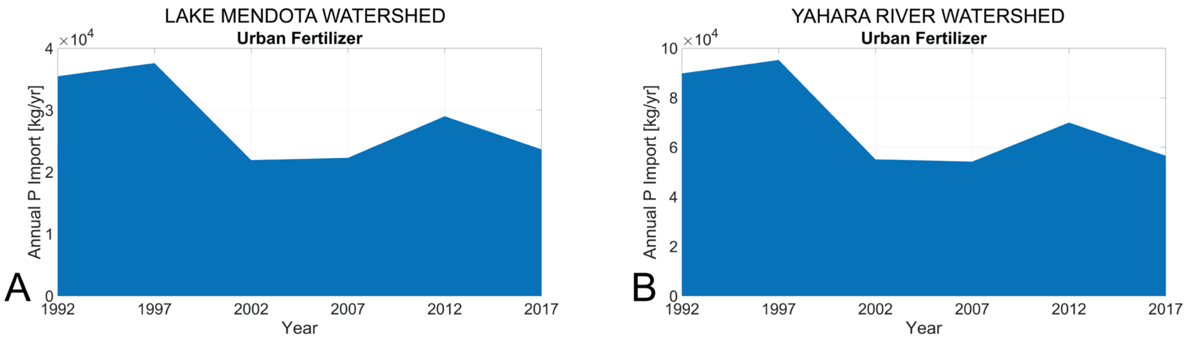

Figure 7. Urban fertilizer P imports to the LMW (A) and YRW (B) from 1992 to 2017

Food and household P demand, which is estimated based on waste discharge to the MMSD WWTP, represents the second largest P import to the YRW after agricultural fertilizer. This flux increases steadily from 1992 to 2007 following increases in watershed population and then declines between 2007 and 2012 primarily due to a reduction in household detergent P concentration [*Keiser*, 2020]. The trend then continues to increase along with population from 2012 to 2017.

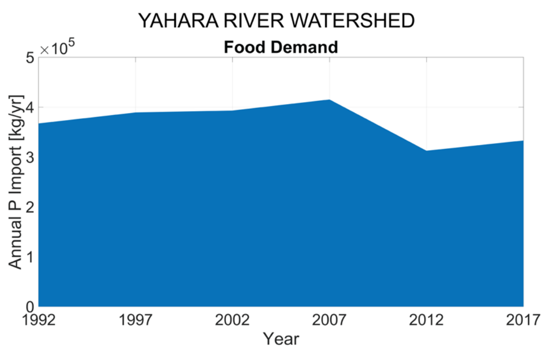

Figure 8. Food/household P demand to the YRW from 1992 to 2017

Biosolids P imports to the LMW from the MMSD WWTP were similar in magnitude to urban fertilizer P imports. However, the trend rises and falls over the time period (Figure 9) and is driven primarily by factors related to the operation of the WWTP. The 78% increase from 1992 to 2002 was driven by enhanced P treatment (biological P removal implemented in 1997) that sent more P to biosolids and less to effluent discharge. The 63% decrease from 2002 to 2017 was primarily driven by a reduction in influent P (mostly from regulations on detergents), the implementation of struvite harvesting in 2014 that reduced biosolids P mass, and a reduction in the proportion of total biosolids produced that is applied in the LMW from 2011 to 2016.

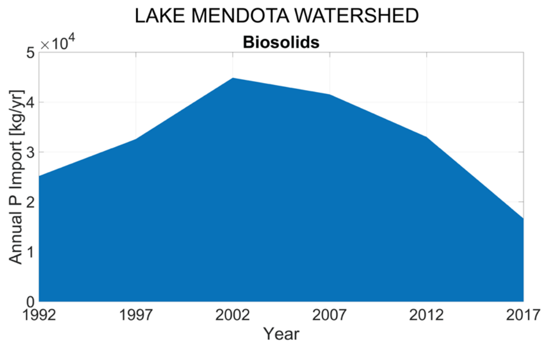

Figure 9. Biosolids P imports to the LMW from 1992 to 2017

Pet feed P demand also represents an urban flux of P that is similar in magnitude to urban fertilizer P and biosolids P. The trend over the 1992-2017 time period shows a 40% and 34% increase in P flux for the LMW and YRW, respectively, that is driven exclusively by increases in human population (Figure 10). A larger human population in the YRW therefore leads to the higher values of pet feed P demand.

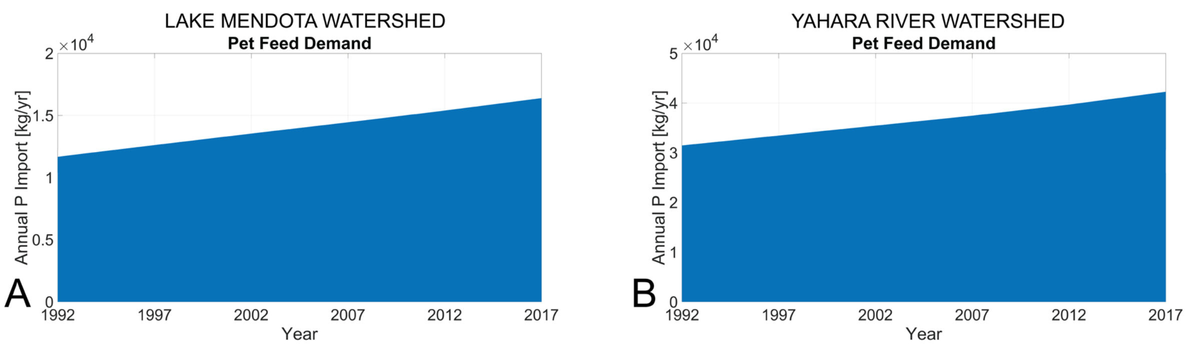

Figure 10. Pet feed P demand to the LMW (A) and YRW (B) from 1992 to 2017

Atmospheric P deposition is another flux that is similar in magnitude to the other non-agricultural P imports. However, no trend was captured due to the lack of data available and the assumption of a constant deposition rate in both watersheds (Figure 11). Therefore, the difference between values for the two watersheds is solely driven by the larger area of the YRW.

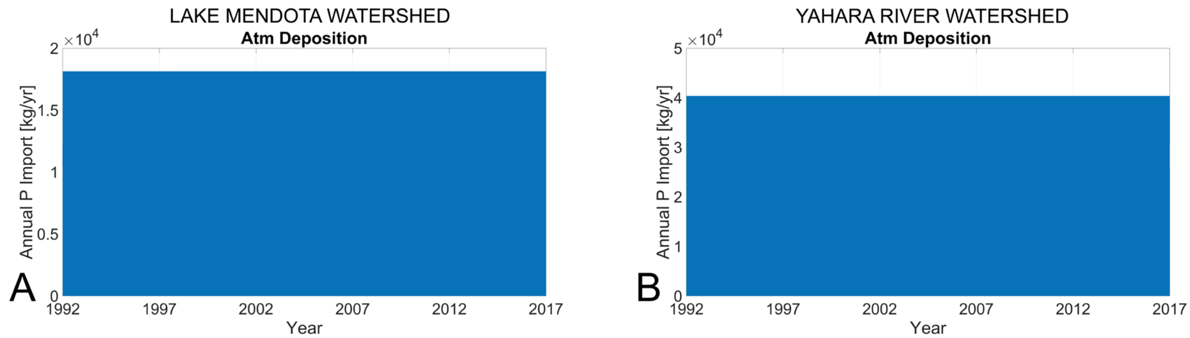

Figure 11. Atmospheric P deposition on the LMW (A) and YRW (B) from 1992 to 2017

Agricultural P Exports

Net crop P export was positive for all years for the YRW and indicates that the demand from the livestock population was less than the total crop production. Therefore, even if all crops grown in the watershed were fed to the livestock within the watershed, there would still be excess available for export. In addition, the net crop P export increases through time from 1992 to 2002 and is driven by the reduction in feed P concentration for lactating cows and the relatively higher growth in crop yields compared to livestock population (Figure 12). Values of net export fluctuate from 2002 to 2017 but show no general trend as growth in the livestock population and milk yield is balanced by increased crop yields.

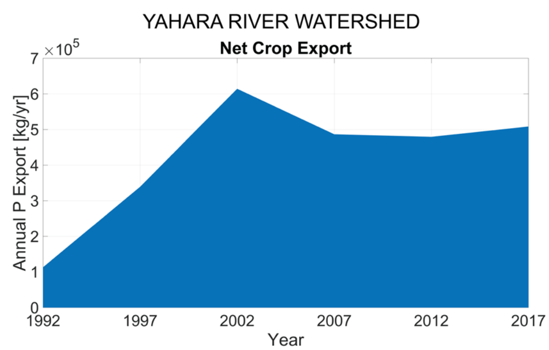

Figure 12. Net crop P export from the YRW from 1992 to 2017

P exports within livestock products are essentially the same for the LMW and YRW as the vast majority of the livestock population in the YRW is located in the LMW (91%). The magnitude of this flux is similar to net crop P export from the YRW during the early time period but approximately half in the later years. The general trend shows an increase in livestock products P from 2002 to 2017 (70% for both watersheds) that is driven primarily by increases in milk production (Figure 13). Relatedly, milk represents a large and growing proportion of the total P in livestock products from 70% in 1992 to 85% in 2017.

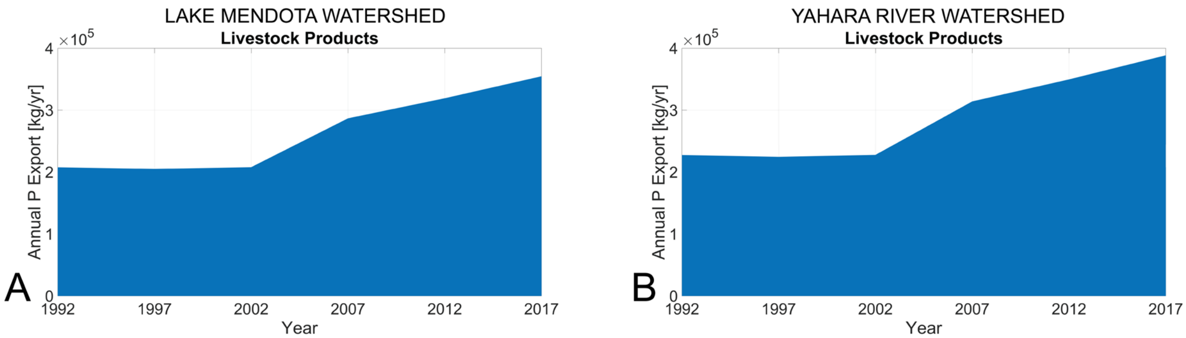

Figure 13. Livestock products P export from the LMW (A) and YRW (B) from 1992 to 2017

Manure P export through the spreading of manure from operations close to watershed boundaries is identical in both watersheds as it only occurs in the LMW. This term represents an outflux that is an order of magnitude lower than net crop export for the YRW or livestock products. The general trend is similar to that of livestock products (increase from 2002 to 2017) and is driven primarily by changes in livestock populations and milk yield in the LMW (Figure 14).

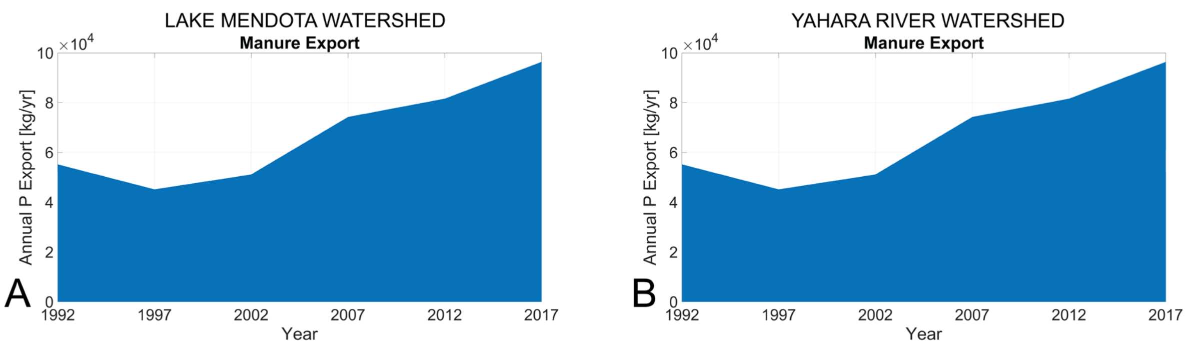

Figure 14. Manure P export from the LMW (A) and YRW (B) from 1992 to 2017

Manure digester P export represents a more recent flux that only appears in 2017 as digesters were constructed starting in 2014 and nutrient concentration and export was implemented thereafter (Figure 15). This new flux is similar in magnitude to manure P that is exported by spreading over watershed boundaries in 2012 and 2017.

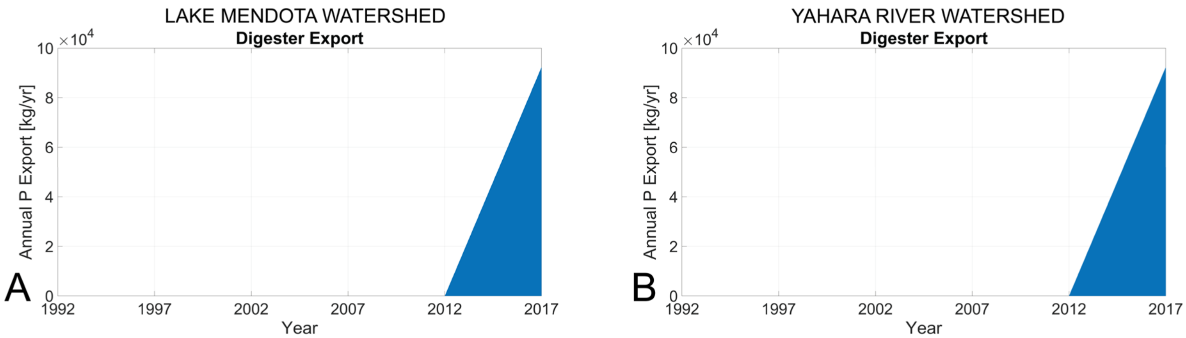

Figure 15. Manure digester P export from the LMW (A) and YRW (B) from 1992 to 2017

Non-Agricultural P Exports

While stream P export from the LMW is the lowest of all P outflux terms, it represents an important flux within the overall watershed P balance as it has a direct connection to surface water quality. Consistent with previous research [*Gillon et al.*, 2016; *Lathrop and Carpenter*, 2011], estimates of stream P export to Lake Mendota show no substantial trend over the 1992 to 2017 time period (Figure 16A). In contrast, stream P export from the YRW was larger than manure export and similar to livestock products in the early time period as effluent concentrations from the MMSD WWTP were substantially higher than in the later period. The sharp 76% decline in stream P export from the YRW from 1992 to 2002 (Figure 16B) is driven primarily by changes in wastewater treatment (biological P removal) that directed more P to biosolids and substantially less to effluent discharge. No trend was present in YRW stream P export from 2002 to 2017 indicating the overall lack of trend in upstream sources and the attenuating effect of the Yahara lakes [*Lathrop and Carpenter*, 2011]. In 1992, effluent P discharge to Badfish Creek (tributary to Yahara upstream of Fulton gage) represented 75% of the total P load at the Yahara River outlet. By 2017, this proportion declined to 21% of the total.

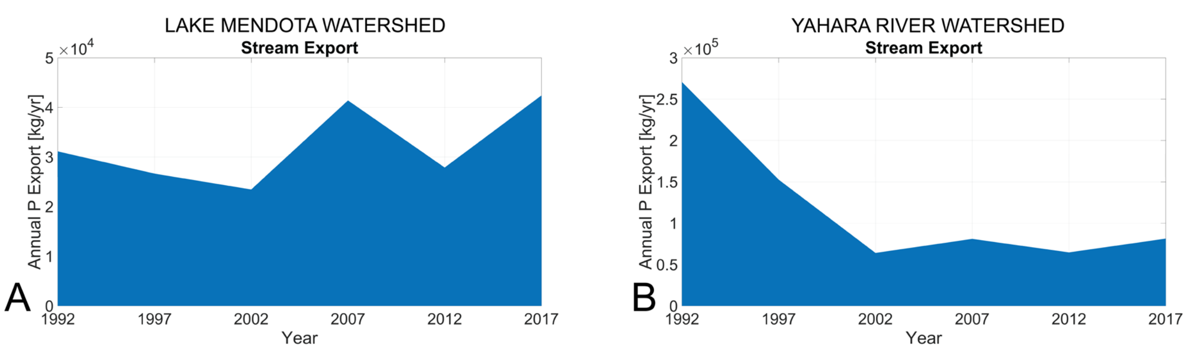

Figure 16. Stream P export from the LMW (A) and YRW (B) from 1992 to 2017

Mass Balances

After accounting for all changes in P imports and exports, the LMW P accumulation rate is positive for all years from 1992 to 2017 indicating that P storage continues to increase (Figure 17). However, this accumulation rate has declined substantially over the study period (88% from 1992 to 2017). P imports to the LMW are dominated by net livestock feed demand and agricultural fertilizer (accounting for 78 to 92%) but are still influenced by pesticides, urban fertilizer, biosolids, and pet feed demand (all roughly equal in 2017). P exports from the LMW are dominated by livestock products (accounting for 60 to 75%) but also strongly influenced by manure and digester P export (especially by 2017). The accumulation rate decline is most strongly driven by declines in agricultural fertilizer P (mostly from 1992 to 2002) with increases in livestock products being a secondary driver. However, digesters coming on-line by 2017 led to a noticeable decline in the overall accumulation rate from 2012 to 2017.

For the YRW, a similar trend is present in the P accumulation rate but eventually goes very close to zero by 2017 (Figure 18). This declining trend is also primarily driven by reductions in fertilizer P imports in the early period but also strongly influenced by increases in net crop export, livestock products, and stream export over the full time period, the addition of manure digester export in 2017, and the overall decline in human food and household P demand/waste (that also represents the second largest P import to the YRW). Net crop export became a larger outflux than livestock products through the 1992 to 2017 time period as crop yield increases (on a larger cropland base than in the LMW) and P feed concentrations for lactating cows outweighed increases in livestock populations and their associated increases in overall feed demand and product export.

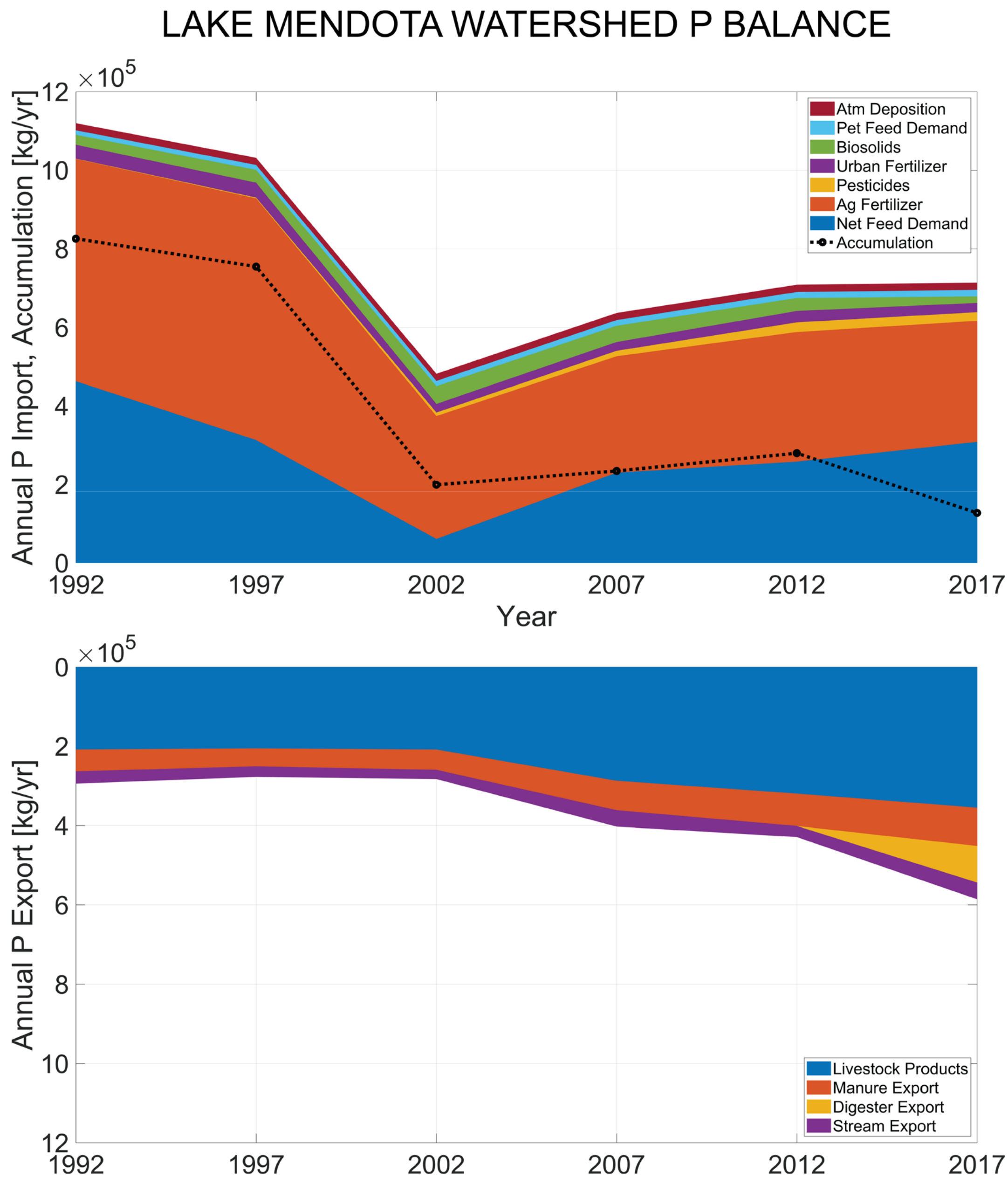

Figure 17. Lake Mendota Watershed P mass balance for 1992 to 2017 with net P accumulation (block dotted line)

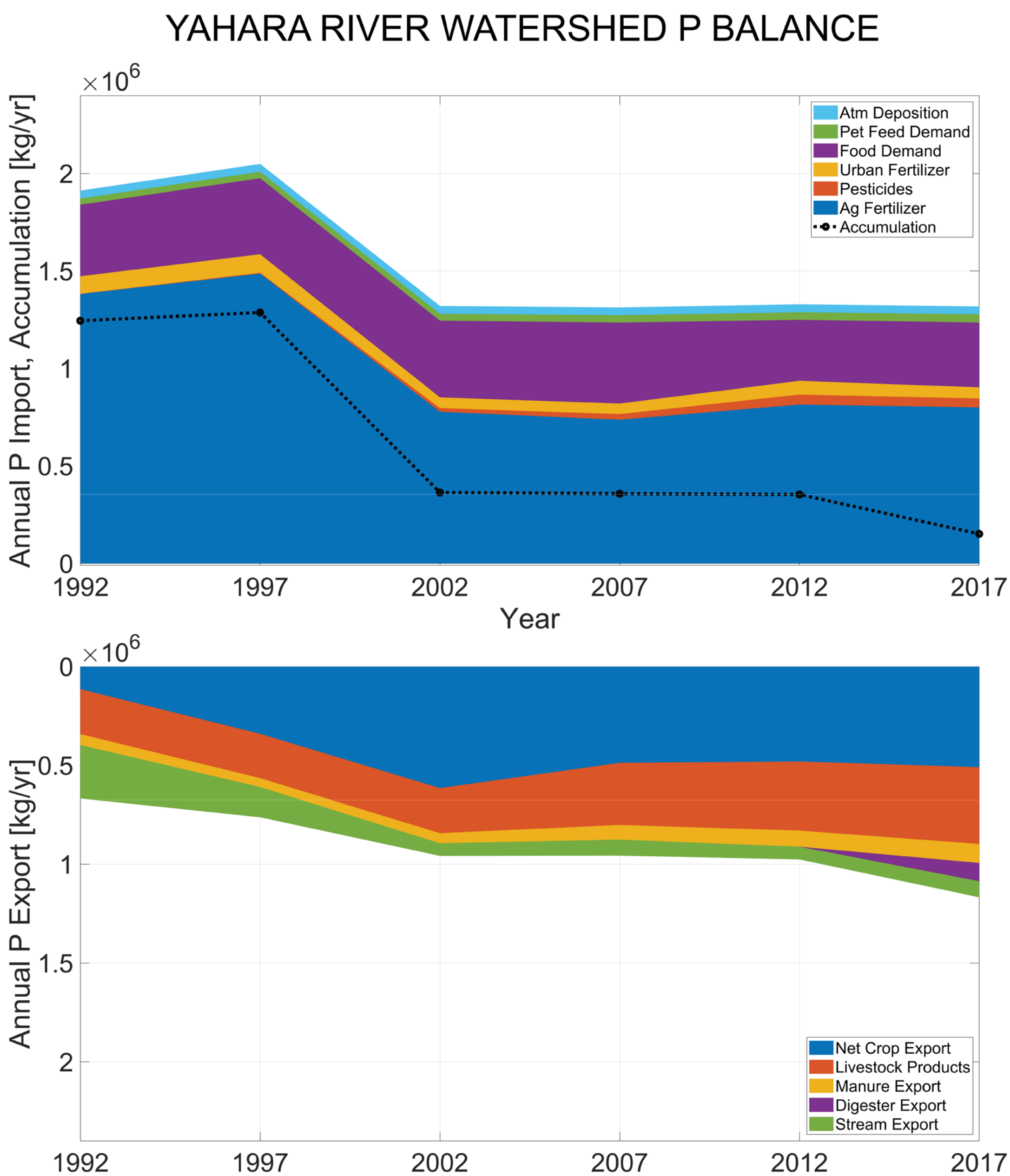

Figure 18. Yahara River Watershed P mass balance for 1992 to 2017 with net P accumulation (block dotted line)

Discussion

Both the LMW and YRW P mass balances through the 1992 to 2017 time period indicate substantial progress on the reduction of P accumulation in both watersheds that is known to influence long-term water quality outcomes [*Motew et al.*, 2017]. While the overall P accumulation rate is estimated to be very slightly negative for the YRW, the rate for the LMW is still positive indicating that the legacy effect is still building. While assessing what stores within the LMW landscape are accumulating this excess P is beyond the scope of this study, one can infer that soil P concentration is still increasing on some agricultural land in the watershed even though some evidence suggests the opposite is happening on intensively managed cropland where land managers are involved in water quality improvement efforts [pers. comm., Kyle Minks, Dane County]. However, the sharp decline and low accumulation rate in 2017 indicates that the land area that is accumulating P is substantially smaller.

For the YRW, the drivers of the mass balance outcomes are more difficult to disentangle as a consequence of the addition of an urban/human influence that is not as present in the LMW. Human and household P demand and waste was accounted for in the YRW because the flows of P (from consumption to wastewater effluent discharge) are all occurring within the watershed boundary. This is not the case for the LMW. However, the primary store of P on the landscape is still agricultural soils and the mass balance changes suggest that land with increasing soil P levels is largely balanced between land with decreasing soil P. It is worth noting that landfill P fluxes and stores was not considered in this analysis because they are treated as largely unavailable for transport to surface water bodies. But these fluxes are likely of similar – but less – magnitude to food/household demand (representing food thrown in the trash) and could be important opportunities to rebalance nutrients to agricultural lands and replace fertilizer imports.

The overall accumulation rate for the LMW in 1997 is 31% higher than that estimated by Bennett et al. [1999] while the rate estimated in this analysis for 2007 is 16% lower than that estimated by Kara et al. [2012]. While these numbers do differ slightly, their relatively similar magnitudes provide additional confidence to the overall analysis. The power of the current study is to analyze the changes in this accumulation through time and allow for future tracking using an established and repeatable methodology.

This analysis also provides a useful comparison of external and internal fluxes of P that may be useful for management activities and outreach. For instance, human and household waste P from the YRW is 32 to 53% of that from livestock manure P production (calculated by subtracting livestock products P from livestock feed demand; Figure A1). This relatively large internal P flux from direct human activity is largely ignored during conversations of watershed water quality because it is sent and treated downstream of the lakes. However, this comparison of similar relative magnitudes could help contextualize the larger challenge of P security and risk within the food and agricultural system. For instance, there are potential opportunities that can be explored – beyond the commendable history of the MMSD MetroGro program – to rebalance and recycle P through the food system from agricultural lands to households and back again [*Nesme and Withers*, 2016].

The largest P flux in both the LMW and YRW is agricultural fertilizer P imports. While the methodology presented leads to a slightly higher estimate of the average annual fertilizer P application rate on non-manured cropland than the county-scale estimate (18 vs 14 kg-P/ha), their relative similarity is encouraging. In addition, it is reasonable to assume that fertilizer rates are slightly higher in the YRW than for the county as a whole due to lands with slightly higher productivity potential in the YRW [*Lark et al.*, 2020]. However, fertilizer P estimates represent the largest source of uncertainty in the P mass balance of both watersheds and would benefit from future investigations to enhance accuracy and reduce uncertainty.

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Appendix

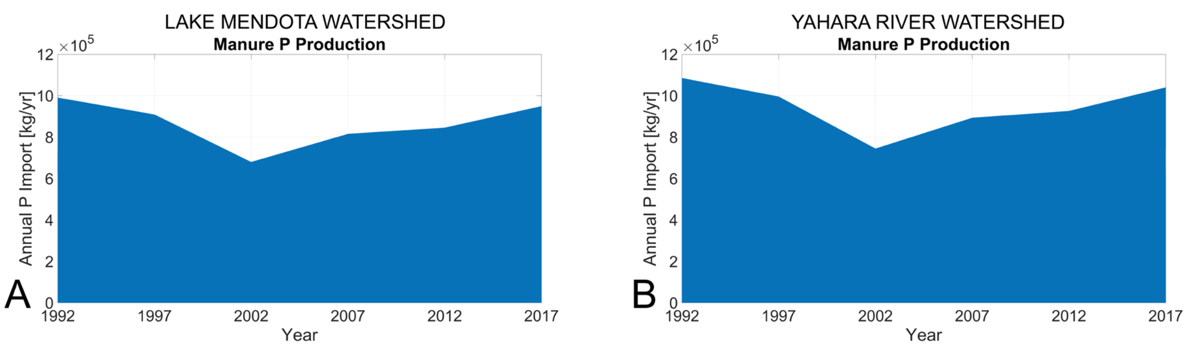

Figure A1. Manure P production in the LMW and YRW from 1992 to 2017 calculated by subtracting Livestock Products P export from Livestock Feed Demand P.

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