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The Influence of Legacy P on Lake Water Quality in a Midwestern Agricultural Watershed

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Abstract

Decades of fertilizer and manure applications have led to a buildup of phosphorus (P) in agricultural soils and sediments, commonly referred to as legacy P. Legacy P can provide a long-term source of P to surface waters where it causes eutrophication. Using a suite of numerical models, we investigated the influence of legacy P on water quality in the Yahara Watershed of southern Wisconsin, USA. The suite included Agro-IBIS, a terrestrial ecosystem model; THMB, a hydrologic and nutrient routing model; and the Yahara Water Quality Model which estimates water quality indicators in the Yahara chain of lakes. Using five alternative scenarios of antecedent P storage (legacy P) in soils and channels under historical climate conditions, we simulated outcomes of P yield from the landscape, lake P loading, and three lake water quality indicators. Legacy P had a significant effect on lake loads and water quality. Across the five scenarios for Lake Mendota, the largest and most upstream lake, average P yield (kg ha^{-1}) varied by −41 to +22%, P load (kg y^{-1}) by −35 to +14%, summer total P (TP) concentration (mg l^{-1}) by −25 to +12%, Secchi depth (m) by −7 to +3%, and the probability of hypereutrophy by −67 to +34%, relative to baseline conditions. The minimum storage scenario showed that a 35% reduction in present-day loads to Lake Mendota corresponded with a 25% reduction in summer TP and smaller reductions in the downstream lakes. Water quality was more vulnerable to heavy rainfall events at higher amounts of P storage and less so at lower amounts. Increases in heavy precipitation are expected with climate change; therefore, water quality could be protected by decreasing P reserves.

Key words: legacy phosphorus; eutrophication; nonpoint source pollution; watershed modeling; agricultural runoff; manure.

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This article contains a supporting appendix that is available online at https://github.com/mmotew/Appendix-Motew-et-al-2017.git.

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Introduction

Freshwater eutrophication caused by phosphorus (P) enrichment is a serious environmental problem throughout the world, characterized by low water clarity, potentially toxic algae blooms, oxygen depletion, fish kills, and loss of economic benefits associated with clean water (MA 2005; Smith and others 2006; Bennett and Schipanski 2013). In many watersheds, a history of agricultural land use has resulted in elevated levels of P in surface soils, ditches, riparian zones, wetlands, and stream and lake sediments (Sharpley 2016). Accumulation of P can occur over finite periods when P inputs exceed agricultural demand, and this accumulation can continue to mobilize long after inputs decline (Powers and others 2016). A global P budget performed in 2001 indicated that the increase in net P storage in terrestrial and freshwater ecosystems was at least 75% greater than preindustrial levels, with a large portion occurring in agricultural soils (Bennett and others 2001). Regions with high densities of livestock operations are especially prone to buildup of soil P when manure is applied to meet nitrogen demand and subsequently supplies P in excess of crop removal (Nowak and others 1998; Staver and Brinsfield 2001; Sturgul and Bundy 2004).

Legacy P buildup can act as a long-term source of P to surface waters, in many cases delaying intended reductions in catchment P fluxes associated with best management practices (BMPs) (Meals and others 2010; Hamilton 2012; Sharpley and others 2013). The release of legacy P into surface waters can overwhelm factors favoring water quality improvement, making it difficult to distinguish the effects of current conservation measures from historical land management (Hamilton 2012; Jarvie and others 2013a; Sharpley and others 2013). Drawdown by crops may be the best mitigation measure for legacy P, but can take decades or longer depending on how fertilizer inputs and agricultural exports are managed (McCollum 1991; Schulte and others 2010).

In addition to catchment-scale problems associated with legacy P, there is also widespread recognition of a broken global P cycle that has arguably created the legacy P problem. Finite reserves of geologic phosphate rock are being mined at unsustainable rates while also accumulating in parts of the world where it causes significant degradation to aquatic ecosystems, such as the Midwestern USA (Elser and Bennett 2011). Proposed solutions for managing P resources, as well as legacy P, have focused on closing the global P budget, attempting to connect the surplus of legacy P to regions where P is scarce (Childers and others 2011; Sattari and others 2012; Metson and others 2015). Methods to recover P from human waste, wastewater, manure, agricultural residues, as well as soils are being developed (Elser and Bennett 2011; Kahiluoto and others 2015; Wilfert and others 2015). The recovery of legacy P could help substitute manufactured fertilizers, preserve critical reserves of phosphate rock, as well as improve water quality (Rowe and others 2015; Wolfe and others 2016).

Lake eutrophication is a major concern in the Yahara Watershed (YW) of southern Wisconsin, home to a thriving dairy industry, the city of Madison, and a chain of four highly valued lakes (Stumborg and others 2001). The largest and furthest upstream lake, Lake Mendota, receives the greatest average annual direct P loads of the four and has a large influence on river loading to the downstream lakes that include Monona, Waubesa, and Kegonsa (Lathrop and Carpenter 2013). Eutrophication has plagued the Yahara lakes since the mid-1800s, when the landscape was first transformed from native vegetation to farms and urban areas (Carpenter and others 2006). Until the mid-twentieth century, sewage effluents were the greatest source of P to the lakes, but since wastewater diversion in 1971, nonpoint sources have dominated, most importantly agricultural runoff (Soranno and others 1996; Lathrop and others 1998; Lathrop 2007). As in other agricultural watersheds, there is strong evidence that inputs of P to the YW exceed outputs and that soil P levels are significantly greater than needed to meet plant demand and sustain crop yields (Bennett and others 1999; Reed-Andersen and others 2000; Kara and others 2011). According to a P budget conducted in the Lake Mendota watershed in the late 1990s, only 4.6% of annual P outputs from the watershed were exported to Lake Mendota. The majority of P outputs mostly consisted of exported corn and dairy products (Bennett and others 1999). The authors estimated drawdown of soil P to 1974 levels by crops could take decades to centuries, depending on how inputs and outputs are managed (Bennett and others 1999). An updated P budget showed that inputs to the Lake Mendota watershed have likely declined since the mid-1990s, but have continued to exceed outputs (Kara and others 2011).

Loads to Lake Mendota have not changed over the past three decades despite significant nutrient reduction interventions in that subwatershed (Genskow and Rumery Betz 2012; Lathrop and Carpenter 2013). Loads to Mendota vary year to year due to weather, but average loading from direct drainage sources has fluctuated around 29,600 kg y^{-1} over the 1976–2008 period, and the median summertime lake TP concentration has exceeded the mesotrophic threshold of 0.024 mg l^{-1} in most years (Lathrop and Carpenter 2013). The failure of management interventions to improve water quality has been blamed on the counteractive influence of other factors affecting water quality over this time period. These include an increase in annual precipitation and increased frequency of extreme rainfall, continued intensification of dairy production, and a persistent legacy P problem (Gillon and others 2016; Rissman and Carpenter 2015). It can be argued that without management interventions, water quality would have declined over this period.

Prior studies have identified the important role that legacy P plays in affecting water quality and the long-term challenges it poses for water resource management (Haygarth and others 2014; Powers and others 2016). However, establishing causal links between factors affecting water quality, like legacy P, and their outcomes, is challenging, especially at the catchment scale. This is because many factors may simultaneously affect water quality and/or its measurement and are subject to change over time (Edwards and Withers 1998; Hamilton 2012; Gillon and others 2016; Rissman and Carpenter 2015). In this study, we used models to bypass this challenge. We quantified the influence of legacy P on water quality in isolation from confounding factors such as land use, land management, and climate. Specifically, we examined the sensitivity of water quality outcomes to legacy P within the YW using alternative scenarios of P storage in watershed soils and stream sediments. We asked how much does legacy P influence lake water quality, and what would present water quality be like if past levels of stored P had been different? The study was conducted using a newly developed numerical modeling framework to simulate the scenarios and generate outcomes of water quality.

Methods

Study Area

The YW is located mainly within Dane County, Wisconsin, with small portions in Rock and Columbia counties. The 1344 km^2 watershed contains a chain of four eutrophic lakes that drain from north to south in the following order: Mendota, Monona, Waubesa, and Kegonsa. Roughly half the landscape in the YW is devoted to agriculture, with corn, soy, and dairy the principal products. The northern and southern parts of the watershed are predominantly agricultural, with dairy operations more common in the north and cash grain in the south. The Wisconsin state capital city of Madison (43°6′N, 89°24′W) and surrounding urban area comprises roughly one-quarter of the watershed and is centered on an isthmus between Lakes Mendota and Monona. The remaining quarter of the watershed is covered in natural vegetation, including forests, wetlands, and prairie.

Models

We developed a watershed modeling framework that can simulate an array of ecosystem services, including land-to-lake flows of water, sediment, and nutrients, and surface water quality (Figure 1). The framework includes process-based representation of natural and managed ecosystems (Agro-IBIS), hydrologic routing of water, sediment, and nutrients through the surface hydrologic network (THMB), and prediction of Secchi disk depth (a measure of lake transparency), summertime lake total phosphorus (TP) concentration, and the probability of hypereutrophy in each lake (Yahara Water Quality Model).

The Agro-IBIS model is a spatially explicit land surface model that simulates the movement of water, energy, momentum, carbon, nitrogen, and now phosphorus, in both natural and managed ecosystems. The structure of Agro-IBIS has been described in detail (Foley and others 1996; Kucharik and others 2000; Kucharik 2003), and many components of the model have been validated across a range of ecosystems at various spatial and temporal scales (El Maayar and others 2001; Kucharik and Brye 2003; Kucharik and others 2006; Kucharik and Twine 2007; Soylu and others 2014; Zipper and others 2015). Recently, Agro-IBIS was integrated with the variably saturated soil water flow model HYDRUS-1D to enable simulation of groundwater–vegetation interactions (Soylu and others 2014). For this study, several additional updates have been made to Agro-IBIS. First, biogeochemical cycling of P and loss of P to runoff were added to Agro-IBIS to enable simulation of P dynamics, including interactions between surface water quality, climate, and land management. The new terrestrial P module is largely based on SurPhos, a state-of-the-art dissolved P loss model for agricultural systems receiving manure (Vadas and others 2004, 2005, 2007; Vadas and White 2010). The new P module features P application, transformation, and loss of dissolved

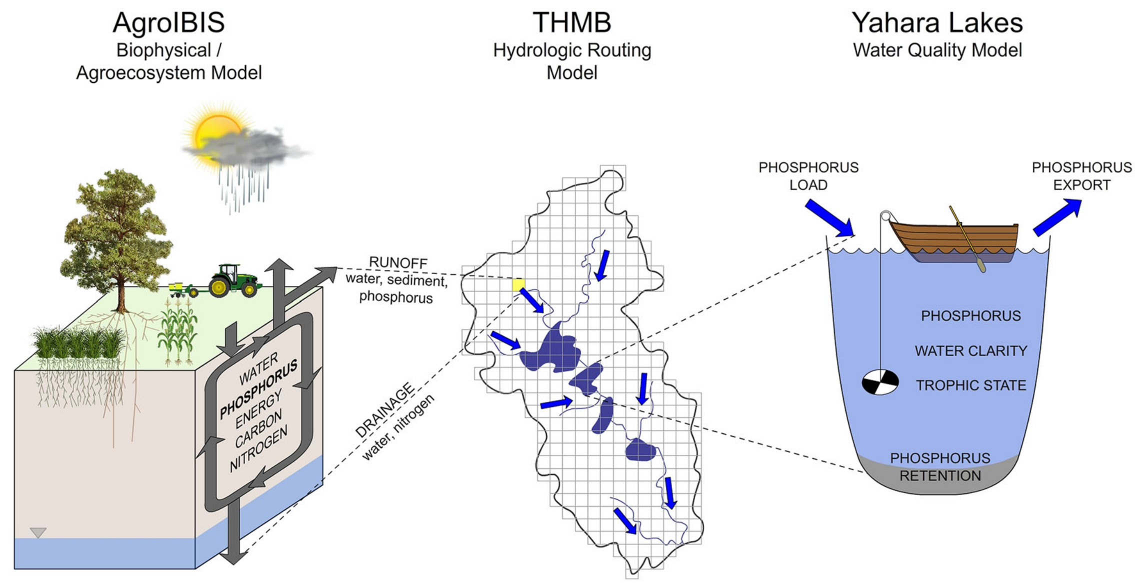

Figure 1. Suite of three models used in this study.

P to runoff; in-soil cycling of organic and inorganic forms of P; and loss of particulate-bound P with erosion (Figure 2). Land cover categories have also been expanded in Agro-IBIS to include four classes of urban areas (high, medium, and low intensity as well as open space) along with alfalfa, pasture, and wetlands. Finally, changes were made within the soil physics routines to improve representation of surface hydrology and improve model stability. Details for all changes made to Agro-IBIS are provided in Appendix A in Electronic supplementary material (ESM).

Agro-IBIS was linked to the terrestrial hydrology model with biogeochemistry (THMB) to enable simulation of water, sediment, and P transport at the watershed scale, including delivery of both dissolved and particulate P loads to the YW lakes and the Rock River. The THMB model, formerly called HYDRA, is a physically based hydrologic routing model. By linking the topographic data of a prescribed stream network and river morphological characteristics to a set of linear reservoir functions, the model simulates the temporal variability of water flow and storage in hydrologic systems (Coe 1998, 2000; Coe and others 2008). Donner and others (2002) added a nitrogen transport module to the THMB modeling framework. In this study, transport and cycling of sediment and P have been added to THMB. Using spatially explicit outputs of surface runoff, subsurface drainage, sediment yield and P yield generated for each grid cell by Agro-IBIS, THMB transports water, sediment, and P across the 2-D landscape. THMB calculates direct drainage and total loading of P and sediment to the lakes, as well as the movement of these quantities through the lake chain (Figure 1). Further details for the new THMB capabilities are provided in Appendix B in ESM.

The Yahara Water Quality (WQ) Model predicts summer water quality variables in the four main-stem Yahara lakes given annual direct drainage loads to each lake from THMB. The model computes a mass balance for each lake given empirical relationships (Carpenter and Lathrop 2014). Total annual loads to Lake Mendota are calculated using direct drainage loads only since there are no P inputs from other lakes. For the downstream lakes, total annual loads are determined using direct drainage loads from THMB as well as estimates of P export from upstream lakes. Summer water quality is computed from terms of the P mass balance using regressions (Lathrop and Carpenter 2013; Carpenter and Lathrop 2014). Further details of the Yahara WQ Model are presented in Appendix C in ESM.

For water quality purposes, summer is defined as 30 June to 7 September, which is reliably a period of summer lake stratification. Water quality variables are TP (total P concentration in surface water), DRP (dissolved reactive phosphorus concentration in surface water), chlorophyll (chlorophyll a concentration in surface water), and Secchi disk transparency. Lake trophic state and water quality are multivariate phenomena, and multiple indicators are commonly used to obtain different perspectives (Carlson 1977). We defined

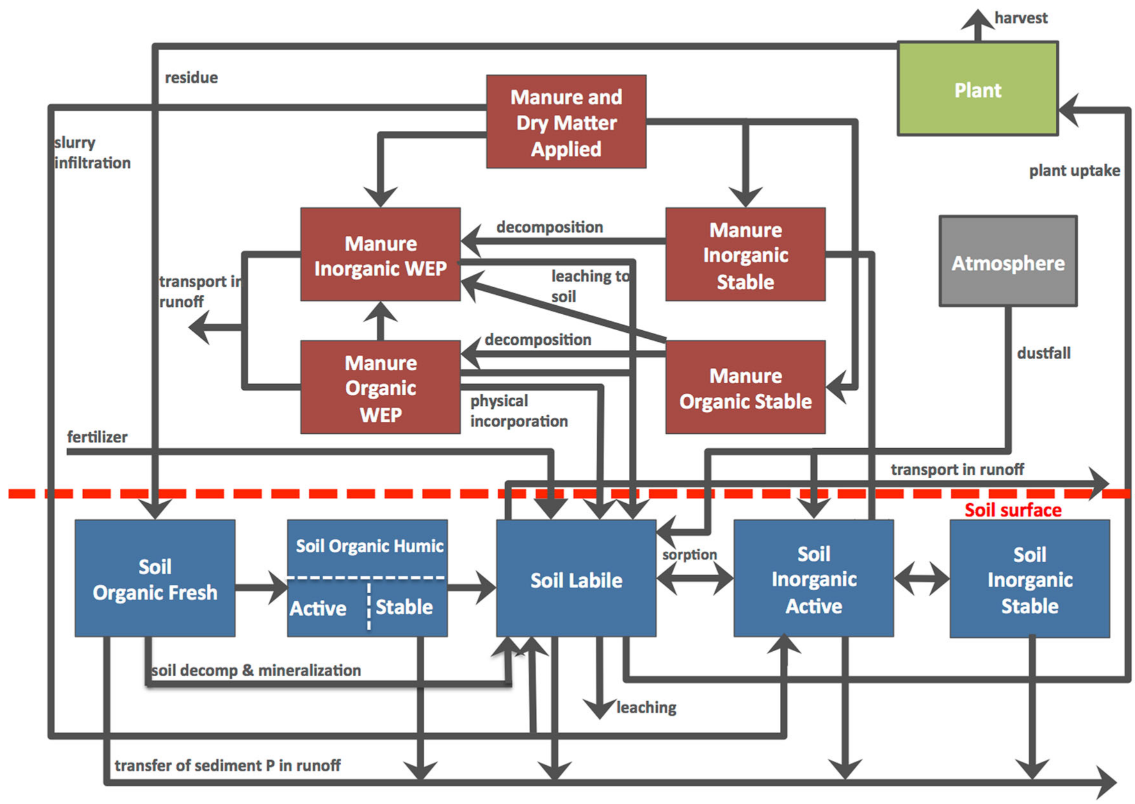

Figure 2. Terrestrial P module in Agro-IBIS, largely based on the SurPhos model of Vadas and others (2007).

eutrophic (vs. mesotrophic conditions) by TP above 0.024 mg l^{-1} or Secchi transparency less than 2 m (Carlson 1977). We define hypereutrophy by the presence of DRP at concentrations greater than 0.005 mg l^{-1}, indicating that phytoplankton are not removing all detectable DRP from the water. Empirical relationships in the Yahara WQ Model were computed using 33 full years of data for lakes Mendota and Monona (1975–2008) and 28 full years of data for lakes Waubesa and Kegonsa (1980–2008). Datasets and regression models are described by Lathrop and Carpenter (2013) and Carpenter and Lathrop (2014). Data are archived by the North Temperate Lakes Long-Term Ecological Research program (http://lter.limnology.wisc.edu).

Experimental Design

Prior to all simulations, a spin-up of C and N pools in Agro-IBIS was first performed. The dependency of P cycling on water, C, and N cycles necessitated the need for a spin-up since this is how C and N pools are brought to equilibrium within soils and vegetation (Kucharik and others 2000). Because the model was initialized without any C and N in the soil pools, the amount of time to bring the pools to equilibrium was roughly 200 years, making the spin-up period span 1786–1985. Calibration and validation results for Agro-IBIS, THMB, and the Yahara WQ Model are presented in Appendix D in ESM. Results of parameter sensitivity analyses conducted for Agro-IBIS and THMB are presented in Appendix J in ESM.

To investigate the effect of legacy P on surface water quality, we ran five simulations over the recent historical period (1986–2013) in which the initial amount of P stored in soils and river channels was varied (Figure 3). The simulations were named according to their ranking in initial P content: low (LO), medium–low (ML), medium (ME), medium–high (MH), and high (HI). Using output from the Yahara WQ Model, we analyzed three water quality indicators for each simulation including summer lake TP concentration, Secchi depth, and the probability of hypereutrophy. Model inputs of climate, soil, groundwater, LULC, and nutrients are described in Appendices E–I in ESM.

The simulations were designed to span a range of plausible values for P storage in surface soils and streams based on available data and past studies. The third simulation, ME, was determined through

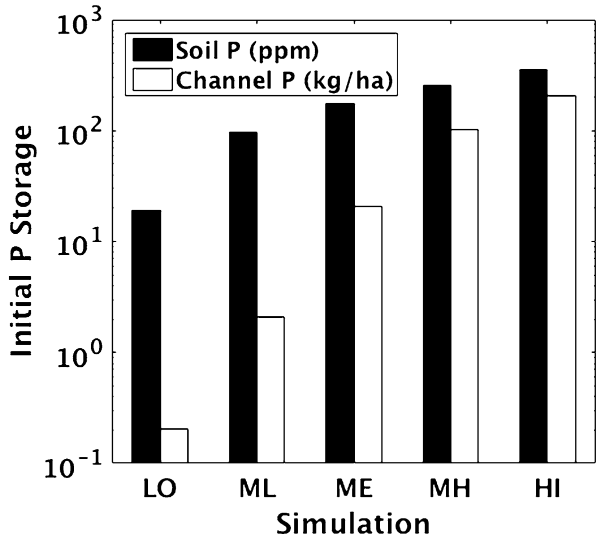

Figure 3. Surface soil P concentration (ppm) and in-channel P storage (kg ha^{-1}) used to initialize Agro-IBIS and THMB in the five simulations. Simulation names correspond to their ranking of initial P: low (LO), medium–low (ML), medium (ME), medium–high (MH), and high (HI).

the calibration process and was used to represent the best estimates of historical soil P, P yield, and P loading to the lakes. The LO and ML simulations were initialized with less soil and channel P storage than ME, and likewise the MH and HI simulations were initialized with more soil and channel P storage. All Agro-IBIS simulations were preceded by a 25-year soil P spin-up (1961–1985) to allow soil P pools in croplands to build up gradually to five contrasting levels in 1986. The levels were chosen to span a plausible range of soil test P observed in surface soil layers in croplands (Sharpley and others 1994; Andraski and Bundy 2003; Smith and Warnemuende-Pappas 2015). These values were approximately 20, 95, 175, 255, and 350 ppm for LO, ML, ME, MH, and HI, respectively. An upper limit of 350 ppm was chosen because a simulated equilibrium was observed near this value for silt loam soil within the watershed. The spin-ups were conducted by first initializing the labile P pools to 6, 20, 30, 37.5, and 50 ppm, respectively, for the five simulations. Assuming the labile pool is equal to half of Bray-1 soil test P (Vadas and White 2010), these values would correspond to 12, 40, 60, 75, and 100 ppm soil test P, respectively, in the surface layer. Manure was applied annually throughout each spin-up at rates of 15, 35, 50, 62, and 80 kg ha^{-1}, respectively, for the five simulations, in order to build up soil P to the desired levels. The soil P spin-up was conducted only in cropland grid cells, where corn, soy, wheat, or alfalfa was grown. This was done because croplands represent the primary source for nonpoint P pollution in the YW, and also because P interventions are mostly targeted at croplands, where soil P dynamics are heavily influenced by management. Labile P in noncrop grid cells was initialized at 15 ppm (30 ppm soil test P), which is consistent with local observations of soil test P in lawns and remnant prairies (Bennett and others 2004).

Channel P storage, or P stored in stream sediments, is an important contributor to in-stream P concentration (Walling and others 2008), and so channel P storage in THMB was also varied across the five simulations to represent a range of conditions consistent with low storage (0.2 kg ha^{-1}, LO), medium–low storage (2.0 kg ha^{-1}, ML), calibrated storage (21 kg ha^{-1}, ME), medium–high storage (103 kg ha^{-1}, MH) and high storage (207 kg ha^{-1}, HI). Levels of in-stream P storage vary significantly among watersheds, and actual in-stream P storage data are highly limited. Owens and Walling (2002) reported in-stream total P storage at different locations of a rural watershed in the UK, which had similar sediment P concentration levels as the YW (Water Resources Management Practicum 2015). Owens and Walling (2002) presented a range of in-stream P storage levels from 0.4 to 63.2 kg ha^{-1}, which was consistent with the range of storage levels used in our experiment. The higher cases of MH and HI allowed us to also explore P load variability under more extreme conditions of legacy P buildup. Lake P storage was not adjusted in the experiment. Land cover and meteorological input data as well as manure and fertilizer P applications were the same for all five simulations, identical to those used in the calibrated historical run (ME). This was done in order to isolate the effect of legacy P storage on lake loads and water quality indicators.

Analysis

To analyze P transport through the watershed and assess water quality outcomes for each simulation, the following quantities were analyzed: average annual P yield calculated by Agro-IBIS; total annual direct drainage load to each lake calculated by THMB; and summertime lake TP, Secchi depth, and probability of hypereutrophy in each lake calculated by the Yahara WQ Model. Phosphorus yields represented the average amount of P coming off of land-based grid cells within the total drainage area for each lake (Appendix Figure B1 in ESM). These were calculated by summing both dissolved and particulate forms of P in runoff. Loads calculated in THMB represented the annual amount of total P reaching each lake via direct drainage (that is, did not include riverine contributions from upstream lakes since those are determined by the Yahara WQ Model). The loads calculated by THMB were used as inputs to the Yahara WQ Model where calculations of summer TP concentration, Secchi depth, and the probability of hypereutrophy were made for each lake. All annual variables were calculated for the water year, defined as November 1–October 31. For each variable analyzed, we performed a randomized block design ANOVA with year as the random effect and simulation as the fixed effect to test the null hypothesis that the means of each P storage treatment in the five simulations were the same. Standard deviations in error were also calculated according to the randomized block ANOVA and were reported in the results along with means and used as error bars in figures. All analyses were performed using the MATLAB software package and statistics toolbox (The MathWorks, Inc 2015).

Results

Simulated surface soil P concentration varied among the five experimental runs (Figure 4) as well as among lake drainage areas (Appendix Figure B1 in ESM). The variations across drainage areas could be explained by differences in the spatial distribution of land cover types and the corresponding applications of fertilizer and manure. Soil P was greatest in the Lake Mendota drainage area because of the prevalence of dairy operations and high rates of manure application north of Lake Mendota. Average surface soil P in croplands changed over time for each simulation as well (Figure 5) due to climatic variability as well as equilibration processes within the soil system. For example, in the case of LO and ML, soil P increased between 1986 and 2013 as applications of fertilizer and manure accumulated in the surface layer. In contrast, for the high starting concentrations in MH and HI, surface soil P decreased through time as rainfall leaching and physical mixing caused excess P to transport to lower soil layers.

In general, the magnitude of P fluxes within the watershed increased and water quality degraded with each simulation as P storage increased. This was the case for mean P yield which increased across the simulations (Figure 6). Mean yields were statistically different according to ANOVA ($p < 0.01$) and ranged from 0.70–1.43 ± 0.12 (SD) kg ha^{-1} for LO and HI, respectively, in the Lake Mendota drainage area. Similarly, mean P yield spanned 0.67–1.39 ± 0.12 kg ha^{-1} for Lake Monona, 0.64–1.36 ± 0.12 kg ha^{-1} for Lake Waubesa, and 0.61–1.33 ± 0.12 kg ha^{-1} for Lake Kegonsa.

Direct drainage loads into the four lakes also increased with P storage (Figure 7A). For Lake Mendota, mean loads for the five simulations were statistically different ($p < 0.05$) and ranged from 18,662–32,500 ± 4276 kg y^{-1}. Mean loads to the other three lakes were also statistically different across the simulations ($p < 0.01$) and ranged from 1389–4016 ± 953 kg y^{-1} in Lake Monona, 773–2382 ± 495 kg y^{-1} in Lake Waubesa, and 2350–5783 ± 1133 kg y^{-1} in Lake Kegonsa. Lake Men-

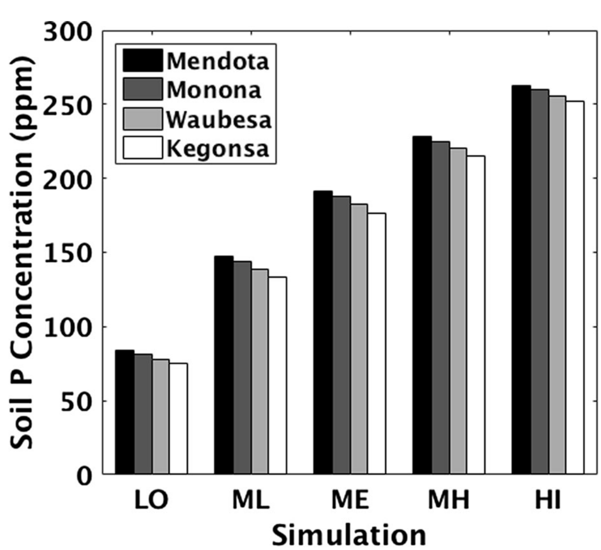

Figure 4. Surface soil P concentration in croplands (ppm) averaged over each lake's drainage area and the 1986–2013 time period.

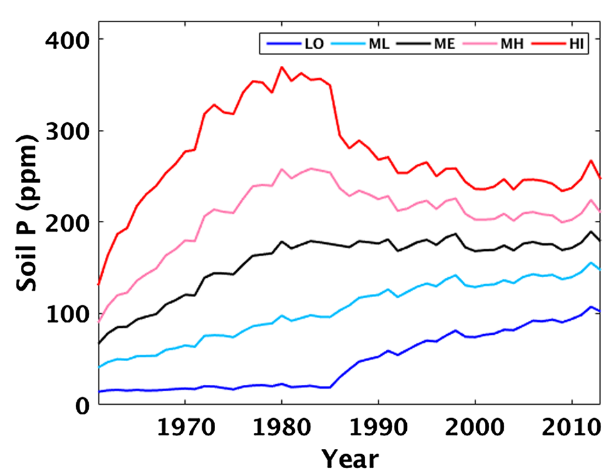

Figure 5. Watershed-average surface soil P concentration in croplands for both the spin-up period (1961–1985) and the simulation period (1986–2013).

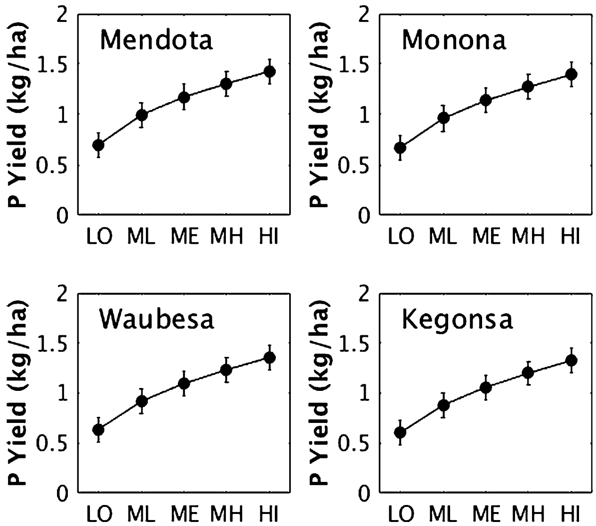

Figure 6. Mean P yield (kg ha^{-1}) in each lake's drainage area for the five simulations, 1986–2013.

dota received the highest direct drainage loads of any lakes followed by Kegonsa, Monona, and Waubesa in descending order. Direct drainage loads to Lake Mendota were more sensitive to P storage than the other lakes.

Summer TP concentration in each of the lakes was positively related to P storage (Figure 7B). Mean concentrations of TP were statistically different (p $<$ 0.05 for Lake Mendota; $p < 0.01$ for Lakes Monona, Waubesa, and Kegonsa) and ranged from 0.026–0.039 ± 0.006 mg l^{-1} in Lake Mendota, 0.028–0.033 ± 0.002 mg l^{-1} in Lake Monona, 0.047–0.059 ± 0.004 mg l^{-1} in Lake Waubesa, and 0.056–0.074 ± 0.008 mg l^{-1} in Lake Kegonsa. Among all four lakes, Kegonsa experienced the highest TP concentrations followed by Waubesa, Mendota, and Monona in descending order.

Consistent with the patterns observed in load and TP, Secchi depth decreased with increasing soil P (Figure 7C). Secchi depths were significantly different ($p < 0.05$ for Lake Mendota; $p < 0.01$ for Lakes Monona, Waubesa, and Kegonsa) and ranged from 2.05–2.26 ± 0.054 m in Lake Mendota, 1.75–1.81 ± 0.16 m in Lake Monona, 1.05–1.05 ± 0.002 m in Lake Waubesa, and 0.96–1.02 ± 0.009 m in Lake Kegonsa. Lake Mendota had the highest observed Secchi depths, with all five simulations exceeding the 2 m index for mesotrophy (Carlson 1977). None of the other lakes reached a 2-m depth in any simulation. The probability of hypereutrophy increased across the simulations (Figure 7D). Probabilities were significantly different ($p < 0.05$ for Lake Mendota; $p < 0.01$ for Lakes Monona, Waubesa, and Kegonsa) and ranged from 0.04–0.15 ± 0.074 in Lake Mendota, 0.04–0.06 ± 0.013 in Lake Monona, 0.18–0.32 ± 0.050 in Lake Waubesa, and 0.28–0.46 ± 0.052 in Lake Kegonsa. Lake Kegonsa had the highest probability of hypereutrophy followed

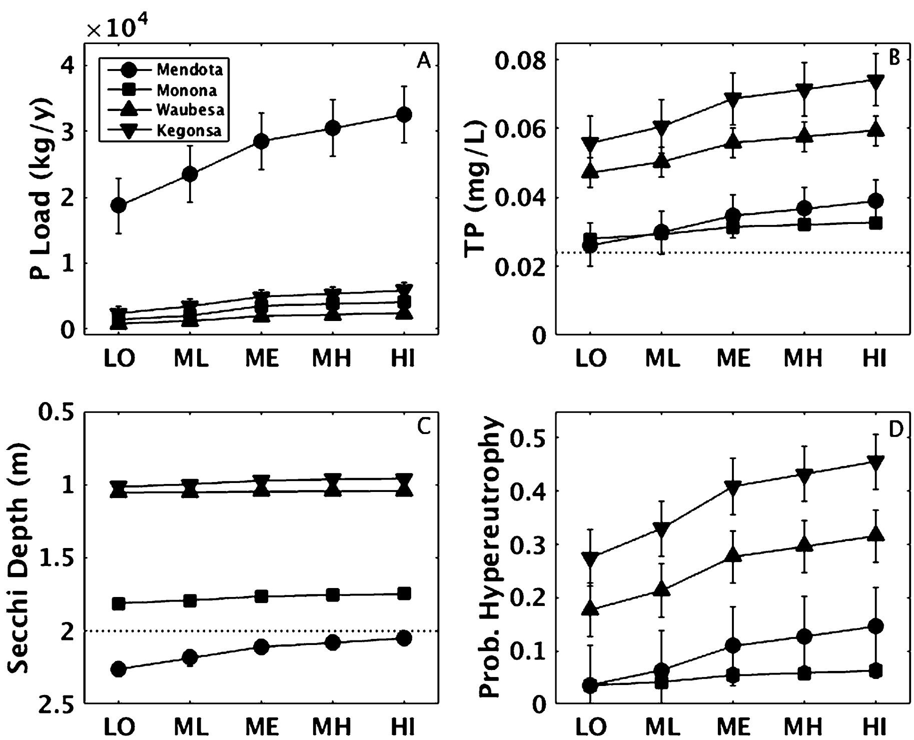

Figure 7. Water quality indicators for each simulation including **A** direct drainage P load (kg y^{-1}), **B** in-lake summer TP concentration (mg l^{-1}), **C** Secchi depth (m), and **D** probability of hypereutrophy. *Dashed lines* indicate the mesotrophic boundary.

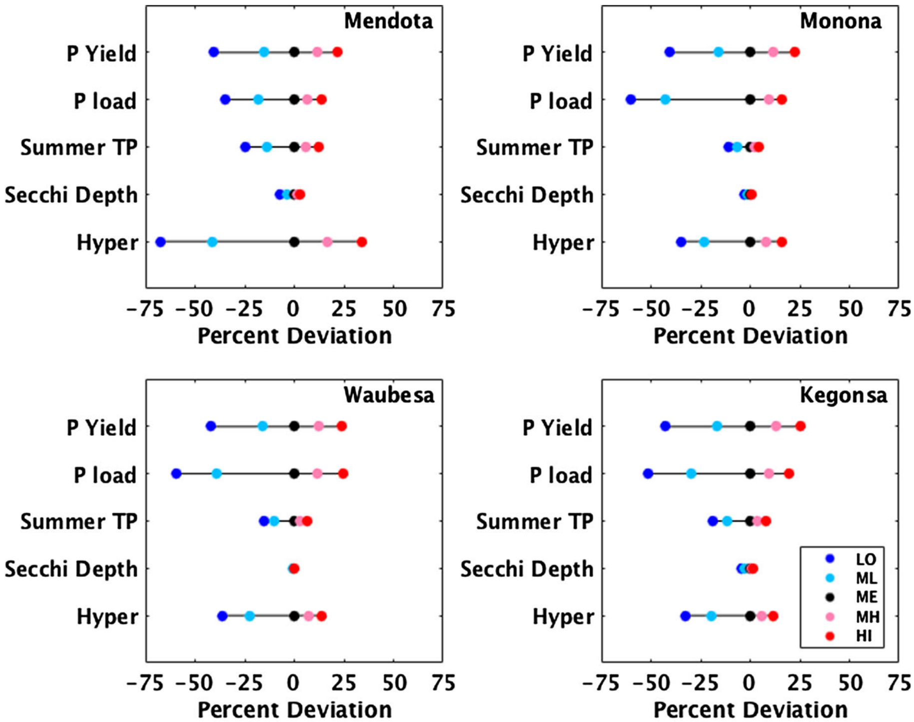

Figure 8. Percentage deviation in P yield, direct drainage P load, Secchi depth, and probability of hypereutrophy for each simulation compared with recent historical conditions (ME).

by Lakes Waubesa, Mendota, and Monona, in descending order. The lowest probability of hypereutrophy for any lake was 0.008, observed for Lake Mendota in LO.

We examined the percentage change in P yield, direct drainage load, summer TP, Secchi depth, and probability of hypereutrophy compared to the ME case, which best captured the recent historical record of P loads (Figure 8). In general, the deviations from ME were different among the five variables and the four lakes. For Lake Mendota, the deviations in P yield matched the relative spacing among soil P trajectories (Figure 5), with a greater absolute deviation occurring in LO (−40%) than in HI (+22%). The relative deviations in P load and summer TP were similar to the relative deviations in P yield although they spanned a smaller range. Secchi depth was the least responsive of the five variables, spanning the greatest range in Lake Mendota (−7 to +3%) and the smallest range in Lake Waubesa (−0.6 to +0.2%). The probability of hypereutrophy varied the most among all five variables, spanning the greatest range for Lake Mendota (−67 to +34%), and the smallest range in Lake Kegonsa (−33 to +11%). For the downstream three lakes, the deviations in P load for LO and ML were greater than the deviations in P yield. The responses of summer TP, Secchi depth, and the probability of hypereutrophy were muted, however, compared with the responses of these indicators in Lake Mendota. This suggests that load reductions in the lower lakes do not result in the same degree of water quality improvement that would be expected in Lake Mendota.

We also analyzed the response of summer TP concentration to the frequency of extreme precipitation events for each simulation (Figure 9). Linear regression analysis of log(TP) versus the number of weeks per year with rainfall exceeding 100 mm (r100) was performed for each simulation and lake combination. The analysis indicated that the relationship between r100 and log(TP) was significant in all cases ($p < 0.01$), with R^2 values for all regressions near 0.66. After verifying that plots of the residuals lacked discernible pattern, we used ANCOVA to confirm that the slopes of the regression lines were significantly different ($p < 0.01$) among the simulations for each lake. In general, the slopes increased across the simulations, indicating a higher sensitivity of TP to extreme precipitation events as P storage increased. This suggests that lake water quality may be more vulnerable to extreme precipitation when P storage is high, and conversely less vulnerable when P storage is low.

Discussion

We tested the effect that different levels of legacy P at the onset of a 27-year simulation period would

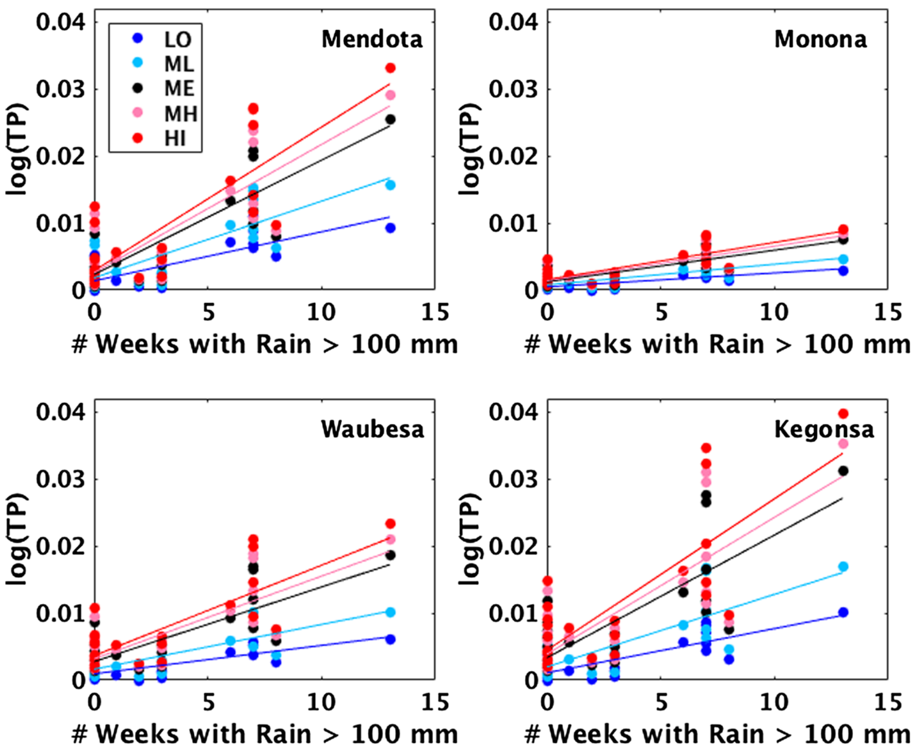

Figure 9. Log of summer TP concentration (mg l^{-1}) versus the number of weeks per year with rainfall exceeding 100 mm.

have on the magnitude and trajectory of P loading to the Yahara lakes and a suite of water quality variables. Our results showed that legacy P was an important determinant of future outcomes; the amount of P stored in soils and channel beds had a significant impact on lake loads and water quality. For Lake Mendota, the drop in initial P storage between ME and LO resulted in a 35% reduction in direct drainage P loads (Figure 8). This reduction led to a 25% decrease in mean annual summer TP concentration in that lake and increased the frequency of reaching mesotrophic status in summer (TP < 0.024 mg l^{-1}) by 25% (from 12 to 15 of 27 years). The water quality response to the LO legacy P reduction was muted in the lower lakes. Direct drainage load reductions of 60, 60, and 52% in Monona, Waubesa, and Kegonsa, respectively, did not cause TP concentrations to reach the mesotrophic threshold in any year despite exceeding the current management target for the Yahara lakes of a 50% reduction in lake loading (Lathrop and others 1998). Our results suggest that lower direct drainage loads than those simulated in LO would be needed to achieve the target TP concentration in all four lakes. It should also be noted that total loads to the lower lakes are generally dominated by riverine loads from upstream lakes (Carpenter and Lathrop 2014). Reductions in direct drainage loads achieved through changes in land use therefore represent only a partial reduction in total loads to the lower lakes. It may be necessary to also remove P stored in upstream lake sediments to decrease total loads to the downstream lakes sufficiently to alter their trophic status. The effect of lake sediment P storage was not examined in this analysis since it is implicitly handled by the Yahara WQ Model.

By using the same historical nutrient applications for all simulations, we isolated the effect of antecedent legacy P storage on water quality. This approach also showed the effect of historical nutrient applications on mean soil P concentration in the watershed. Specifically, the steady accumulation of soil P in LO over the 1986–2013 period indicated a buildup of P in the surface soil layer occurring as a direct result from historical nutrient applications and land management (Figure 5). Given that the starting soil P concentration for LO in 1986 corresponded roughly with the recommended level for crops (20 ppm), the buildup between 1986 and 2013 demonstrated that historical nutrient applications exceeded crop needs and were great enough to increase soil P above agronomic recommendations within a 27-year period. Additionally, ME represented the best estimate of soil P, P yield, and lake loads over the historical period and indicated that surface soil P was approximately in equilibrium at 176 ppm (Figure 5), and the second soil layer (2.5–15 cm) was roughly in equilibrium around 73 ppm (Appendix D in ESM).

This implied that historical application rates were high enough to sustain a soil P concentration roughly nine times the recommended level for crops in the surface layer, and roughly four times the recommended level in the second soil layer. Overall, these findings suggest that there is currently a substantial excess of P stored in watershed soils that is likely to influence outcomes of P cycling and lake water quality well into the future.

Among the four lakes, Mendota had the greatest absolute reduction in direct drainage P loading when comparing ME and LO (Figure 7A). This is likely due to its large direct drainage area and close proximity to dairy operations in the northern YW. Additionally, all loads entering Mendota come from direct drainage sources, whereas the majority of total loads to the lower lakes come from the upstream lakes. As a result, water quality is generally worse in the lower lakes and less variable year to year. Water quality in the Yahara lakes is strongly influenced by the presence of *Daphnia pulicaria*, an effective grazer of algae (Lathrop and Carpenter 2013). Among zooplankton species, *D. pulicaria* is exceptionally effective in reducing lake TP concentrations because of its rapid grazing rate, broad diet, high metabolic demand for P, and sedimentation of P in its feces (Carpenter and Kitchell 1993). Our results were calculated assuming the presence of *Daphnia*. Without the presence of *Daphnia*, summer TP concentrations in Lake Mendota would be approximately 30% higher in Figure 7b for each simulation (results not shown). Populations of *Daphnia* were relatively steady over the time period examined in this study; however, the recent invasion of spiny water flea (*Bythotrephes longimanus*) within the lakes has resulted in a collapse of *Daphnia* populations, making the future presence of *Daphnia* in the lakes uncertain (Walsh and others 2016). Given its important role, managing the Yahara lakes to maintain *Daphnia* populations should be a priority.

Prior studies have shown that heavy precipitation events can mobilize large amounts of soil P in runoff (Kleinman and others 2006; Shigaki and others 2007). Carpenter and others (2014) showed that heavy runoff events are responsible for the majority of annual total P loading to Lake Mendota. They found that on average 29 precipitation days per year (roughly one-quarter of all days with measurable precipitation) accounted for 74% of annual loading to the lake over the 1976–2008 period. Our findings were consistent with these observations; however, we also observed that the response of summer TP to high precipitation (and thus high load) events was greater at higher levels of P storage (Figure 9). This effect is due to greater amounts of stored P being mobilized during heavy rain events. Supporting analysis (results not shown) did not find this effect to be more or less pronounced for dissolved or sediment forms of P yield, or for different sources of P yield, such as manure, fertilizer, or soil. The implication of this finding is that water quality may be more vulnerable to heavy rainfall events at higher levels of P storage and conversely less vulnerable at lower levels. Given anticipated increases in annual and extreme precipitation with climate change (WICCI 2011; Trenberth 2011), this provides a twofold incentive for reducing P reserves in the watershed. Not only will the consequences of legacy P be exacerbated if accumulation continues, but if drawdown occurs, the benefits may include additional protection from extreme precipitation.

Gillon and others (2016) suggested that the water quality benefits of BMPs within the YW have likely been stymied by changes in precipitation, land use, and the problem of legacy P. Adoption of P control strategies within the YW has been fueled by collaborations among researchers, NGOs, stakeholder groups, resource managers, and local government. These include a variety of policies, regulations, and incentives geared toward the use of BMPs. Formalized nutrient management plans, conservation tillage, vegetative buffers, and manure digesters represent some of the BMPs that have been emphasized and implemented in the YW. However, adoption of these measures has not always been effectively targeted to areas of the landscape where they are most needed (Wardropper and others 2015). Additionally, many BMPs that target soil erosion and loss of particulate P will not fully address the problem since loads to Lake Mendota are roughly equally comprised of dissolved and particulate forms. Dissolved P loading from manured plots, particularly in late winter and early spring, represents a significant source of bioavailable P to surface waters. No-till conservation practices, which have been widely adopted in the YW, are effective at preserving topsoil and reducing sediment P loads but also have the unintended consequence of increasing DRP loads, particularly in systems receiving broadcast manure applications (Bundy and others 2001; Kleinman and others 2008). Conservation practices aimed at preventing P runoff can potentially worsen legacy P buildup, creating a ‘‘chemical time-bomb’’ that remains vulnerable to runoff and other transport mechanisms (Stigliani and others 1991). Conversion to perennial systems, including grasslands and forests, may not necessarily reduce loads since legacy P effects have been observed to be persistent in such systems even in the absence of high-intensity rainfall (Scott and others 2001; Bilotta and others 2007; Horrocks and others 2014).

Research elsewhere has shown that while BMPs can effectively stem P runoff at the field scale, there has been disappointingly little improvement in downstream water quality and ecology (Jarvie and others 2013a, b). Phosphorus losses from soil and fluvial sediment stores still occur in places of high P buildup and can effectively mask the water quality benefits of conservation efforts (Sharpley and others 2013; Sharpley 2016). By quantifying the direct effect that legacy P exerted on water quality in the Yahara, our study supports the general hypothesis that BMPs and conservation efforts are counteracted by the slow release of legacy P from watersheds and water bodies (Jarvie and others 2013a, b). Our results may also provide some support to observations of the legacy effects of other nonpoint source contaminants, such as nitrogen, sediment, and chloride (Bester and others 2006; Meals and others 2010; James 2013; Van Meter and others 2016).

With its strong agricultural history and growing population, the YW is an exemplar of many of the world's watersheds that support intensive agriculture. As such, the results of our study have important implications for other regions where there is intensive agricultural production and high P abundance, such as China (Chen and others 2012; Dai and others 2010), UK and other European countries (Withers and others 2001; Azevedo and others 2015), and elsewhere in the USA (Ardón and others 2010; Dubrovsky and others 2010). P yield, loading, and water quality will vary across landscapes according to differences in land use, management, topography, soil type, climate, etc. However, the basic principles governing the supply and transport of P apply to all regions. Using physically based models, we demonstrated a causal connection between legacy P and surface water quality. The strength of this connection may differ across regions, but its important role within the Yahara implies a similarly important role in other landscapes having a surplus of P and poor freshwater quality.

Our findings suggest that drawing down soil P reserves, for example through recovery and recycling of P in soils, channels, and manures, would help buffer against the unintended effects of management as well as climate change. For example, numerous conservation efforts were enacted over the past decade in an attempt to decrease P loading to Lake Erie. Yet, the actual changes in management and climate over this time period likely hindered those efforts. The adoption of conservation tillage and the subsequent need for broadcast P applications increased P accumulation in surface soils, thereby increasing P available to runoff (Sharpley 2016). Additionally, widespread adoption of tile drainage created an easy path for nutrient-rich subsurface drainage to enter waterways and ultimately Lake Erie. On top of that, extreme precipitation events increased over the time period and according to our results may have exacerbated the effect of increasing surface soil P on P losses. An alternative strategy for management that focuses on drawing down the P surplus, and not just preventing its loss in runoff, would address a root cause of the eutrophication problem. Removing P from the system precludes much of the uncertainty in management and climate, and how those factors may affect P cycling and transport. Our results suggest that successful removal of P from saturated watersheds promises improvements in water quality, as well as protective benefits.

The Yahara is a well-studied watershed having long-term observations and many prior studies from which to draw. Despite the wealth of data available, challenges still exist in performing simulations at the watershed scale. One challenge is insufficient data to initialize the model and represent the diversity of management practices that affect hydrology and nutrient cycling. For example, soil P may vary widely within watersheds (Wang and others 2009) and even within individual fields as a result of past management (Page and others 2005). In this study, the starting soil P concentration in 1986 across the watershed at a 220-m spatial resolution was not known and could only be approximated for different land cover types based on prior independent observations (for example, Bennett and others 2004). We relied on small-scale validation studies to verify the correct sensitivity of the model (that is, ability to simulate long-term soil P dynamics over a range of soil types, rainfall regimes, and management conditions) and calibrated the starting concentration to be within a reasonable range while also optimizing performance of P loading in THMB. Although simulated soil P and lake loads were plausible for each simulation, further research is needed to understand how spatial diversity in soil P is related to P loads at the watershed scale and to what extent input data must be improved.

Another challenge stemmed from the fact that in this study, the Yahara WQ Model was used to predict water quality outcomes using P loads that sometimes extended above and below the range of observations used to build the model (1975–2008). Therein lies a drawback of statistical models when used to predict outcomes under anomalous conditions, which stands in contrast to their relative ease of development and use. One motivation of process-based model design is to be robustly parameterized so as to respond appropriately to a wide range in inputs and therefore be well suited to simulate novel conditions of change. With the exception of the Yahara WQ Model, our modeling framework employed a process-based approach. This approach has helped us advance our understanding of the processes that affect water quality, such as the mechanisms driving soil P imbalance at the field scale and rates at which accumulation and depletion may occur at the watershed scale. We demonstrated that soil P may change rapidly over several years or remain unchanged for decades or longer. Our model is thus poised to investigate timescales of watershed accumulation and depletion and the resulting impacts to water quality, an important knowledge gap in P research (Jarvie and others 2013a, b; Haygarth and others 2014).

Further studies using the modeling framework will focus on the effects of climate, land use, and land management to better understand the roles that these drivers play in P cycling and water quality, and the timescales over which they act. The framework also includes representation of other ecosystem services, including groundwater quality and recharge, flood protection, carbon storage, and food production. Representation of these will enable study of the interactions, tradeoffs, and synergies among multiple ecosystem services. The framework thus represents a comprehensive watershed modeling tool that can simulate the dynamic effects of major change drivers on water quality and other services key for human well-being.

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