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ORIGINAL ARTICLE

Shifting drivers and static baselines in environmental governance: challenges for improving and proving water quality outcomes

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Abstract Understanding the conditions that enable or constrain success in environmental governance is crucial for developing effective interventions and adapting approaches. Efforts to achieve and assess success in environmental quality improvement are often impeded by changes in conditions that drive outcomes but lie outside the scope of intervention and monitoring. We document how long-term changes in land use, agriculture, and climate act as non-stationary, shifting drivers of change that combine to render water quality management interventions less effective and increasingly difficult to assess. Focusing on the Yahara River watershed of south-central Wisconsin, USA, we ask how baselines influence program modeling, monitoring, and evaluation, as well as adaptation in governance approach. Through historical trend, GIS, and policy and qualitative data analyses, we find that changes in long-term land use and precipitation pattern dynamics exert tremendous pressure on water quality outcomes but are not captured in snapshot baseline assessments used in management planning or evaluation. Specifically, agricultural sector change related to the intensification of milk and manure production is increasingly challenging to address through best management practices, and flashier precipitation associated with climate change makes it difficult to achieve goals and establish a causal connection between management interventions and outcomes. Analysis of shifting drivers demonstrates challenges facing environmental governance in the context of climatic and social–ecological change. We suggest that goal setting, program design, and evaluation incorporate new modes of analysis that address slowly changing and external determinants of success.

Keywords Environmental governance · Shifting drivers · Water quality · Climate change · Land use change · Agricultural intensification

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Introduction

Understanding the conditions that enable or constrain success in environmental governance is crucial for adapting and improving approaches (Ostrom 2009). Researchers and practitioners suggest that effective intervention requires knowledge of social and ecological context and outcomes, so learning and adaptation can take place

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(Dietz et al. 2003; Armitage et al. 2008; Herrick et al. 2012). Determining institutional and policy effectiveness is, however, challenging because of the need to identify causal links between policies, interventions, and environmental conditions (Young 1999).

Common outcomes assessment difficulties include multiple and conflicting goals for intervention and monitoring, scientific uncertainty in outcome monitoring and modeling, and mismatches between the spatial and temporal scales of implementation and monitoring (Rissman and Carpenter in press). Incomplete or inaccurate problem definition and assessment can misrepresent the causes and consequences of social–ecological change and intervention efficacy, which may misinform future governance (Forsyth 2003).

Developing a set of starting or baseline conditions to measure change against is a common strategy for monitoring and evaluating policy effectiveness. For example, water quality improvement projects often develop baselines to first predict and then measure the impact of policy interventions (e.g., Cadmus Group 2011). By definition, the establishment of a baseline casts social and ecological conditions as static so that future outcomes can be measured against past conditions (e.g., land use, management practice, policy, economy, climate; see Campbell et al. 2009). Social and environmental change, however, continues after baselines are set and often goes unacknowledged, even if changes influence or determine outcomes (Duarte et al. 2009). Here, we refer to changes in baseline conditions that drive outcomes as "shifting drivers"; these drivers shift relative to the static baselines set to evaluate policy effectiveness and often lie outside the scope of intervention and monitoring.[1] Even when a particular intervention is expected to produce successful outcomes, shifting drivers can mask improvement and make an intervention appear as a failure (Sharpley et al. 2009).

The challenges posed by shifting drivers apply to both measured and modeled outcome evaluations (see Holmes et al. 2009; Wagner et al. 2011; Radcliffe et al. 2009). In a predictive modeling and statistical context, shifting drivers are described as non-stationary. Where a stationary time series has a probability distribution and statistical properties that do not vary over time and provide a predictable basis for calculations of likely future conditions, non-stationary factors do vary. Non-stationary factors in water resources modeling often include climate, land use, artificial drainage, and water infrastructure (Milly et al. 2008; Hirsch 2011). Due to unacknowledged non-stationarity, models can systematically ignore certain signals or misrepresent likely outcomes, and these challenges are likely to increase under uncertain climate change (see Milly et al. 2008; Galloway 2011). Water resources management research has focused on non-stationary precipitation's implications for water quantity, water supply, and flood control (e.g., Villarini et al. 2011), but changes in water quality (Johnson et al. 2012) and land use and ecosystem dynamics (Olsen et al. 2010; Craig 2010) can also be considered non-stationary.

Shifting non-stationary drivers is one of the many challenges for agricultural watershed nutrient management, in which governance failure is much more common than success (Duarte et al. 2009; Sharpley et al. 2009; Harris and Heathwaite 2012; Jarvie et al. 2013).

While the challenges of shifting drivers for baselines and measured evaluation are recognized, few analyses have documented the long-term effects of shifting drivers on environmental outcomes and associated governance implications (see Bechmann et al. 2008; Duarte et al. 2009; Joosse and Baker 2011; Reckhow et al. 2011; Haygarth et al. 2012). Here, we document how slowly, unpredictably, and directionally shifting drivers of change that lie outside the scope of management and evaluation frustrate scientist and practitioner ability to establish management goals, accurately measure intervention outcomes, and predict and plan for change (see Jarvie et al. 2013). We differentiate shifting drivers and diagram how they alter intervention outcomes and assessment (Fig. 1), tracking the history of these trends through a case study of the Yahara River Watershed (YRW), an urbanizing agricultural watershed in Wisconsin, USA. Specifically, we ask: (1) How have water quality improvement efforts relied on quantitative goals, baselines, and models for program design and evaluation? (2) How have long-term changes in land use, agriculture, and climate impacted efforts to reduce watershed nutrient levels to target conditions? And (3) How does persistent failure to achieve goals due to shifting drivers affect governance dynamics? We find that intensification of milk and manure production, long-term land use change dynamics, and increasingly flashy precipitation events combine to render water quality management interventions less effective and increasingly difficult to assess. We conclude with lessons

[1] Historical marine ecologists have drawn attention to "shifting baselines," where "…each generation of scientists accepts as a baseline the stock size and species composition that occurred at the beginning of their careers and uses this to evaluate changes. When the next generation starts its career, the stocks have further declined, but… serve as a new baseline" (Pauly 1995: 430). Over time, this results in a gradual decline of baseline conditions that accommodate ineffective governance responses. When baseline conditions are set and used to measure governance efficacy in more compressed timeframes, however, they are static baselines, rather than baselines that shift over generations or to acknowledge ongoing changes in baseline conditions in order to more accurately measure intervention effect. See Campbell et al. (2009) on the human-environment implications of shifting baselines and the limits of baseline use, in general.

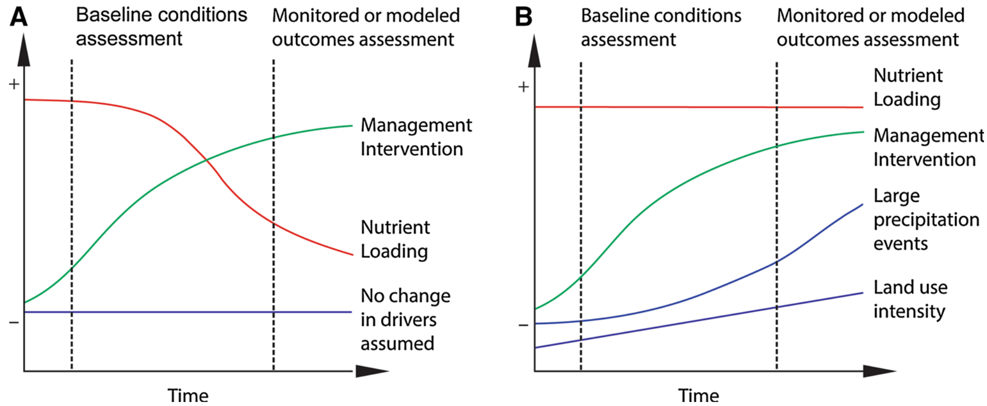

Fig. 1 Shifting drivers challenge assumptions about intervention–outcome relationships. **a** Expected trajectories of management intervention and nutrient loading outcomes, given assumption of stationary drivers of change. **b** Actual trajectory of management intervention and nutrient loading outcomes, given shifting drivers that counteract management effort

for environmental governance affected by shifting drivers and non-stationarity.

Method

We answer the questions above through quantitative and qualitative analyses to demonstrate both the quantitative influence and effects of shifting drivers and the challenges they pose for governance. In "Baselines, goals, and frustrated efforts to improve Yahara River watershed water quality" section, we draw on historical, policy, and YRW water quality management data to address the first question and explain baseline use and outcomes in YRW water quality improvement efforts. The second and third research questions on shifting drivers and their governance implications are answered in "Shifting drivers in land use, land use intensity, and climate" section with GIS-based analysis of land use change between 1939 and 2010, analysis of historical agricultural trends, and analysis of water quality outcomes (see Supplement for explanation of data sources). The nonparametric Mann–Kendall test (Mann 1945; Kendall 1975) and linear regression (Helsel and Hirsch 2002) determined statistically significant trends for quantified drivers of change (Table 2).

Throughout, quantitative data analysis is contextualized with analysis and explanation of policy documents, scientific and management literature, news media, and qualitative data from attendance at regional water management meetings and interviews ($n = 56$), surveys ($n = 51$), and workshops ($n = 51$), which were conducted as part of a larger project on regional water resources governance (see https://wsc.limnology.wisc.edu/). The 82 participants included municipal, county and state public officials, NGO representatives, agricultural producers, and other residents. Research participants were identified by their role in regional water quality management and snowball sampling; interviews were recorded and transcribed; and data were analyzed using a grounded theory approach (see Bernard 2006).

Baselines, goals, and frustrated efforts to improve Yahara River watershed water quality

The Yahara River watershed (YRW) of Dane County, Wisconsin, is an urbanizing, agricultural watershed surrounding the Yahara River chain of lakes (Lakes Mendota, Monona, Waubesa, and Kegonsa; see Online Resource Figure 1). These lakes stretch through the Madison metropolitan area and constitute an important cultural, ecological, economic, and scientific resource for the region. The Madison lakes, which helped secure the city's status as state capital, continue to draw residents and support regional economies and recreation (Mollenhoff 2003). The lakes have a long history of water quality problems and have been targets of intensive management efforts and scientific research for over a century; Lake Mendota is often cited as the most studied lake in the world (Carpenter et al. 2006). Since the late 1800s, the most pressing water quality objective has been to control sometimes toxic blue-green algal blooms nutrient inputs cause. Nutrient sources have included a mix of point and nonpoint sources from urban sewage, eroded agricultural soils, nitrogen fertilizer, phosphorus-rich manure, urban storm water, and construction site erosion (Lathrop 2007, see Online Resource Figure 2).

Nutrients from sewage and agricultural production were first recognized as the cause of algae growth in the mid-1940s (Lackey and Sawyer 1945; Flannery 1949). After sewage was diverted downstream of the lakes in 1971, decreasing nonpoint source nutrient loading (P, in particular) to reduce algal blooms has been the primary water quality management goal. Toward the end of the twentieth century, algal blooms intensified with nonpoint source pollution, negatively impacting lake access and use, as well as human and wildlife health (Lathrop 2007). Recent regional management efforts and research have focused on reducing phosphorus (P) loading, the limiting factor in algal growth, which is typical for eutrophic freshwater lakes, although nitrogen availability and light transmission also limit blooms during certain times of the year, particularly in downstream lakes (Lathrop 2007). Agricultural production now contributes approximately 75 % of Lake Mendota's ~70,000 pounds average annual P load (Yahara CLEAN 2010; MARS 2011). Nutrient loading in Lake Mendota drives nutrient levels in the downstream lakes, making reductions from agricultural sources important for the entire YRW lake system (Carpenter and Lathrop 2013; Lathrop and Carpenter 2014).

Effort to control nutrient pollution in the YRW has been persistent, but resulted in negligible water quality outcome improvements. Unacknowledged shifting drivers have frustrated optimistic efforts to improve water quality by establishing baselines, implementing policies, and gauging outcomes for adaptation throughout persistent YRW management efforts. Since the 1980s, water quality improvement targets have remained fairly consistent at 50 % P loading reduction, while baseline definitions used for planning have changed. Although baselines are updated for each project (or shift; cf. Pauly 1995), each baseline is a static representation of watershed conditions that lags behind actual, dynamic changes occurring during project modeling and implementation phases (Table 1). After each baseline assessment, substantial ongoing change in land use, land cover, climate, and agriculture has taken place within the project timespans, undermining water quality improvement efforts.

To illustrate a typical approach to defining baselines, we first briefly discuss a recent total maximum daily load (TMDL) analysis for total phosphorus (TP) and total suspended solids for the Rock River watershed, a larger watershed basin to which the YRW contributes (Online Resource Figure 1). The Rock River TMDL has become a major organizing force for the construction of baselines, goals, and interventions for watershed nutrient management. As is the case with all baseline assessments described in Table 1, the TMDL defines watershed baseline conditions statically although water quality drivers shift during the implementation period, making gaps between expected and actual outcomes likely. The TMDL report states (Cadmus Group 2011: 24):

> Because [the SWAT model] held other factors (e.g., land cover and agricultural practices) constant [at 1992 levels]…, precipitation was the only dynamic variable that affected loading. …None of the monthly averages differed significantly, which indicates that the 30-year average distribution of precipitation events across the year is well represented by the 1989–1998 period. In addition, this period is comparable with the time period of land use data (1992).

Although the TMDL held land use constant at 1992 conditions, patterns have changed since the baseline condition was set for modeling. Precipitation patterns have also changed. Section four shows that baseline assumptions of average rainfall and static land use are increasingly untenable.

Baseline assessments for watershed nutrient pollution began to receive attention by the 1970s, as agricultural nonpoint source pollution reduction efforts became increasingly targeted and sophisticated. A 1975 Dane County report meant to inform one of the first comprehensive nutrient management plans drew attention to the importance of the problem posed by historical trends in agriculture (Barry et al. 1975). The authors conducted a sub-watershed agricultural baseline assessment describing change

Table 1 Yahara River watershed nutrient management baselines and goals

Project	Land use baseline for modeling	Precipitation baseline for modeling	P goal	Approach	P goal reached?
Lake Mendota Watershed Project (1968–1972)	Undefined	Undefined	Reduce winter manure spreading	Manure storage construction	N/A
Sixmile-Pheasant Branch Creeks PWP (1980–1989)	1976–1977	1976–1977	Reduce sedimentation	Urban and agricultural BMPs	No
Yahara-Monona PWP (1988–1997)	1990; 1980–1988 for urban growth	Average year	30–50 % reduction	Urban and agricultural BMPs	Not monitored
Lake Mendota PWP (1994–2008)	1994–1996	Average year	50 % reduction	Urban and agricultural BMPs	No
Rock River TMDL (2011)	1992, 1998 for tillage survey	1989–1998	Variable, up to 72 % reduction	Urban and agricultural BMPs	In progress
Yahara Clean Strategic Action Plan (2012)	2005, 2008	1995–2008	50 % reduction	Urban and agricultural BMPs	In progress

Describes major Yahara River watershed water quality improvement efforts' baselines, goals, approaches, and phosphorus (P) reduction outcomes. Goals are often established and outcomes assessed based on monitoring, while project planning occurs through modeling. Baselines listed above are snapshots of conditions used as inputs in models that help target management interventions at the beginning of long project implementation timelines. Actual, ongoing changes in these baseline conditions make predictions of management success unreliable between 1965 and 1974 and found that average farm size increased 28 % (171–219 acres); farm numbers decreased 30 %; field corn acres increased 58 % (13,191 acres); and cattle increased 21 % (4325 head; Ibid.: 86). The report concluded that "The analysis of historical trends in the Lake Mendota watershed shows an increase in the agricultural practices which pose the greatest nutrient runoff potential to our lakes. This trend, which shows little sign of slowing, would result in an ever worsening quality of the lake resources, if unmanaged" (Barry et al. 1975: 112). Despite early attention to the problem historical trends posed, subsequent efforts have failed to substantively address these dynamic watershed changes in goal setting and evaluation.

By 1981, YRW water quality management began focusing on sub-watershed analysis and intervention, particularly through Wisconsin Department of Natural Resources (WDNR) designated priority watersheds that received state funding for planning, implementation (with 50–70 % cost sharing), and monitoring. The Sixmile-Pheasant Branch Creeks priority watershed project began in 1981 in the western half of Lake Mendota's drainage basin; project success was assessed in 1990. The assessment concluded that "Data presented failed to document significant change in water quality …resulting from BMP implementation" (Miller 1991: 1). Expected improvements were not realized due, in part, to shifting drivers. The watershed was estimated to contain 17,000 dairy cattle (Miller 1991), a number that grew during project implementation (WDNR, Pers. Comm. 2013). Other reasons for project failure included high legacy sediment and P loading, low farmer participation levels, and urban construction site runoff; each is associated with the shifting drivers discussed below (Miller 1991; Carpenter et al. 2006).

Project goal setting and assessment have become more quantitative, but projects continue to be frustrated by shifting drivers. The Lake Mendota Priority Watershed Project (1994–2008) is one significant example. The goal, as in current projects, was to reduce P loading by 50 %, which models showed would decrease summer algal blooms likelihood from six to two out of ten days (Lathrop 1998; Betz 2000; Yahara CLEAN 2010). The 14-year effort included a baseline conditions assessment phase and an 11-year implementation period. With federal, state and county government support, urban and agricultural BMPs were implemented, including 46 barnyard runoff systems, 58 acres of grassed waterways, 149 acres of buffer strips, over 3000 feet of stream bank protection, 18.8 acres of wetland restoration, ten water diversions, seven sediment basins, and two terraces. Despite this effort, the project failed to achieve its goal (Lathrop 2007; Yahara CLEAN 2010; Genskow and Betz 2012). Lathrop and Carpenter (2011) wrote on the failure:

> Given that a significant number of agricultural and urban best management practices have been installed in Mendota's watershed in recent decades—especially during the implementation phase of the Lake Mendota Priority Watershed Project in 1998–2008—the lack of a significant decline in long-term average P loads is disconcerting. One explanation is that the pollution reduction gains from the installed practices were offset by an increased frequency of extreme precipitation events as well as a worsening manure management problem in the watershed. On the positive side, if the management practices had not been installed, then P loadings would likely have been much higher in recent years.

Despite impressive collective watershed nutrient management efforts and consistent agricultural conservation practice implementation (Wardropper et al. 2015), annual P loading to Lake Mendota has not significantly decreased (Lathrop 2007; Lathrop and Carpenter 2014). Lake Mendota P loads are highly variable over time, but show no significant declining trend (Fig. 2). After multiple projects' failure to meet goals, optimistic predictions about management outcomes that leave shifting drivers unaddressed have become less convincing.

Shifting drivers in land use, land use intensity, and climate

In sections "Land use change in an urbanizing, agricultural watershed," "Agricultural intensification," and "Climate non-stationarity and water quality management," we draw on quantitative analysis of land use, agricultural, and precipitation pattern change to elaborate how each represents a shifting driver of watershed nutrient outcomes that challenge efforts to both improve and prove water quality (Table 2). Throughout, we use policy analyses and qualitative, interview-, survey-, and workshop-based data to provide context for quantitative analyses; in "Persistent failure and changing governance dynamics" section, we use these data to explore the relationship between shifting drivers and regional water governance.

Land use change in an urbanizing, agricultural watershed

Land use and land cover changes have been ongoing and negative for water quality, but largely unaccounted for in water quality efforts, baseline establishment, and evaluation. Key land use and land cover changes influencing water quality outcomes revealed by our detailed analysis of change between 1939 and 2010 include (1) increasing

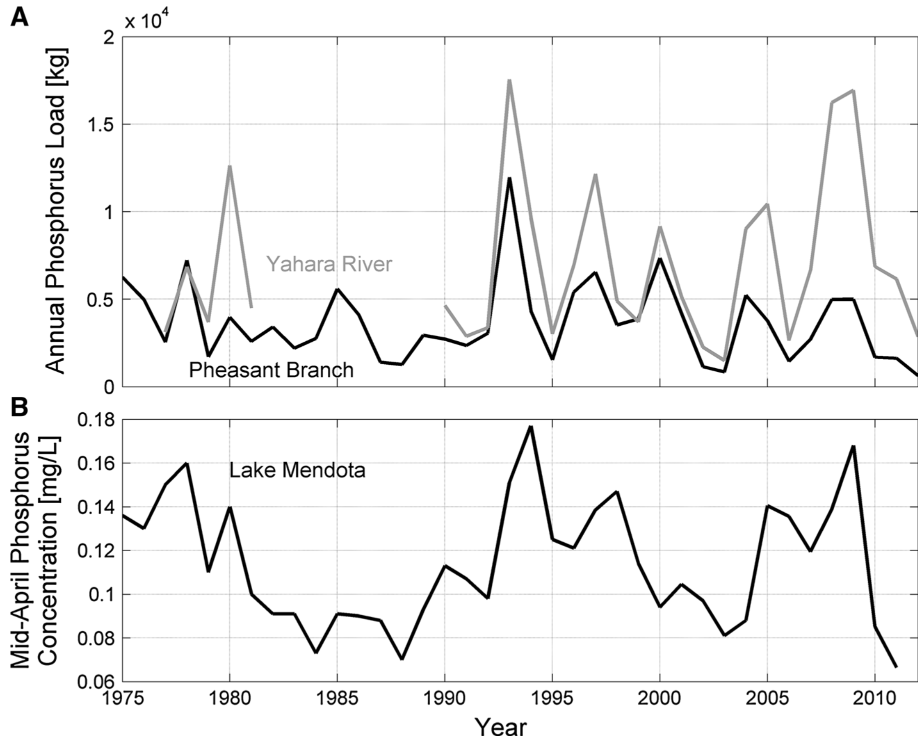

Fig. 2 Observed phosphorus loading to and concentrations in Lake Mendota. Phosphorus loading data are from Lake Mendota's two largest tributaries, the Yahara River and Pheasant Branch. Lake Mendota phosphorus concentration data collected annually during spring mixing (turnover) period. Both time series vary highly with precipitation, but show no significant declining trend (NWIS 2013, NTL LTER 2013)

Table 2 Shifting drivers of water quality outcomes

Driver of change	Direction of change	Magnitude of change	Change period
Phosphorus loading	None	No significant trend*	1975–2012
Nutrient management efforts	Increase	See Table 1	1978–2011
Urbanization	Increase	+1300 acres/year	1939–2010
Agricultural production acres	Decrease	−3100 acres/year***	1939–2012
Corn and soybean production acres	Increase	+2250 acres/year***	1939–2012
Pasture acres	Decrease	−3500 acres/year***	1939–2012
Conservation Reserve Program acres	Decrease	−980 acres/year***	1990–2012
Number of cows	Decrease	−590 cows/year***	1939–2012
Milk production per cow	Increase	+117.9 kg/cow/year***	1939–2012
Manure production per cow	Increase	+72.6 kg/cow/year***	1939–2012
Soil phosphorous levels	Increase	+0.43–1.08 mg/kg/year	1860–2001
Large storm events, >50 mm	Increase	9.5 events/decade, 18 events/decade,	1931–1990, 1991–2010
Large storm events, >75 mm	Increase	1.8 events/decade, 6 events/decade	1931–1990, 1991–2010
Annual precipitation	Increase	+2.2 mm/year**	1931–2012

Major drivers of water quality outcome change in the Yahara River watershed, listed with direction, magnitude and period of change. Trends are significant at $p < 0.05$ (*), $p < 0.01$ (**), and $p < 0.001$ (***). Despite persistent conservation and water quality management intervention, phosphorus loading shows no significant declining trend due to the shifting, non-stationary drivers of change listed. Driver rates of change are averaged over time period listed. Manure production was calculated based on its relationship with milk production (Nennich et al. 2005). Soil phosphorus change was calculated from an analysis of soil P difference among Dane County prairie, dairies, and grain farms (Bennett et al. 2005). Significance not tested for urbanization, soil P levels, or large storm events due to limited data points

urbanization shrinking the agricultural land base available for manure spreading, (2) urbanization increasing watershed impervious surface and runoff reaching the lakes, (3) decreasing pasture area and increasing row crop area that exposes more nutrient-laden soil, and (4) reductions in nutrient-removing wetland acreage. We describe these in further detail. The land use change patterns described here are likely to constitute shifting drivers in many urbanizing agricultural watersheds (Joosse and Baker 2011; Sharpley et al. 2009).

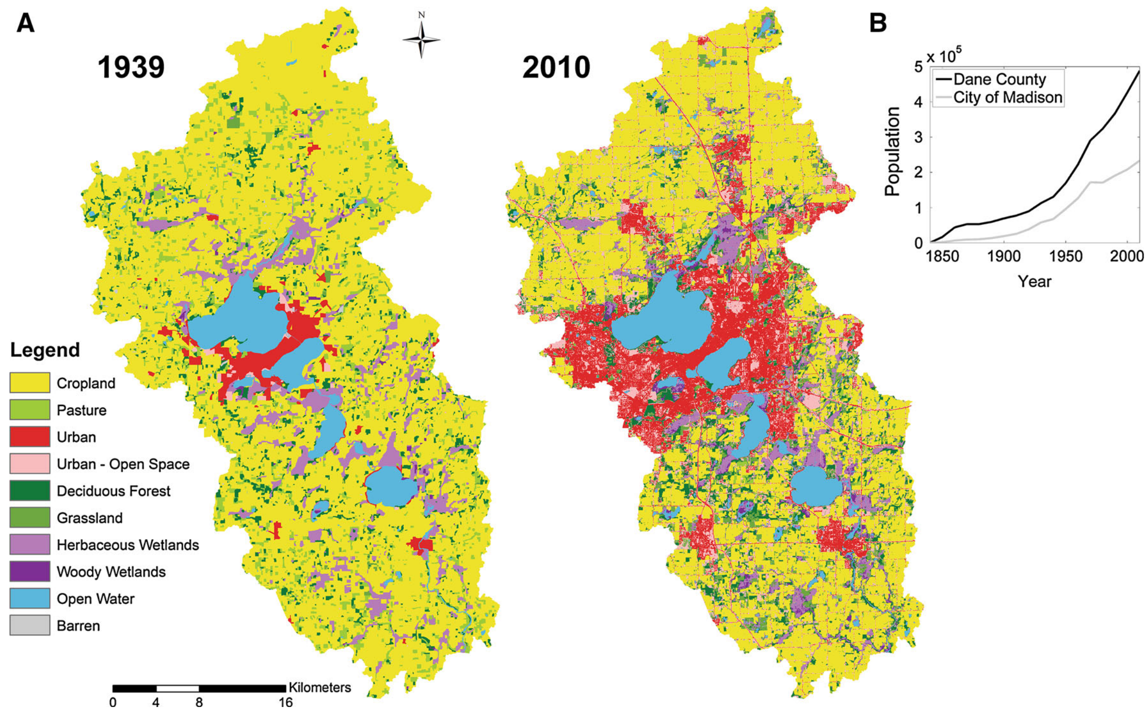

Fig. 3 Yahara River watershed land cover (1939 and 2010) and Dane County and City of Madison Population. **a** The growth of the Madison metropolitan area has been concentrated around the two largest watershed lakes, Mendota and Monona. Agriculture has intensified and remained a dominant land use, despite a shrinking land base (see methods for data sources). **b** The populations of Dane County and Madison (http://www.census.gov/) have steadily increased over time. Dane County's higher relative growth rate after 1950 is due to a period of rapid suburbanization

The watershed has urbanized as Dane County and Madison metropolitan area populations increased over the last 100 years, particularly in the last several decades in areas surrounding Madison (Fig. 3). The urban footprint expanded from 3.4 % of the watershed area in 1939 to 26.7 % in 2010. Urbanization and farmland acreage reductions impact water quality in several ways: (1) as farmland declines, less land is used to spread more manure (Cabot et al. 2004), resulting in higher P loading risk to downstream water bodies (see "Agricultural intensification" section); (2) construction site runoff results in high sediment and P loading to surface waters (Owens et al. 2000); and (3) increased impervious surface results in elevated runoff, sediment, and P loading from stream bank erosion (Krug and Goddard 1986). Increased regulation of urban developments in response to 1987 Clean Water Act amendments and county and municipal regulations have likely reduced relative contributions of urban P sources further.

Although important during phases of rapid (sub)urbanization, phosphorus pollution from urban areas is dwarfed by agricultural sources. Within the YRW, total cropland declined from 68.3 % of the basin area in 1939 to 45.6 % in 2010. Within a declining agricultural land base, the transition away from pasture and small grains to more row crops impacted water quality. Wheat and other small grain production, pasture, and woodland were most common immediately following European settlement. The reduced agricultural land base is now devoted almost exclusively to row crops (76 % in corn and soy production). In 1939, Dane County area in small grains was roughly equal to that in corn and soybean (16.2 and 16.6 %, respectively), but has decreased dramatically to just 1.8 % of county area (Figs. 3, 4). Row crop production grew from 16.6 to 34.7 % of county area between 1939 and 2010, while pasture acreage declined (from 8.45 to 0.85 %) as dairy production intensified. Field studies find corn production, which exposes soil for long periods of time, results in significantly more erosion than the production of other crops such as wheat (Edwards and Owens 1991). Highly variable levels of pasture quality make it difficult to assess the impacts of acreage declines, but pasture reduction does limit opportunities for spreading manure diffusely across pasture and increase soil exposure (see "Agricultural intensification" section).

Major changes in land use and cropping systems extend into recent decades. For example, Conservation Reserve Program (CRP) acreage declines have accompanied corn price rises arriving on the heels of federal biofuel production mandates (Online Resource Figure 3;

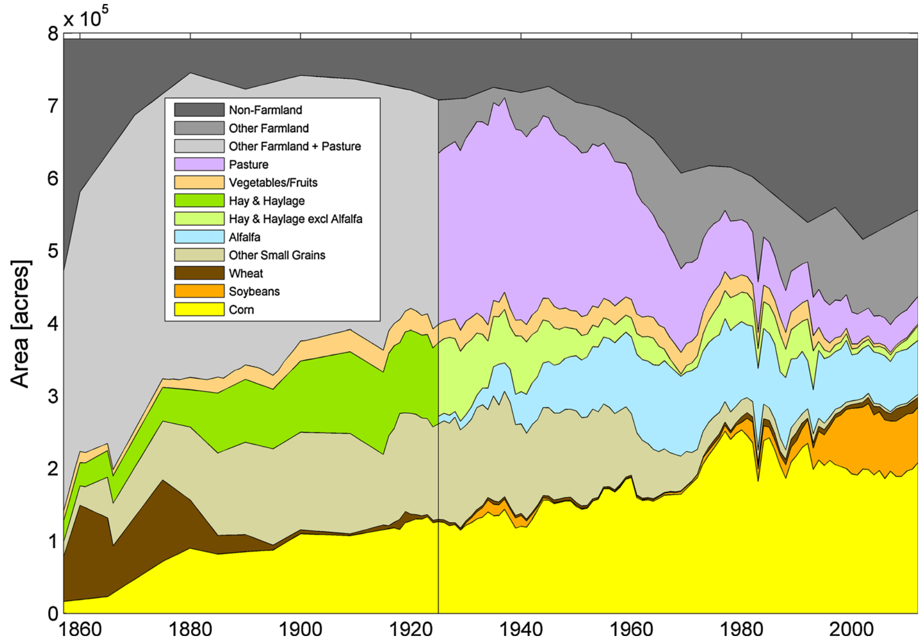

Fig. 4 Changes in farmland use, Dane County, Wisconsin. Total agricultural land in Dane County has declined with significant changes in crop type. Corn, soybeans, and intensive dairy agriculture have replaced wheat and pasture lands. Farmland includes cropland, on-farm woodland, and acreage dedicated to farm buildings. The majority of non-farmland acreage is urban and the remainder includes woodland, parks and water bodies. The USDA Agricultural Census began to separately account for pasture in 1925

Stuart and Gillon 2013; Wright and Wimberly 2013). Although the USDA counts CRP as cropland, converting CRP to row crops means setbacks for soil, water, and habitat conservation (Davie and Lant 1994; Karlen et al. 1999; Secchi et al. 2009). County CRP acreage distribution may be concentrated outside the YRW (spatial data are not public), but decreases have been widespread and county conservation officials have noted YRW CRP loss and row crop production intensification due to commodity price increases (Dane County Land and Water Conservation Department [DCLWCD], Personal communication, 2013).

Wetland loss may also be a shifting driver impacting downstream water quality (Mitsch and Gosselink 2000). YRW wetland extent declined from 13.9 % prior to settlement (defined by hydric soil data [USDA 2013]) to 7.9 % in 1939 and 6.6 % in 2010, consistent with trends throughout the Upper Midwest (Dahl 1990). Wetlands were drained primarily for agriculture and urban development. Wetlands' diminishing effectiveness in water quality improvement due to invasive species and disconnection from water flowpaths is a concern in southern Wisconsin (Zedler and Potter 2008) that may represent another shifting driver. While wetland restoration can reduce nutrient loads and opportunities for re-establishing stream channel connections and riparian wetlands are abundant in the YRW (Rogers et al. 2009), restoring wetlands on soils with high P levels may do little to reduce nutrient loading (see Hamilton 2012). In sum, snapshot baselines (Table 1) do not effectively capture these changing land uses that can drive water quality outcomes.

Agricultural intensification

Analyses of land use change, historical agricultural data, water quality management data, and interviews indicate that the spatial distribution, intensity, and character of YRW agriculture have changed over time, forming another shifting driver of water quality. When agricultural production was established in the watershed in the mid-1800s, wheat production for export was dominant in the context of industrial revolution and demand increases. Regional agricultural producers began to lament decreasing soil fertility that continuous wheat production caused and looked to diversify production strategies (Hibbard 1904). Dairying became a viable farming option as European immigrants brought knowledge and skills with them to expand the production of milk, butter, and cheese. Soon, dairying was argued a superior model for regional agriculture precisely because manure would supply soil nutrients. The high soil P and other nutrient levels that dairying left behind were recognized as a sign of hard work and good farming, a foundation for future regional agricultural success (Hibbard 1904).

Agricultural intensification has turned dairying's nutrient assets into problems. As agriculture industrialized in the first half of the twentieth century, regional specialization, mechanization, farm expansion, and increasing adoption of off-farm inputs for production (i.e., synthetic fertilizers) became more widespread (FitzSimmons 1986; Goodman et al. 1987). Farmers became compelled to constantly reinvest in expanding production and improving efficiency to remain competitive (Cochrane 1979;

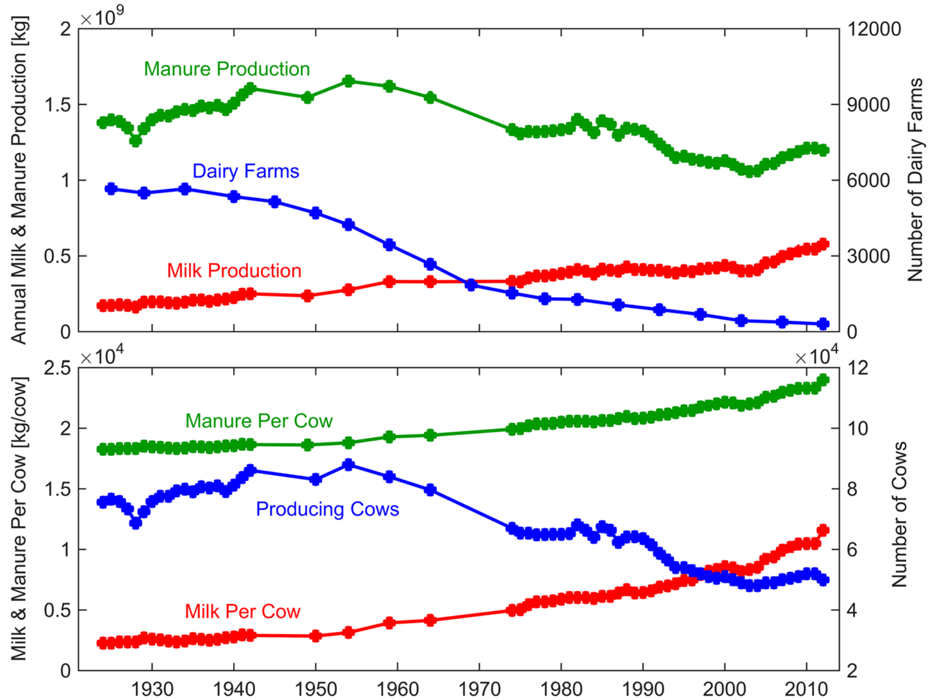

Fig. 5 Changes in dairy production and number of dairy farms in Dane County, Wisconsin. Dairy agriculture has intensified at both county and farm scales. Total county milk production has increased while the number of dairy farms has decreased. Each dairy cow now also produces more milk and manure than they have historically. Although manure production has declined since its peak in the 1950s, consolidation of dairies has concentrated manure on the few remaining dairy farms

MacDonald et al. 2007). This trajectory persists; one area dairy farmer indicated the pressure to expand his operation and herd size in saying that "There's no standing still—you're either going backward or forward" (Personal communication 2012). Pressures to intensify agricultural production have contributed to increasing YRW nutrient pollution levels (Lathrop 1992, 2007). A WDNR interviewee (Personal communication 2013) involved with the 1980s Sixmile-Pheasant Branch Creeks Priority Watershed Project referenced above described this dilemma: "What do you do with manure? Particularly large, growing livestock operations that, to meet their loan obligations, had to get larger. And as they got larger, they outstripped the size of their storage facilities and their animal lots. Those issues were there and are still there….."

With agricultural intensification, the number of farms producing milk has declined over time, but the average herd size and production levels have increased (Fig. 5; see Geisler and Lyson 1991; Cross 2006, 2012). Dane County dairying has become extremely productive, producing over one billion pounds of milk in 2006 (USDA 2013). Productivity increases have been facilitated, in part, by a shift from hay and pasture to corn and corn silage feed (DCLWCD, Personal communication, 2013). Per cow milk production has increased and the average watershed dairy cow now produces over 20 % more manure than it did forty years ago. In the context of urbanization and land use change discussed above, this has pushed more cows and nutrients onto a smaller land base, concentrating manure spreading and raising soil P levels on remaining agricultural land (see Bennett et al. 1999; Bennett et al. 2005; Cabot et al. 2004; Carpenter et al. 2006).

The history of agricultural intensification resides in YRW soils. High soil P levels are an important slowly shifting driver of water quality outcomes, even after P input reduction (see Bennett et al. 2005). While there is some evidence of slightly declining watershed soil P levels (Bennett et al. 1999), the dominant trend over the past century has been increasing soil P levels with agricultural intensification. Kara et al. (2012), for example, estimate a 35 % decrease in watershed P budget inputs due primarily to improved agricultural nutrient management, reductions in cattle feed supplements and an urban P fertilizer ban. These estimated reductions, however, are based on optimistic management assumptions, as Kara et al. (2012) acknowledge, and are far outweighed by a century of P input by agricultural intensification; even significant P input reductions may take decades to result in water quality improvement due to biogeochemical time lags in P movement (see Hamilton 2012). This time lag, in combination with a legacy of P input, results in highly spatially variable soil P levels that are difficult to capture in baseline assessments and act as an additional, hidden driver of water quality that frustrate ongoing efforts (see Carpenter 2005; Sharpley et al. 2013).

Agricultural intensification and accompanying soil P levels are highly influenced by state and federal agricultural policy and markets that encourage intensified production and constrain farmers' ability to reduce nutrient pollution. The agricultural system to which these farms belong asks for increasing productivity of field crops and livestock products that do not easily accommodate row crop production reductions or conservative nutrient management practices. The cross-scale nature of this driver (i.e., locally, spatially variable soil P levels, state and federal agricultural policies, and global markets) makes it particularly challenging for watershed-based nutrient management efforts to address. Agricultural BMP's, no matter how innovative and ambitiously implemented, will struggle to meet the challenge (see Harris and Heathwaite 2012). Agricultural intensification is widespread (MacDonald and McBride 2009; Kellogg et al. 2000) and will likely constitute a shifting driver of water quality in other watersheds.

Climate non-stationarity and water quality management

Analyses of precipitation and P loading data describe how climate non-stationarity functions as a shifting driver of water quality outcomes. Since the middle of the twentieth century, YRW annual precipitation has increased at an average rate of 2.1 mm/year from 1930 to 2012 (Fig. 6), a trend consistent throughout the Upper Midwest (Qian et al. 2007; Pryor et al. 2009; Kucharik et al. 2010; Baker et al. 2012). The frequency of YRW heavy rainfall events also increased in recent decades (Fig. 6); this has occurred throughout the north-central USA (Kunkel et al. 2007; Peterson et al. 2008; Villarini et al. 2013). The number of daily rainfall events larger than 50 and 75 mm per decade averaged 9.5 and 1.8, respectively, from 1931 to 1990 and increased to 18 and 6, respectively, from 1991 to 2010. Both trends are consistent with modeling results that point to increasing intensification of the hydrologic cycle under future climate warming (Trenberth 2011).

Increasing annual precipitation and heavy rainfall event frequency in the YRW result in more surface runoff and erosion of phosphorus-laden sediment (see Michalak et al. 2013). As is the case with sediment, P load has a strong, positive relationship with streamflow; as streamflow increases with heavy precipitation events, P loading increases exponentially (Fig. 7). Runoff can be particularly detrimental to water quality in late winter and early spring when cropland, on which manure is often spread, may be bare and frozen. Water quality observations from two major sub-basins draining to Lake Mendota (Yahara River and Pheasant Branch Creek) indicate that 70–80 % of the P load occurs on only 5 % of days, making increased heavy rain event frequency a serious problem for water quality improvement (Carpenter et al. 2014). A positive, nonlinear relationship is also apparent when streamflow is low; droughts result in very low P loads and improvements in lake water quality (Lathrop 2007).

Despite the confounding effect of precipitation on assessing management effectiveness, analyzing changes in the relationship between P load and streamflow through time can potentially reveal the impact of management effort (Lathrop 1998). A reduction in the slope of the relationship indicates that P loads are lower for a given streamflow, regardless of how precipitation and streamflow may have changed (Fig. 7). For both Pheasant Branch Creek and the Yahara River upstream of Lake Mendota, less P is being exported in the later time period for a given streamflow, suggesting modest improvement. Even though total annual P loads do not exhibit a decreasing trend, higher streamflow values are not associated with substantially higher P loads. Despite the lack of absolute P loading reductions, this decrease in slope from the earlier to later period may reflect the impact of BMPs on preventing soil erosion and/or sediment transport downstream.

Persistent failure and changing governance dynamics

In this subsection, we draw on policy documents, scientific and management literature, news media, and qualitative data results from surveys, workshops, and interviews on regional water quality governance to narrate the challenges that shifting drivers pose for environmental governance. As described, those working to improve YRW water quality have been unable to meet the recurring and challenging objective of a 50 % P loading reduction; this has resulted in public frustration and changes in governance dynamics. Public frustration over failed YRW management efforts is not new. In 1988, the Dane County Rivers and Lakes Management Committee (1988: 18) released a report to dispel the myth that "Not much is being done to protect and manage the lakes" (Ibid.: 21). They wrote that because few programs to control watershed pollution are visible to the public, "…combined with a perception that the lake problems are not getting better very quickly, the public is likely to conclude that not much is being done—despite the fact that lake and watershed management activities have been significantly accelerated and strengthened in recent years" (Ibid.: 21). The report also elaborated on the myth that "If we spend enough money and make enough effort, the lakes will become clear and pristine" (Ibid.: 20):

> All of the Yahara River lakes are highly eutrophic, and it is likely, even with aggressive watershed management programs, that they will continue to be eutrophic throughout the foreseeable future. It is realistic to expect that we may be able to achieve reductions of nutrient loadings to the lakes on the order

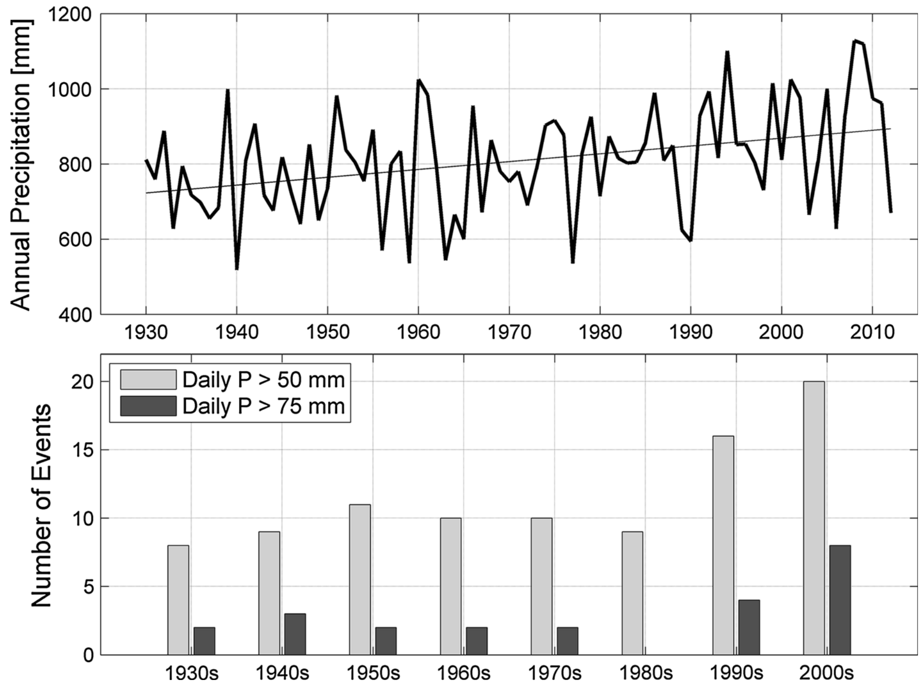

Fig. 6 Annual precipitation and number of large rain events in the Yahara River watershed (1930–2012). Annual precipitation has significantly increased over the twentieth century. Extreme precipitation events with high potential to cause substantial soil erosion and phosphorus loading have also increased, most notably over the last two decades

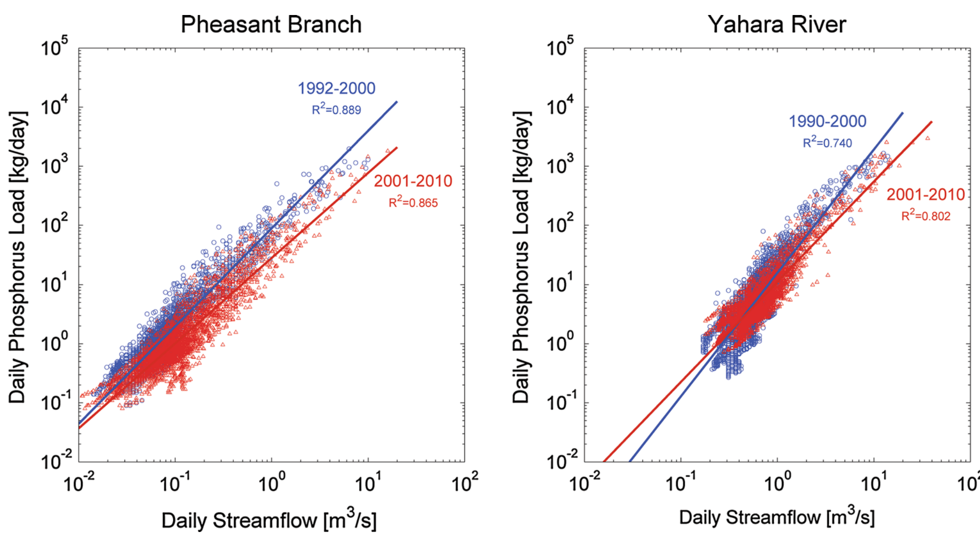

Fig. 7 Changing relationships between streamflow and total phosphorus load in two tributaries to Lake Mendota during the 1990s and 2000s. The decrease in slope between the first and second time periods indicates less P loading occurred during the second time period for a given streamflow. This may indicate that efforts to reduce P loading are effective, even though total P load to the lake is unchanged due to shifting drivers. The slopes in each time period were significantly different ($p < 0.001$) for both tributaries using analysis of covariance

> of 30 percent below current levels, assuming extremely aggressive and well-funded watershed control programs. Reductions beyond this are unrealistic because it is not possible to eliminate all nutrients in runoff from the land surface, and unrealistic to expect that all the land will be treated.

With no improvement in absolute nutrient loading through recent efforts, frustration has persisted and support for reconfigured approaches grown. Assessment of governance efforts as failures can open space for different actors and approaches, whether or not they will be more effective or desirable. Key in the evolving YRW governance landscape are NGO actors and a new watershed-wide partnership; we describe each of these, in turn. The Clean Lakes Alliance (CLA), which acts with a self-described business approach and a fundraising-focused mission, is the most prominent among NGOs. A news editorial from a former Madison mayor described their emergence (Cieslewicz 2013a: 1):

> If it were simple and cheap to clean up Madison's lakes, we would have done it by now. Unfortunately, the answers are complex, uncertain, and costly. But there is a big new wave of activity washing up on the shores of the Yahara lakes, and it is raising hope for relatively rapid and noticeable progress. The [CLA] is a broad coalition of heavy hitters in science, government, environmental organizations, foundations, and business, all committed to cleaning up the lakes in short order. Most striking is the business involvement.

The editorial reported broad support for the organization, as the "most disciplined, comprehensive, strategic, and sustainable approach ever mounted in the history of our lakes" (Goldberg in Cieslewicz 2013a: 1). Local public officials, however, took exception to the lack of credit given them. A follow-up article, prompted by frustrated public officials, quoted one: "[Public officials] take hard votes on increasing taxes and regulation to create programs that actually do yield measurable success. And you get scant credit or thanks. Then, a new organization sweeps in with lots of marketing punch and seems to steal all your thunder" (Cieslewicz 2013b: 1).

These governance dynamics, where assignation of credit and blame are negotiated, are undergirded by the persistent failure described above. This appearance of failure may contribute to governance change, even if the source of failure lies wholly outside of conventional management approaches. CLA has maintained the goal to reduce P loading to the lakes by 50 % and plan to meet it by implementing actions recommended by engineering consultants (Converse 2012; Strand 2013). While the bulk of the proposed interventions were pursued in previous government-led initiatives, two new strategies illustrate the changing nature of watershed nutrient management and shifting drivers' effects potential on governance. First, the group raised the controversial possibility of adding alum (aluminum sulfate) to Lake Mendota tributaries in order to render P inactive and unavailable to feed algal blooms. This could address the problem that heavy rainfall events pose and may be one of the few ways to significantly reduce algal blooms without radically modifying land use or reducing P loading (Strand 2013). This effort represents an attempt to sidestep shifting drivers. Second, CLA is organizing farmers interested in non-regulatory, non-governmental, voluntary approaches to BMP implementation. CLA is developing a voluntary farm certification scheme to acknowledge participating farmers. Unlike government cost-share programs, it would not expose farmers to the possibility of further regulation. Adapting governance by including new business-focused actors and certification efforts implies farmer participation and government regulation of farmers are the key problems, rather than the capacity of BMPs to produce improved outcomes in the context of shifting drivers. More robust explanation of past efforts' failure may have led to different conclusions for adapting governance.

Yahara WINs (watershed improvement network) is a second major watershed water quality improvement effort representing a reconfiguration of watershed governance. Yahara WINs was precipitated by a change in Wisconsin's phosphorus regulations that allows point sources to pay for nonpoint source nutrient pollution reductions rather than complying with stricter P pollution limits through end-of-pipe solutions to comply with a TMDL. In the initial stages of the proposed project, the Madison Metropolitan Sewerage District (the point source organizing nonpoint source pollution reductions) can demonstrate water quality criteria compliance by modeling, not measuring, project outcomes. Measured outcomes would eventually be required, a major difference between this approach and water quality trading. While modeling results may demonstrate a high likelihood of success, model assumptions based on static drivers mean that positive outcomes will be difficult to demonstrate through measurement. Although Yahara WINs includes innovative funding and compliance approaches, the bulk of the BMP management interventions will remain the same. Despite the project's innovative approach and contributions to partnership building, its conventional methods raise questions about its potential to modify a persistent history of failure when success is measured against multiple, shifting drivers of change that make achieving P loading reduction goals increasingly difficult. Changes in governance dynamics are likely in other watersheds challenged by shifting drivers; the extent to which new approaches acknowledge shifting drivers will likely have a significant impact on their success.

Conclusion: learning from the YRW: adapting governance to account for shifting drivers

In summary, despite persistent YRW nutrient management efforts, water quality improvement goals have not been realized. Shifting and non-stationary drivers of change have challenged efforts to achieve and demonstrate success. While goals and baselines typically rely on static measures of watershed conditions, we documented long-term changes in land use, agriculture, and climate that complicate these static baseline-based assessments. As agriculture intensified and dairy production concentrated, soil phosphorus accumulated. As row crop production and urban construction increased, phosphorus-laden soil was exposed. As the YRW urbanized and precipitation patterns became increasingly flashy, runoff events carried more soil and phosphorus to YRW lakes. The drivers we analyzed impact water quality outcomes in urbanizing agricultural watersheds globally (Bennett et al. 2001; Sharpley et al. 2009; Duarte et al. 2009; Harris and Heathwaite 2012; Jarvie et al. 2013). Shifting drivers and persistent failure are common across many environmental governance efforts. For example, shifting drivers of land use, climate, and agricultural change have been documented in Europe (Kronvang et al. 2005; Bechmann et al. 2008; Haygarth et al. 2012) and the United States Mississippi River Basin (NAS 2007), Chesapeake Bay (Sharpley et al. 2009; Reckhow et al. 2011), and the Great Lakes (Joosse and Baker 2011). The YRW case offers suggestions for how future governance might confront shifting drivers that challenge baseline definition, modeling, and monitoring.

Adaptive governance depends on accurate understanding of intervention consequences and causes of failure in order to learn and adapt approaches. When shifting drivers that lie outside of management scope obfuscate intervention effects, it becomes difficult to discern between ineffective managers or management techniques and context-based failure that may require a radically different approach. As discussed above, governance efforts assessed as failures can open space for different actors and approaches, whether or not approaches are new or will be more effective or desirable. This problem exists in many governance efforts beyond water quality.

One solution to the challenge of producing accurate information to inform governance adaptation may lie in improving modeling efforts to capture shifting drivers' contribution to outcomes. Modeled predictions could be improved by incorporating forecasts into model inputs, rather than using snapshot baseline assessments alone (Clark et al. 2001). In the YRW context, this would require improved access to high-resolution data on land use, climate, agriculture, and markets, a need that complements those of cyber infrastructure and data initiatives (NSTC 2013; Wright and Wang 2011). Data, however, may be limited in many contexts and improved forecasts alone, however, are not likely to address underlying problems (Herrick and Sarewitz 2000; Harris and Heathwaite 2012). Better predictions of failure do not improve outcomes when approaches remain largely consistent (e.g., BMPs). Transdisciplinary analyses of drivers of change, such as is presented here, can complement models and monitoring to inform environmental governance design and assessment. Scenario planning and analysis may also be helpful to explore shifting drivers, broaden conversation on desired and possible outcomes, and explore opportunities for more transformative change; such an approach may be relevant in diverse and complex environmental governance problems, especially when uncertainty, the likelihood for conventional thinking, and failure are high (Peterson et al. 2003; Polasky et al. 2011; Carpenter et al. in press).

Goals or measures of success may need to be reconsidered when drivers that lie outside the scope of intervention make achieving outcomes difficult. This may mean recognizing that some determinants of success lie beyond the direct control of managers. Success and failure can be treated as multi-dimensional constructs, where multiple categories of criteria are used and it is recognized that "the determination of relative success or failure is very much a function of the standpoint or perspective from which the evaluation is taking place" (Campbell 2002: 261). This suggests that multiple and different definitions of success may need to be considered and debated when environmental governance efforts consistently fail to meet goals. In the water quality context, agricultural production increases have been achieved, providing low-cost food supplies for some and maintaining agriculture's profitability as a source of capital investment. This success, however, has been accompanied by water quality management failures, among others. Without dramatic changes in agricultural production and phosphorus inputs, water quality goal attainment is unlikely. Directly engaging these difficult conversations may help redefine goals and measures of success in this context, as well as others where environmental governance is frustrated by persistent failure and unacknowledged goal conflict (see Jarvie et al. 2013).

Most challengingly, transformational change would likely be needed to achieve water quality outcomes. This analysis suggests that unless underlying drivers of change are confronted, little progress toward water quality goals should be expected. Shifting drivers have the potential to counteract management interventions, but this should not justify abandoning water quality improvement effort. As noted, controlling for precipitation reveals a slight decline in the relationship between phosphorus loading and streamflow over the last two decades, suggesting YRW management efforts may be somewhat effective, despite being overwhelmed by shifting drivers (Fig. 7). Nonetheless, continuous challenge in meeting absolute nutrient loading reduction goals in many contexts (see Rissman and Carpenter in press; Duarte et al. 2009) may suggest that management and evaluative approaches based on static baseline conditions should change.

Approaches to addressing persistent problem drivers will vary by case, but in general, this may mean a movement toward more transformative approaches to environmental governance that O'Brien (2012: 7), in the context of climate change adaptation, characterizes as a "tension between accommodating change and consciously creating alternatives." The nature of transformative environmental governance would vary by problem issue. In the water quality context, transformative approaches could be considered to address heavy rain events and dramatic fluctuations in water quantity that can drive outcomes. These might include widespread land cover change to reduce erosion, such as through the restoration of native landscapes (e.g., Santelmann et al. 2004), or engineered ecosystem development to attenuate floods and retain water for supplemental irrigation (e.g., Baker et al. 2012). Engineered, biochemical approaches to reducing algal blooms (e.g., alum) without altering upstream land use are also under consideration, although concerns about viability, cost, and unintended consequences remain. Treating nutrients as assets rather than wastes may become more viable in the context of "peak phosphorus" (Cordell et al. 2009). Modifications to the historical pattern of agricultural intensification may also be required if nutrient loading targets are to be reached. Wisconsin, for example, has had success with managed intensive rotational grazing (Kriegl and McNair 2005), which may offer opportunity to reduce exposed soil, maintain agricultural land and farm viability, and reduce climate-related vulnerabilities of agricultural systems. This would require a renegotiation of agriculture's goals and definitions of success and may also entail focusing water quality improvement effort outside of the watershed, such as toward federal energy, farm, and food policy.

In summary, lessons about policy interventions and assessment from the YRW suggest that to enable learning and adaptive governance, scenario planning and transdisciplinary analyses of drivers of change can complement models and monitoring to better inform environmental governance. Future efforts may need to engage with the challenging task of confronting drivers through innovative, adaptive, and transformative intervention. When shifting drivers remain beyond the scope of management, goals and measures of success may need to be renegotiated.

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