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Review

Assessing and addressing the re-eutrophication of Lake Erie: Central basin hypoxia



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ABSTRACT

Relieving phosphorus loading is a key management tool for controlling Lake Erie eutrophication. During the 1960s and 1970s, increased phosphorus inputs degraded water quality and reduced central basin hypolimnetic oxygen levels which, in turn, eliminated thermal habitat vital to cold-water organisms and contributed to the extirpation of important benthic macroinvertebrate prey species for fishes. In response to load reductions initiated in 1972, Lake Erie responded quickly with reduced water-column phosphorus concentrations, phytoplankton biomass, and bottom-water hypoxia (dissolved oxygen <2 mg/l). Since the mid-1990s, cyanobacteria blooms increased and extensive hypoxia and benthic algae returned. We synthesize recent research leading to guidance for addressing this re-eutrophication, with particular emphasis on central basin hypoxia. We document recent trends in key eutrophication-related properties, assess their likely ecological impacts, and develop load response curves to guide revised hypoxia-based loading targets called for in the 2012 Great Lakes Water Quality Agreement. Reducing central basin hypoxic area to levels observed in the early 1990s (ca. 2000 km²) requires cutting total phosphorus loads by 46% from the 2003–2011 average or reducing dissolved reactive phosphorus loads by 78% from the 2005–2011 average. Reductions to these levels are also protective of fish habitat. We provide potential approaches for achieving those new loading targets, and suggest that recent load reduction recommendations focused on western basin cyanobacteria blooms may not be sufficient to reduce central basin hypoxia to 2000 km².

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Contents

Introduction	227
Phosphorus loading trends	227
Total phosphorus loading	227
Dissolved reactive phosphorus	228
Water quality trends	228
Phytoplankton biomass	228
Dissolved oxygen (DO)	229
Impacts of hypoxia on the Lake Erie fish community	230
Modeling impacts of hypoxia on Lake Erie fishes	232
A new look at P loading targets	233
Exploring loading targets for water quality	233
Potential loading targets for fishes	234
Approaches to meet new targets	234
Spatial distributions of loading sources	235
Agricultural BMPs	236
Focus on management of DRP	236
Evaluating watershed-scale effectiveness of traditional agricultural BMPs	236
Climate change implications	237
Watershed impacts	237
Hypoxia formation impacts	238
Fish impacts	238
Climate impacts on BMP effectiveness	240
Implications for policy and management action	240
Acknowledgments	243
References	243

Introduction

Several anthropogenic stressors have impacted Lake Erie since European settlement. However, phosphorus (P) loading has been particularly influential (Ludsin et al., 2001). During the 1960s and 1970s, increased P inputs degraded water quality and reduced hypolimnetic oxygen levels (Bertram, 1993; Makarewicz and Bertram, 1991; Rosa and Burns, 1987). Reduced oxygen, in turn, eliminated thermal habitat vital to cold-water organisms in the central basin (CB) (Hartman, 1972; Laws, 1981; Leach and Nepszy, 1976; Ludsin et al., 2001) and contributed to the local extirpation of important benthic macroinvertebrates and declines of several fish species (Britt, 1955; Carr and Hiltunen, 1965; Ludsin et al., 2001). This development and control of freshwater eutrophication by phosphorus loads is ubiquitous and well documented (e.g., Schindler, 2006, 2012; Smith and Schindler, 2009).

In response, P abatement programs were initiated in 1972 as part of the Great Lakes Water Quality Agreement (GLWQA) (DePinto et al., 1986a). Lake Erie responded relatively quickly, as indicated by measurable decreases in total phosphorus (TP) loads (Dolan, 1993), water-column TP concentrations (DePinto et al., 1986a; Ludsin et al., 2001), phytoplankton biomass (especially cyanobacteria; Bertram, 1993; Makarewicz et al., 1989), and bottom-water hypoxia (dissolved oxygen <2 mg/l) (Bertram, 1993; Charlton et al., 1993; Makarewicz and Bertram, 1991), as well as by recovery of several ecologically and economically important fishes (Ludsin et al., 2001). Although P abatement was primarily responsible for improving water quality through the mid-1980s, zebra (*Dreissena polymorpha*) and quagga (*D. rostriformis bugensis*) mussel invasions during the late 1980s and early 1990s, respectively, likely magnified these changes (Holland et al., 1995; MacIsaac et al., 1992; Nicholls and Hopkins, 1993) and might have contributed to the recovery of some benthic macroinvertebrate taxa (Botts et al., 1996; Pillsbury et al., 2002; Ricciardi et al., 1997). Since the mid-1990s, however, Lake Erie appears to be returning to a more eutrophic state (EPA, 2010; Murphy et al., 2003), as indicated by increases in cyanobacteria (e.g., *Microcystis* spp., *Lyngbya wollei*; Bridgeman et al., 2012; Michalak et al., 2013; Stumpf et al., 2012), the resurgence of extensive benthic algae growth (particularly *Cladophora* in the eastern basin)

(Depew et al., 2011; Higgins et al., 2008; Stewart and Lowe, 2008), and the return of extensive CB hypoxia (Burns et al., 2005; Hawley et al., 2006; Rucinski et al., 2010; Zhou et al., 2013).

In 2005, EcoFore-Lake Erie – a multi-year, multi-institutional project supported by the National Oceanic and Atmospheric Administration – began with the goal of developing a suite of management-directed models useful for exploring causes of changes in P loading, their impacts on CB hypoxia, and how these changes might influence Lake Erie's highly valued recreational and commercial fisheries. The EcoFore-Lake Erie project focused on CB hypoxia because of uncertainty about the mechanisms underlying its return to levels commensurate with the height of eutrophication during the mid-20th century (Hawley et al., 2006) and because of its great potential to harm Lake Erie's valued fisheries (sensu Ludsin et al., 2001).

Herein, we provide a synthesis of the results from those efforts, as well as work undertaken through other related projects, leading to science-based guidance for addressing the re-eutrophication of Lake Erie and in particular, CB hypoxia. In the following sections, we document recent trends in key eutrophication-related properties and assess their likely ecological impacts. We develop P load response curves to guide revision of hypoxia-based loading targets, consistent with the 2012 Great Lakes Water Quality Agreement (GLWQA, IJC 2013), and provide potential approaches for achieving the revised loading targets.

Phosphorus loading trends

Total phosphorus loading

Total P loading into Lake Erie has changed dramatically through time, with temporal trends driven in large part by implementing P abatement programs as part of the GLWQA and inter-annual differences responding to variable meteorology (Dolan, 1993). Following initial implementation of nutrient abatement programs beginning in 1972, TP inputs declined precipitously, reaching the GLWQA target loading level of 11,000 MTA during the 1980s (Fig. 1; see Dolan and Chapra, 2012 for methods). Since then, loading has remained below the

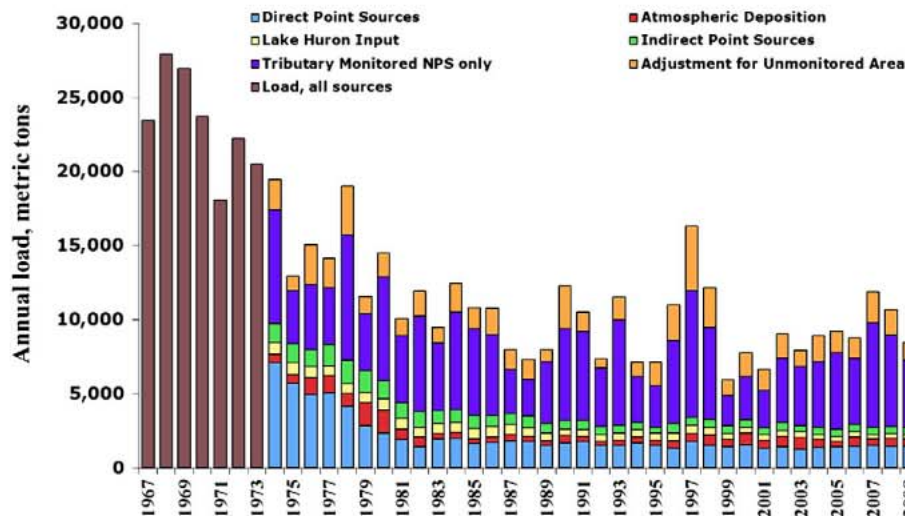


Fig. 1. Total phosphorus loads (TP) into Lake Erie during 1967–2011 from municipal and industrial point sources, monitored and estimated non-point sources (NPS), atmospheric deposition, and inter-lake transfers. Sources of TP loads: Dolan (1993); Dolan and McGunagle (2005), Dolan and Chapra (2012), and D. Dolan, unpublished data. Current GLWQA loading goal is 11,000 metric tons per year.

GLWQA target in most years. The initial declines were due primarily to programs that reduced point sources of P (e.g., P restrictions in commercial detergents, enhancements of sewage treatment plants), leaving non-point sources as dominant (Table 1, Fig. 1) (Dolan, 1993; Richards et al., 2001, 2010).

Dissolved reactive phosphorus

The earlier GLWQA (IJC, 1978) focused on TP as a key water quality parameter by which Lake Erie eutrophication could be measured (DePinto et al., 1986a). However, recent focus has turned to dissolved reactive phosphorus (DRP) (Richards, 2006; Richards et al., 2010) because this form of P is more highly bioavailable (DePinto et al., 1981, 1986b, 1986c) to nuisance algae (e.g., *Cladophora*) and cyanobacteria (e.g., *Microcystis* spp.). Moreover, DRP loads from several Lake Erie tributaries (e.g., Maumee River, Sandusky River, Honey Creek, and Rock Creek) have increased dramatically since the mid-1990s (Fig. 2, Richards et al., 2010). Increases in DRP loading are in contrast to the relatively constant TP loads from those same watersheds. As a result, the portion of TP that is DRP more than doubled from a mean of 11% in the 1990s to 24% in the 2000s.

To help understand this increase in the proportion of TP as DRP in non-point sources, Han et al. (2012) calculated net anthropogenic P inputs (NAPI) to 18 Lake Erie watersheds for agricultural census years from 1935 to 2007. NAPI quantifies anthropogenic inputs of P from fertilizers, the atmosphere, and detergents, as well as the net exchange in P related to trade in food and feed. During this 70-year period, NAPI increased through the 1970s and then declined through 2007 to a level last experienced in 1935. This pattern was the result of (1) a dramatic increase in fertilizer use, which peaked in the 1970s, followed

by a decline to about two-thirds of maximum values; and (2) a steady increase in P exported in the form of crops destined for animal feed and energy production (Han et al., 2012). The decline in fertilizer and manure application between 1975 and 1995 overlapped with increased efforts to reduce sediment and particulate P loading by controlling erosion through no-till and reduced-till practices. In particular, these tillage changes occurred in the Maumee and Sandusky River watersheds mostly during the early 1990s (Richards et al., 2002; Sharpley et al., 2012).

During 1974–2007, individual riverine TP loads fluctuated (e.g., Fig. 2), and were correlated with variations in water discharge. However, riverine TP export did not show consistent temporal trends, and did not correlate well with temporal trends in NAPI or fertilizer use. Interestingly, the fraction of watershed TP inputs exported by rivers (Han et al., 2012) increased sharply after the 1990s, possibly because of changing agricultural practices. Farm practices also may be responsible for the increasing fraction of TP exported as DRP, which appears to have been exacerbated by increases in extreme rainfall-runoff events over the last 10 years (Daloğlu et al., 2012; Sharpley et al., 2012).

Daloğlu et al. (2012) used the Soil and Water Assessment Tool (SWAT) watershed model to explore these potential contributions to the increase in DRP. The SWAT results suggest increased DRP export was driven by increasing storm events, changes in fertilizer application timing and rate, and management practices that increase P-stratification of the soil surface. The frequency of extreme rain events has increased since the early 1900s in this region, as has the number of prolonged wet periods (Karl et al., 1998; Mortsch et al., 2000). However, weather might not be the only source of this change. For example, Daloğlu et al. (2012) also demonstrated that while the current more extreme storms appeared to stimulate large fluxes of DRP, those same weather patterns imposed on agricultural landscapes of the 1970s did not.

Water quality trends

Phytoplankton biomass

The observed increases in DRP loading rates are important because they may underlie increases in phytoplankton biomass in the western basin (WB) and CB in recent decades, including potentially inedible and toxic cyanobacteria such as *Microcystis* (Bridgeman et al., 2012; Michalak et al., 2013; Ohio EPA, 2010; Stumpf et al., 2012). Phytoplankton biomass in both the WB and CB decreased between the 1970s and the mid-1980s, and then increased between 1995 and 2011 due to high abundance of cyanobacteria, predominantly *Microcystis* spp.

Table 1
Distribution of total phosphorus loads among major source categories to Lake Erie (Dolan and Chapra, 2012).

2003–2011 Average total Lake Erie loads (metric tons per year)	
Non-point inputs to Lake Erie	6183
All point sources inputs	1884
Atmospheric inputs	525
Inputs from upstream Lake Huron	336
Total	8929

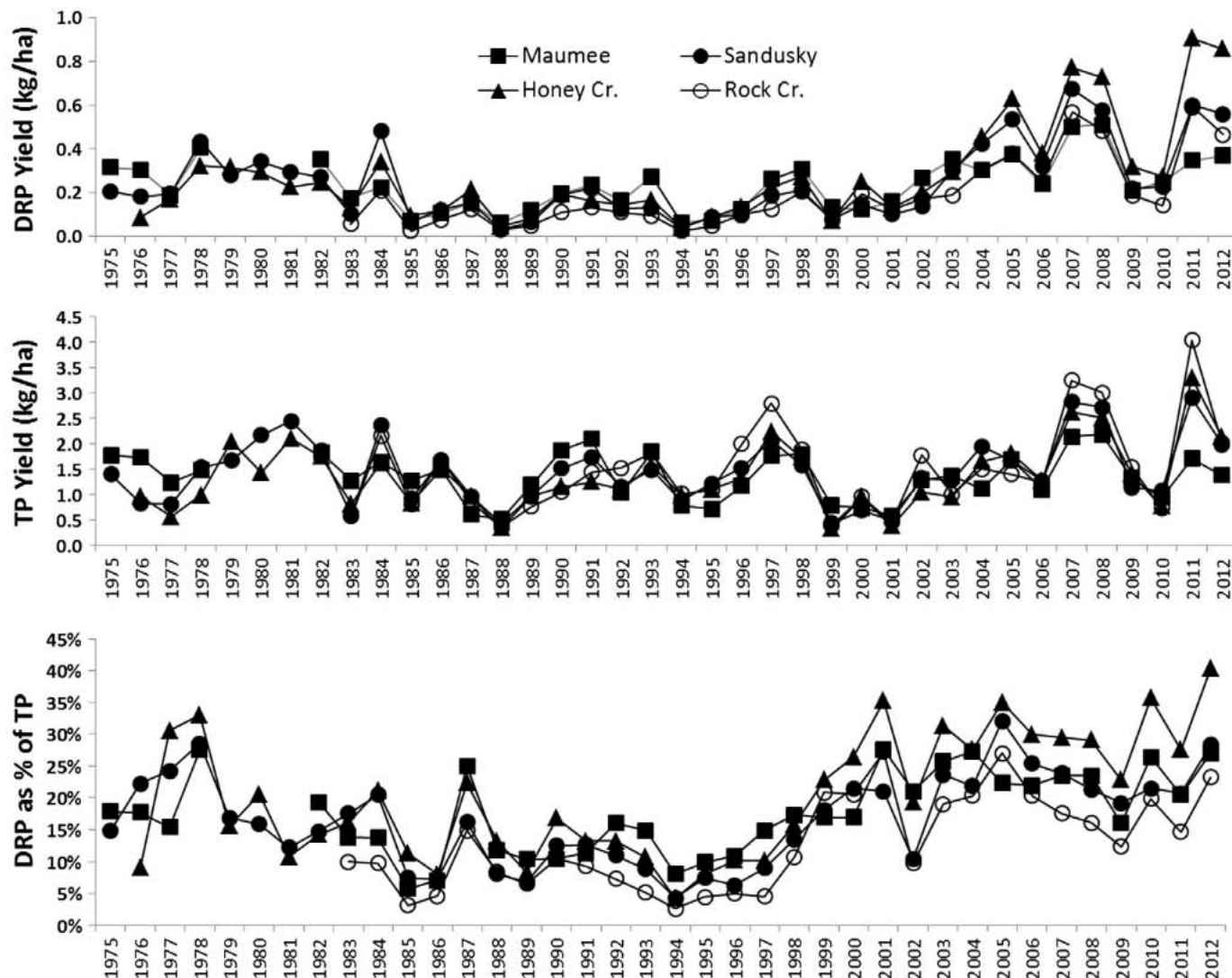


Fig. 2. Yields of total phosphorus (TP) and dissolved reactive phosphorus (DRP), as well as DRP yield as a % of TP, from four agricultural watersheds in the west and central basins of Lake Erie, 1976–2012. Source: Richards et al. (2010); R.P. Richards (unpublished).

(Fig. 3). TP concentrations in the CB increased and water transparency in the WB decreased during this same time period (Fig. 4). CB spring surface chlorophyll *a* (CHL) concentration increased from ~3 µg/l in 1985–2000 to >19 µg/l in 2007, even though TP loads remained relatively constant, doubling the CHL:TP ratio during this time period (Fig. 5).

Dissolved oxygen (DO)

Sedimentation of algae and fecal material drives DO depletion in the hypolimnion of lakes by stimulating bacterial respiration. Correspondingly, ecosystems undergoing eutrophication often demonstrate increases in the magnitude, frequency, and duration of hypolimnetic hypoxia (Diaz and Rosenberg, 2008; Hagy et al., 2004; Rabalais et al., 2002; Scavia et al., 2004, 2006). In the case of Lake Erie, we would expect its largest basin, the CB, to be most prone to hypolimnetic hypoxia because it is deep enough to stratify but shallow enough that the thermocline sets up relatively close to the lake bottom, reducing the hypolimnion thickness (Charlton, 1980; Rosa and Burns, 1987). One of the important mechanisms producing a deeper thermocline (and thinner hypolimnion) is Ekman pumping due to the anticyclonic winds (Beletsky et al., 2012, 2013). By contrast, the hypolimnetic volume of the Eastern Basin (EB) is too large to be substantially depleted of

DO before fall turnover, and the shallowness of the WB causes its water column to remain mixed most of the time (Bridgeman et al., 2006).

While some CB hypolimnetic hypoxia is likely natural (Delorme, 1982), human activities during the second half of the 20th century exacerbated the rate and extent of DO depletion (Bertram, 1993; Burns et al., 2005; Rosa and Burns, 1987; Rucinski et al., 2010). P inputs stimulated algal production; with subsequent algal settlement and decomposition, DO depletion rates increased during the mid-1990s with corresponding hypoxic areas as large as 11,000 km² (Beeton, 1963). Average hypolimnion DO concentrations in August–September for CB stations with an average depth greater than 20 m increased from less than 2 mg/l in 1987 to over 6 mg/l in 1996, followed by an abrupt decrease to below 3 mg/l in 1998 with concentrations remaining low and quite variable through 2011, the most recent year for which data are available (Fig. 6). Zhou et al. (2013) used geostatistical kriging and Monte Carlo-based conditional realizations to quantify the areal extent of summer CB hypoxia for 1987 through 2007 and develop a probabilistic representation of hypoxia extent. While substantial intra-annual variability exists, hypoxic area was generally smallest during the mid-1990s, with larger extents during the late 1980s and the early 2000s (Fig. 7).

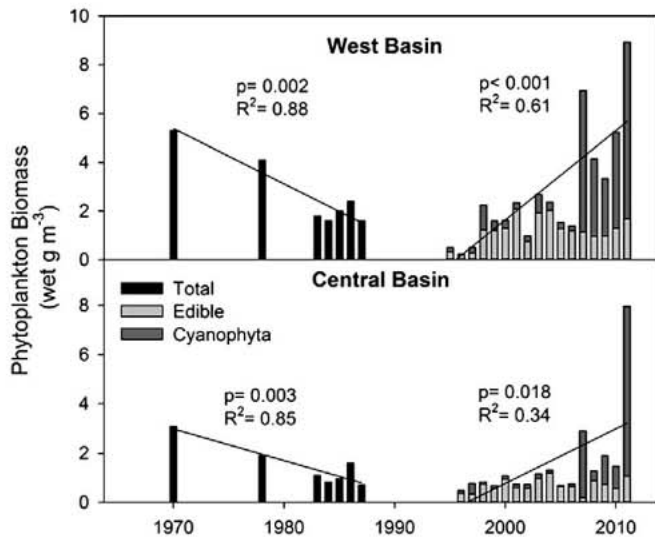


Fig. 3. Annual May–September mean phytoplankton biomass in Lake Erie's west (WB) and central (CB) basins, 1970–2011. Historical data (1970–1986) bars are the arithmetic mean of total phytoplankton biomass (Makarewicz, 1993); whereas, those from 1995–2011 are the geometric means of edible species and Cyanophyta. Probability (p) and R^2 represent regression of biomass across time for the periods shown (Data source: R. Briland and Ohio Division of Wildlife, unpublished). Note that the slope for 1995–2011 is strongly influenced by 2011.

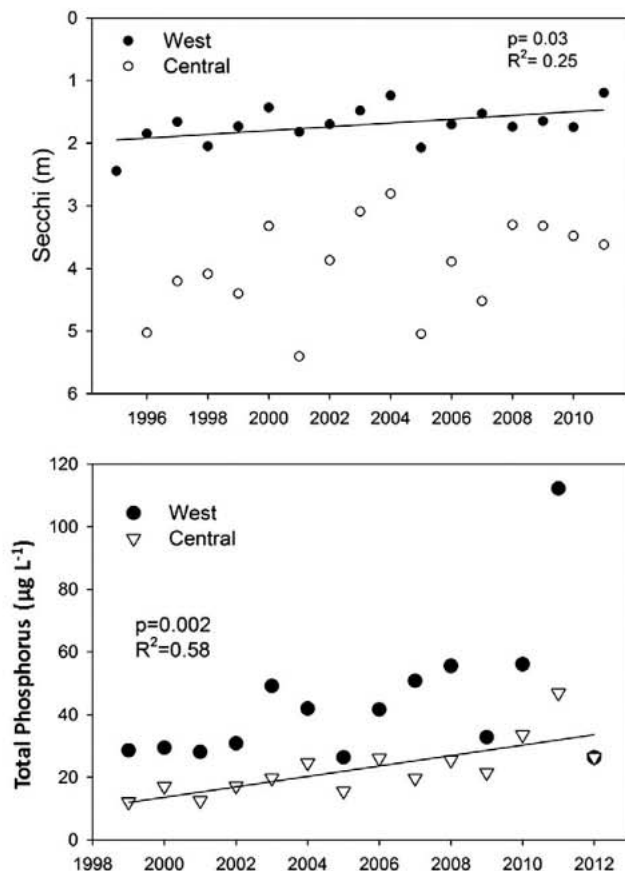


Fig. 4. (Upper panel) Annual summer (June–August) mean water transparency in Lake Erie's west basin and central basin, 1995–2011. Probability (p) and R^2 represent regression of Secchi disk transparency across time. (Lower panel) Annual (May–September) mean total phosphorus (TP) concentration in 1995–2012. Probability (p) and R^2 represent regression of TP across time. Regression lines in both panels are only shown for significant ($p < 0.05$) trends (Ecofore-Lake Erie Forage Task Group Report, unpublished data).

The increase in hypolimnetic DO from the 1980s to mid-1990s and the subsequent decline during the late 1990s and 2000s (Fig. 6) are consistent with trends in the DO depletion rate. Based on a simple DO model, driven by a one-dimensional hydrodynamic model (Beletsky and Schwab, 2001; Chen et al., 2002), Rucinski et al. (2010) demonstrated that the change in DO depletion rates reflected changes in TP loads, not climate, between 1987 and 2005. Similarly, Burns et al. (2005) showed that the depletion rate is related to the previous year's annual TP load.

Impacts of hypoxia on the Lake Erie fish community

Several ecological processes that are influenced by hypoxia have the potential to negatively affect individual fish growth, survival, reproductive success and, ultimately, population growth (e.g., Breitburg, 2002; Coutant, 1985; Ludsin et al., 2009; Wu, 2009). Rapid changes in oxygen concentrations may trap fish in hypoxic waters and lead to direct mortality. In fact, there is recent evidence of such events in nearshore Lake Erie, whereby wind-driven mass movement of hypoxic waters into nearshore zones appears to have led to localized fish mortalities (J. Casselman, Queen's University personal communication). While such direct mortality due to low DO is possible, a more common immediate fish response to hypolimnetic hypoxia is avoidance of bottom waters. Such behavioral responses can lead to shifts away from preferred diets (e.g., Pihl, 1994; Pihl et al., 1992), increased total metabolic costs and potential reproductive impacts by occupying warmer waters and undertaking long migrations (e.g., Craig and Crowder, 2005; Taylor et al., 2007), and enhanced compensatory density-dependent effects through vertical and horizontal compression (e.g., Eby and Crowder, 2002). However, documenting these effects on fish growth, survival, and significant, long-term population-level responses has proven difficult. Bottom hypoxia in many north temperate systems, such as Lake Erie, persists for a short time period (days to months; Rucinski et al., 2010), making hypoxia effects on fish difficult to distinguish from other seasonal processes. In addition, while nutrient additions can exacerbate hypoxia, they can also increase system productivity and increase prey production through bottom-up processes. Such positive effects can be particularly strong if bottom hypoxia forces prey organisms higher in the water column where many zooplankton taxa have higher growth rates because of higher temperature, light, and phytoplankton abundance (e.g., Goto et al., 2012).

While definitive *in situ* ecological impacts have been hard to quantify, laboratory studies have demonstrated the potential for some Lake Erie fish and zooplankton to be negatively affected by direct exposure to low DO concentrations. For example, while the relatively tolerant yellow perch (*Perca flavescens*) can survive at low DO concentrations, both consumption and growth rates decline under hypoxia (Roberts et al., 2011). Further, hypoxia may lead to reduced prey production because some zooplankton prey species experience poor survival under hypoxia (e.g., *Daphnia mendotae*; Goto et al., 2012). In contrast, other zooplankton taxa seem to be able to survive prolonged hypoxia (see Vanderploeg et al., 2009a), but may use the hypoxic zone as a refuge from predation. Additionally, the growth and survival rates of some preferred benthic prey (e.g., Chironomidae) are largely unaffected by low DO conditions (Armitage et al., 1995).

Potential *in situ* impacts of hypoxia on mobile fish species in Lake Erie appear to be indirect and vary among species. For example, hypoxia-intolerant rainbow smelt (*Osmerus mordax*) entirely avoid hypoxic waters in CB by migrating horizontally or moving up into a thin layer of the water column just above the hypoxic zone (Pothoven et al., 2012; Vanderploeg et al., 2009b). By contrast, while some yellow perch move horizontally away from the CB hypoxic region, many remain in this region, but move higher in the water column, and undertake short feeding forays into the hypoxic zone (Roberts et al., 2009, 2012). Owing to these taxon-specific responses, hypoxia may reduce the overlap between predator and prey or facilitate predator foraging

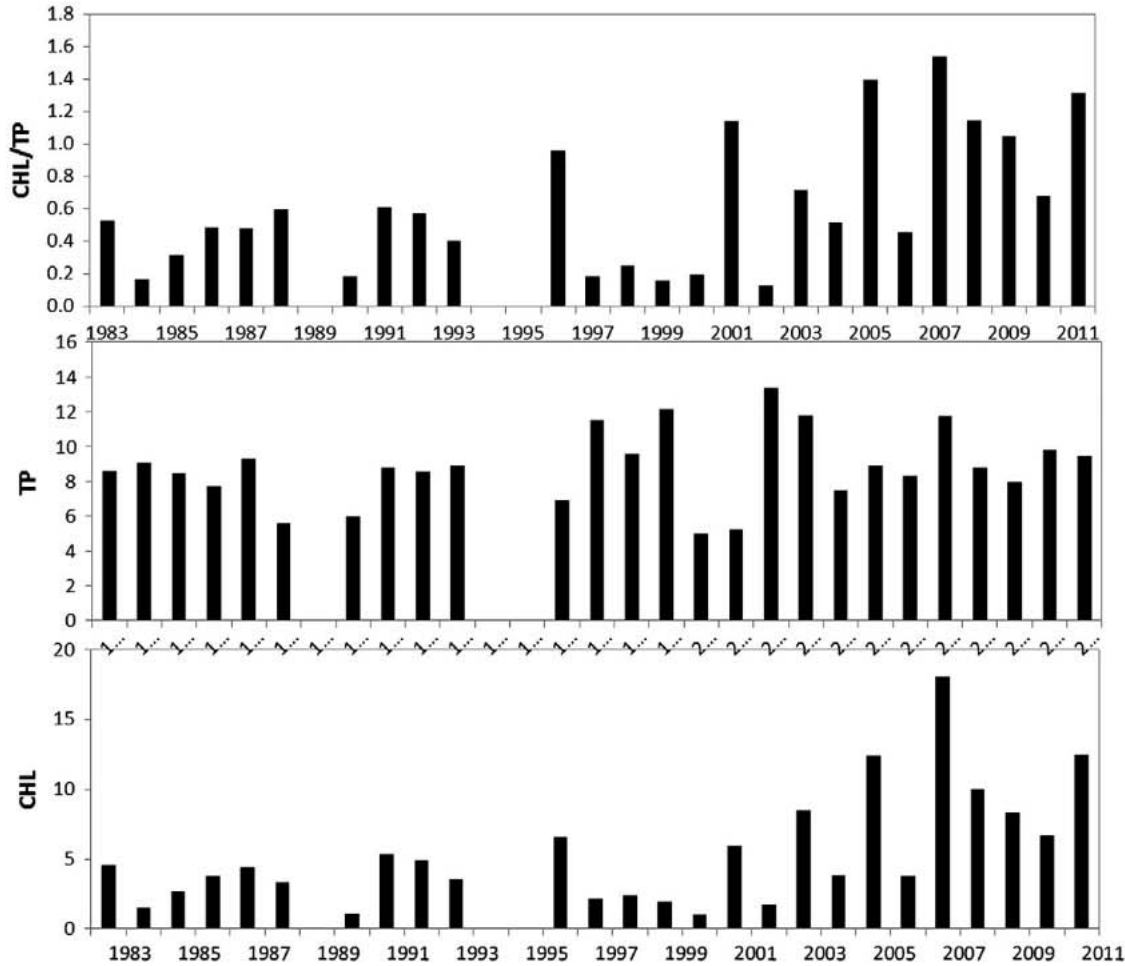


Fig. 5. Mean spring chlorophyll (CHL, µg/l) and total phosphorus (TP, µg/l) concentrations and the CHL:TP ratio for central basin stations between 1983 and 2012. Data obtained from EPA GLENDa website (EPA 2013). Apparent zero values in 1988 and 1993–1994 actually represent years when no data were reported.

success, as both prey and predator are squeezed into the same area of the water column. In Lake Erie, the diets of emerald shiner, a warm-water epilimnetic zooplanktivore, seemed unaffected by hypoxia (Pothoven et al., 2009) and their foraging rates may even be increased as zooplankton are forced into the epilimnion. By contrast, intolerant, cold-water rainbow smelt displayed strong selection for Chironomidae pupae and larvae during oxygenated periods, but consumed almost

entirely zooplankton during hypoxia (Pothoven et al., 2009). More tolerant fish species, such as white perch (*Morone americana*) and yellow perch also altered their diets to consume more zooplankton in response to hypoxia, but these shifts were more subtle (Roberts et al., 2009, 2012). Finally, these species-specific distributional and foraging responses to hypoxia are generally supported by seasonal trends in fish condition in CB. While condition of emerald shiner improved from

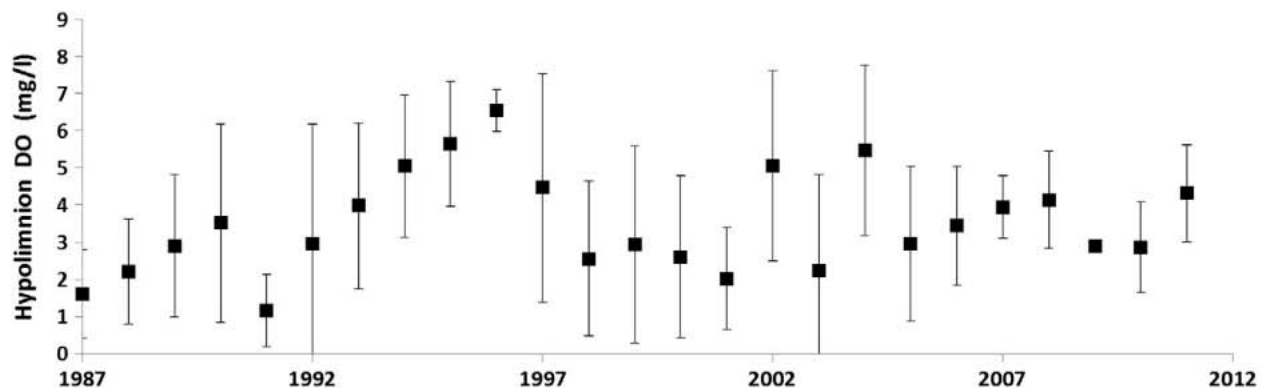


Fig. 6. Mean $\pm$ 1 standard deviation of August–September mean hypolimnetic dissolved oxygen concentrations for central basin stations greater than 20 m depth compiled from the Great Lakes National Program Office (GLNPO), Environment Canada: Water Science & Technology Branch (S. Watson pers. comm.), and the International Field Years on Lake Erie Program (S. Ludsin and T. Johengen, unpublished data). Numbers of samples and sampling dates differ from year to year.

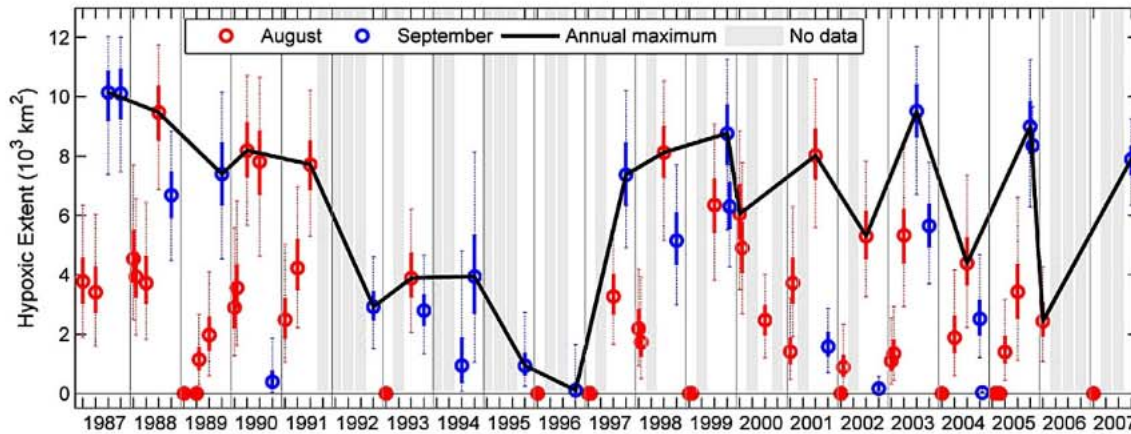


Fig. 7. Estimated areal extent of central basin hypoxia developed through universal kriging and conditional realizations of bottom-water DO. The up to four sampling periods for each year are defined as measurements taken in the following date ranges: August 1–12, August 13–22, August 23–September 5, and September 9–26. Solid circles on the x-axis represent cruises where no DO values were reported below 2 mg/l. Source: Zhou et al. (2013). Solid line connects maximum values for each year.

summer into fall, rainbow smelt condition declined during hypoxia (Ludsin et al. unpublished). Condition of tolerant yellow perch in Lake Erie did not decrease during the height of hypoxia (Roberts et al., 2009) and yellow perch RNA:DNA ratios (an index of short-term condition) did not reveal a strong negative response to hypoxia (Roberts et al., 2011).

Modeling impacts of hypoxia on Lake Erie fishes

While empirical evidence points to a variety of taxon-specific negative and positive effects of hypoxia on fish feeding, growth, and production in Lake Erie, the magnitude of such potential effects and their population-level consequences remain open questions. Through the Ecofore-Lake Erie program, we have explored such effects through a variety of models. Given the variety of pathways through which hypoxia may affect fish vital rates, models differ in their relative emphasis on diverse processes. The simplest and most straightforward approach has consisted of developing statistical relationships between measures of hypoxia and fish population metrics at the lake-basin scale. For example, we found a significant negative relationship between the number

of modelled hypoxic ($\text{DO} \leq 2 \text{ mg/l}$) days and the condition (relative-weight based) of both mature (2+) female and male yellow perch captured in the CB during fall (September–October) 1990–2005 (Fig. 8), suggesting that observed distributional and foraging responses at hypoxic CB sites during summer (Roberts et al., 2011) may have population-level impacts.

Brandt et al. (2011) and Arend et al. (2011) modeled growth rate potential (GRP) of selected fishes in the CB as a surrogate for fish habitat quality. Brandt et al. (2011) argued that hypoxia had a temporary positive effect on walleye (*Sander vitreus*) GRP as prey fish were forced into areas where temperature, DO, and light conditions were favorable for efficient walleye foraging and growth. In contrast, Arend et al. (2011) found that GRP of yellow perch, rainbow smelt, emerald shiner, and round Goby (*Neogobius melanostomus*) improved with reductions in P loading and hypoxia prior to the mid-1990s, but did not continue to improve from the mid-1990s through 2005 (and may even have decreased). Arend et al. (2011) also showed that hypoxia impacts were most severe for adult stages of non-native species, including cold-water rainbow smelt and round Goby, a benthic species that typically forages on the lake bottom. Hypoxia's impacts were least severe for

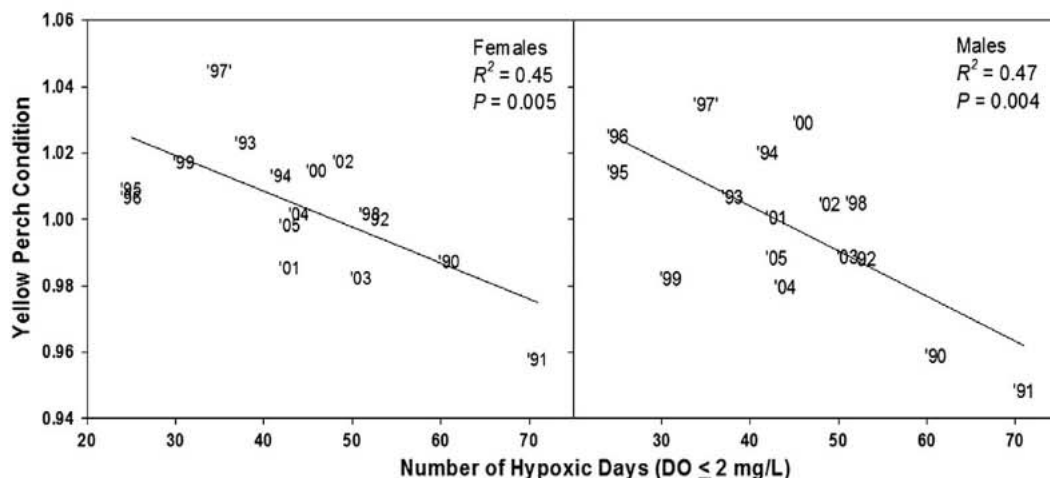


Fig. 8. Relationship between the number of hypoxic days in the central basin of Lake Erie and the condition (relative weight) of yellow perch captured in central basin bottom trawls during fall (September–October), 1990–2005. Condition was defined as the mean relative weight, i.e. observed mass divided by predicted mass, which was estimated from a sex-specific length–mass relationship developed for Lake Erie yellow perch during this time period. Condition values greater than or less than one signifying above-average or below-average condition, respectively. Data sources: hypoxic days (Rucinski et al., 2010); fish condition (Troy Farmer and the Ohio Division of Wildlife, unpublished data).

adult and juvenile stages of yellow perch, a species that is native to Lake Erie, and hence, may have evolved with hypoxia (sensu Delorme, 1982).

While a GRP modeling approach offers a more mechanistic means than linear regression to estimate target nutrient loads, this approach is static, and hence, cannot account for the likely feedbacks and indirect effects that might exist as temperature and hypoxia vary through space and time. For example, behavioral avoidance of hypoxia has been shown to lead to highly dynamic predator–prey interactions and density-dependent growth, and these changes in predator–prey interactions can cascade to not only affect a single predator–prey pair, but also the entire food web. Thus, we also have been exploring the effects of hypoxia and other habitat attributes (e.g., temperature, prey availability) on fish using more dynamic approaches, such as individual- and population-based bioenergetics simulations (individual-based modeling; D. Goto, personal communication), fish population behavior (patch-choice modeling; K. Pangle, personal communication), trophic interactions (Ecopath with Ecosim; e.g. Langseth et al., 2012), and comprehensive ecosystem responses (Comprehensive Aquatic Systems Modeling, CASM; e.g. Bartell, 2003). These modeling approaches differ greatly in their spatial and temporal resolution and focus on the entire foodweb versus a subset of abundant, representative species. The differential emphasis on behaviorally mediated habitat selection, trophic interactions and trophic cascades among these models may lead to somewhat dissimilar predictions regarding ecological effects of hypoxia in Lake Erie. The integration of output from these diverse modeling approaches collectively provide a suite of plausible forecasts, as well as by help to identify key uncertainties that can guide future monitoring and research decisions.

A new look at P loading targets

Because of increases in hypoxia since the mid-1990s and because other eutrophication symptoms and potential impacts have become stronger since then, consideration of new phosphorus loading targets seems warranted. The use of models to assist in developing nutrient loading targets for the Great Lakes has a long history. Bierman (1980) reviewed their use as part of the negotiation of the earlier GLWQA, at which time five models were used to develop P loading objectives. The models ranged from simple, empirical correlations to complex mechanistic models (Bierman and Dolan, 1976; Bierman et al., 1980; Chapra, 1977; DiToro and Connolly, 1980; DiToro and Matystik, 1980; Hydroscience, 1976; Thomann et al., 1975, 1976; Vollenweider, 1977). Since that time, a variety of biogeochemical models have been developed to understand ecological interactions within Lake Erie and other Great Lakes. While some models were constructed during the 1980s (e.g., DePinto et al., 1986c; Di Toro et al., 1987; Lam et al., 1987a, 1987b; Scavia, 1980; Scavia and Bennett, 1980; Scavia et al., 1981a, 1981b; 1988), a new generation of models has emerged more recently (e.g., Bierman et al., 2005; Fishman et al., 2009; Leon et al., 2011; LimnoTech, 2010; Rucinski et al., 2010, 2014; Zhang et al., 2008; 2009).

For Lake Erie, Zhang et al. (2008) developed a two-dimensional ecological model to explore potentially important ecosystem processes and the contribution of internal vs. external P loads. Rucinski et al. (2010) developed a one-dimensional model to examine the inter-annual variability in DO dynamics and evaluate the relative roles of climate and P loading. Leon et al. (2011) developed a three-dimensional model to capture the temporal and spatial variability of phytoplankton and nutrients. LimnoTech (2010) developed a fine-scale linked hydrodynamic, sediment transport, advanced eutrophication model for the WB that relates nutrient, sediment, and phytoplankton temporal and spatial profiles to external loads and forcing functions. Stumpf et al. (2012) developed a model to predict the likelihood of cyanobacteria blooms as a function of average discharge of the Maumee River.

As part of EcoFore-Lake Erie, Rucinski et al. (2014) developed and tested a model specifically for establishing the relationship between P loads and CB hypoxia. This model is driven by a one-dimensional

hydrodynamic model that provides temperature and vertical mixing profiles as described in Rucinski et al. (2010). The Ekman pumping effect described above and in Beletsky et al. (2012, 2013) was in essence parameterized as additional diffusion in the one-dimensional hydrodynamic model. The biological portion of the model is a standard eutrophication model that used constant sediment oxygen demand (SOD) of $0.75 \text{ gO}_2 \text{ m}^{-2} \cdot \text{d}^{-1}$ because it has not varied significantly over the analysis period (Matisoff and Neeson, 2005; Schloesser et al., 2005; Snodgrass, 1987; Snodgrass and Fay, 1987). Earlier analysis (Rucinski et al., 2010) indicated that SOD represented on average 63% of the total hypolimnetic oxygen demand, somewhat larger than the 51% and 53% contribution that Bouffard et al. (2013) measured in 2008 and 2009, respectively. However, for load-reduction scenarios, a new formulation was needed to adjust SOD as a function of TP load. This relationship (Rucinski et al., 2014), while ignoring the 1-year time lag suggested by Burns et al. (2005), was based on an empirical relationship between SOD and deposited organic carbon (Borsuk et al., 2001).

The model was calibrated over 19 years (1987–2005) using chlorophyll *a*, zooplankton abundance, phosphorus, and DO concentrations, and was compared to key process rates, such as organic matter production and sedimentation, DO depletion rates, and estimates of hypoxic area (Zhou et al., 2013) by taking advantage of a new empirical relationship between bottom water DO and area (Zhou et al., 2013). It was further tested with independent DO concentrations from the period 1960–1985.

Exploring loading targets for water quality

Rucinski et al.'s (2014) model was then used to develop response curves for hypolimnetic DO concentration, hypoxic-days (number of days per year with hypolimnetic DO below 2 mg/l), hypolimnetic DO depletion rates, and hypoxic area as a function of loading of TP and DRP into the WB and CB (Fig. 9). The resulting response curves incorporate uncertainty associated with interannual variability in weather and resulting lake stratification from the 19 calibration years. The response curves for hypoxic area and hypoxic days are used here to explore implications for new loading targets, as well as to discuss how such targets would compare to those aimed at reducing WB cyanobacteria blooms.

While the actual extent of “acceptable hypoxia” needs to be set through public discourse and policy, one reasonable expectation is to return to hypoxic areas of the mid-1990s prior to the increases ($\sim 2000 \text{ km}^2$), which coincided with the recovery of several recreational and commercial fishes in Lake Erie's WB and CB (Ludsin et al., 2001). By inspection (Fig. 9a), the current US/Canadian TP loading target (IJC, 1978) of 11,000 MT (WB + CB equivalent is 9845 MT or 89.5% of total lake TP load) is not sufficient. In fact, if the desired outcome is for average hypoxic area to not exceed 2000 km^2 for roughly 10 days per year, the WB + CB TP load would have to be approximately 4300 MT/year (4804 MT/year total lake load; Table 2). This is a 46% reduction from the 2003–2011 average loads and 56% below the current target, or a reduction of 3689 MT/year (4122 MT/year from the total lake load).

If this same hypoxic goal were used to set new targets for DRP loading (Fig. 9b), the WB + CB load would have to approach 550 MT/year (total equivalent load is 598 MT/year because WB + CB is 92% of the total DRP), which is roughly equivalent to values in the early 1990s. Because DRP load has increased so dramatically since that time, this represents a 78% reduction from the 2005–2011 average DRP load, or a reduction of 1962 MT/year (2133 MT/year from the total lake load). Importantly, these response curves indicate that a focus on DRP requires about half of the reduction of the TP target which is consistent with the higher bioavailability of DRP.

Also noteworthy is the fact that recent recommendations to reduce the occurrence of WB cyanobacteria blooms may not be sufficient to also meet a CB hypoxia goal of 2000 km^2 . For example, the Ohio Lake Erie Phosphorus Task Force recommended that to keep blooms to acceptable levels, the March–June Maumee River TP loads (as a surrogate for all WB tributaries) should be less than 800 MT (Ohio EPA, 2013),

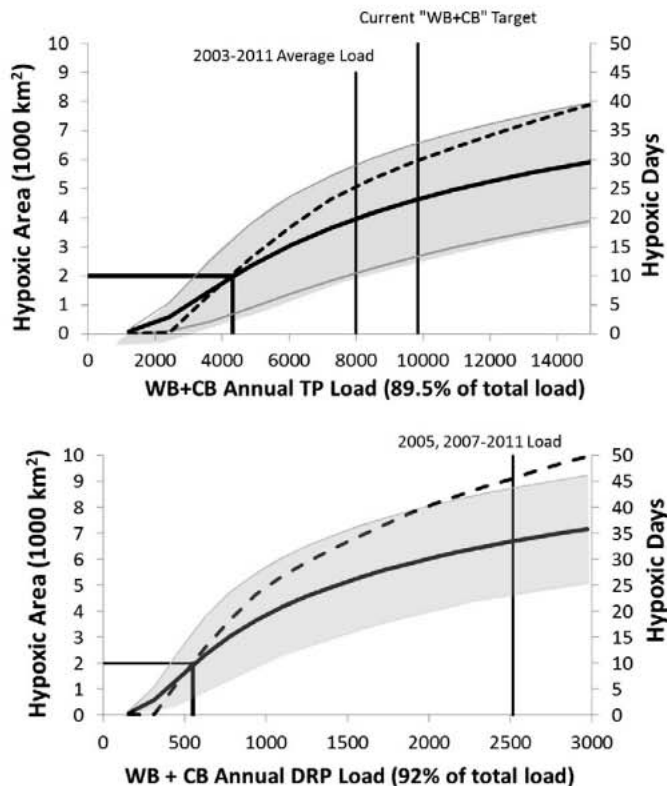


Fig. 9. Relationship between hypoxic area and hypoxic days and annual loading of total phosphorus (TP; upper panel) and dissolved reactive phosphorus (DRP; lower panel) in Lake Erie's west and central basins. The vertical lines represent recent and current target loads, and the shaded areas represent uncertainty around the hypoxic area response curves associated with interannual weather variability. Horizontal lines represent potential target hypoxia areas corresponding to those of the 1990s. Source: Rucinski et al. (2014).

which is a 31% reduction from the 2005–2011 average of 1160 MT (R.P. Richards, pers. comm.). If all CB and WB non-point sources (5534 MT; Table 2) were reduced by the same 31% and applied across the full year, the resulting annual CB + WB TP load would be reduced from 7989 to 6273 MT/year, which is still considerably higher than the 4300 MT/year target identified above.

In setting lake-wide loading targets, a single solution to address both water quality problems may be difficult (or impractical) to achieve. Our analyses suggest that WB cyanobacteria and CB hypoxia endpoints need

to be considered separately (Stumpf et al., 2012; Rucinski et al., 2014). The focus on spring load in controlling WB cyanobacteria blooms (e.g., Ohio EPA, 2013) is a logical focus for CB hypoxia because much of the load, particularly from non-point sources, enters the lake during that period (Richards et al., 2010).

Potential loading targets for fishes

While estimating reductions in nutrient loads necessary for attaining water quality goals is relatively straightforward, using fish metrics to estimate appropriate nutrient loads presents a greater challenge for various reasons. First, fish species (and ontogenetic stages) vary in their thermal responses and sensitivity to low oxygen conditions and direct responses to low oxygen will be species- and life stage-specific. Second, nutrient inputs and hypoxia do not only influence fish health directly; they also indirectly affect fish by altering the availability of quality habitat (e.g., DO availability, prey availability, water clarity) for growth, survival, and reproduction. Further, individual- and population-level responses to nutrient-driven changes in habitat quality can be mediated by a variety of individual behaviors that we do not fully understand (e.g., horizontal and vertical movement) and both intra-specific and inter-specific interactions that vary through both space and time (Eby and Crowder, 2002; Rose et al., 2009). Third, the variety of individual, population, and community indices that could be used to quantify responses of fish to hypoxia (e.g., habitat suitability, spatial distributions, feeding patterns, growth, survival, reproductive success, and overall production of population biomass) will not respond uniformly to hypoxia. As such, hypoxia targets based on expected fish responses would need to consider not only differential responses across species and ontogenetic stages, but also potentially different responses across population and community metrics.

As described above, different modeling strategies allow for focusing on various pathways through which hypoxia may affect fish populations. Relatively straightforward approaches may include statistical relationships based on several years of monitoring of hypoxia and population metrics or quantifying the amount of suitable habitat for a specific species (e.g., Arend et al., 2011) while more dynamic models may emphasize how behavior and biological interactions may mediate species-specific responses. To illustrate how models can be used to identify nutrient loading targets based on fish responses, we applied Arend et al.'s (2011) model of growth rate potential based on outputs from Rucinski et al.'s (2014) one-dimensional (daily, 0.5 m depth cells) limnological model, applied under various annual nutrient loading levels and climate conditions. Specifically, we applied the model for adult and juvenile yellow perch (i.e., a cool water species, relatively tolerant of low oxygen concentrations) and rainbow smelt (a cold water species, sensitive to low oxygen), as well as adult emerald shiner and round goby (Fig. 10). For each species and climatic scenario, habitat quality (e.g., the percent of modeled habitat with positive growth potential) declined with increasing annual TP loads, with the sharpest reductions in habitat quality occurring after TP levels exceeded ~5000 MT/year. This modeling exercise clearly illustrates the potential for reductions in nutrient-driven hypoxia to positively influence habitat quality for Lake Erie fishes, especially adult rainbow smelt and round gobies (Fig. 10). Moreover, the greatest increases in fish habitat quality would occur at roughly the same load reduction described above for the potential hypoxia goal (4000–5000 MT/year).

Approaches to meet new targets

If reducing hypoxic area to 2000 km² were desired, the above analyses indicate a load reduction of 3689 MT/year from the WB and CB loads (Table 2). A comparison of the potential reductions from point and non-point sources (Fig. 11), based on the current load breakdown described in Table 1, shows that with even the drastic measure of eliminating all point sources, substantial non-point source reductions would

Table 2

Relationships between west basin (WB) plus central basin (CB) versus lakewide total phosphorus (TP) and dissolved reactive phosphorus (DRP) loads in metric tons per year (MT/year) into Lake Erie, 2003–2011. Data from Dolan (unpublished data) based on methods outlined in Dolan and McGonagale (2005) and Dolan and Chapra (2012).

	WB + CB	Total
WB + CB/total TP	89.5%	
Current TP target	9845	11,000
2003–2011 TP loads	7989	8929
2003–2011 Non-point source loads	5534	6183
TP load to get 2000 km ²	4300	4804
% Reduction from current TP load	46%	
% Reduction from current TP target	56%	
TP load reduced from current	3689	4122
WB + CB/Total DRP	92%	
2005, 2007–2011 DRP loads	2512	2730
DRP load to get 2000 km ²	550	598
% Reduction from current DRP load	78%	
DRP load reduced from current	1962	2133

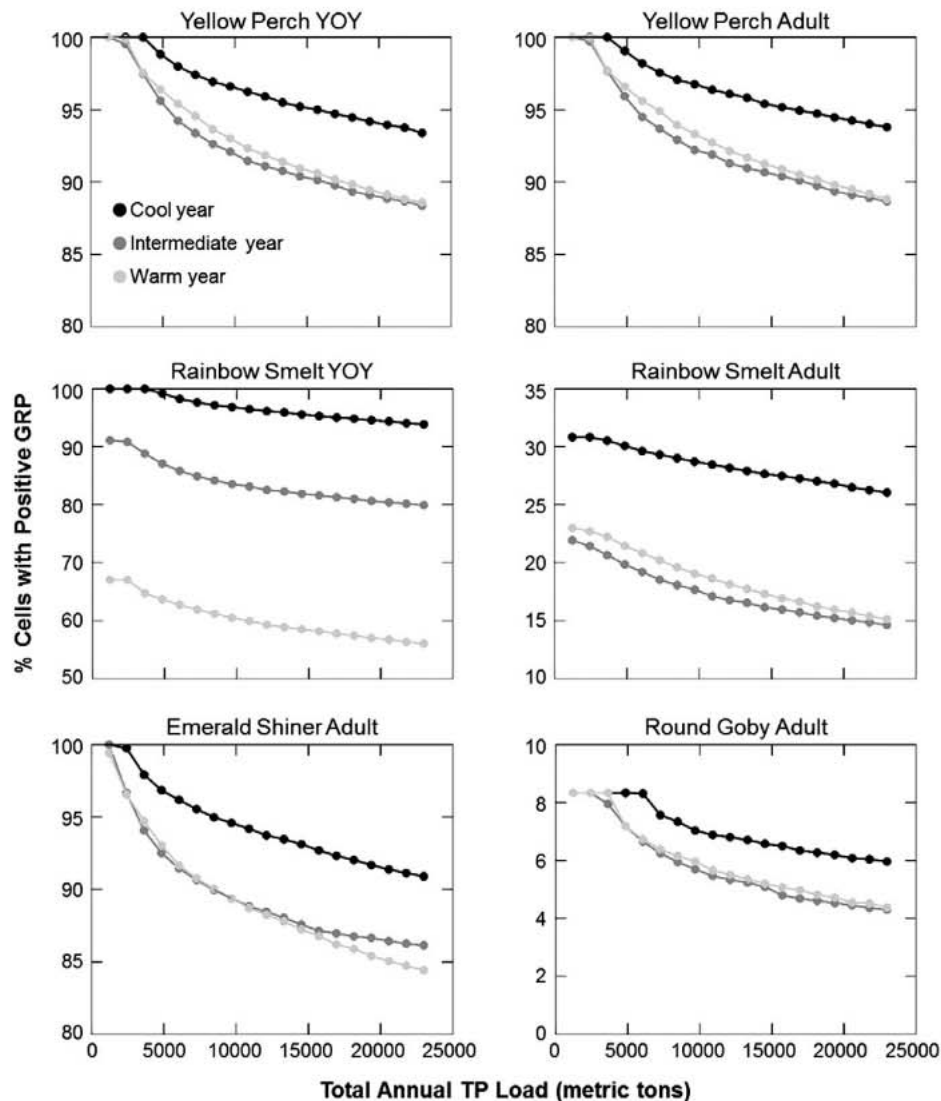


Fig. 10. Annual habitat quality for six species of fish common to central Lake Erie. Annual habitat quality is indexed as the percent of daily 0.5 m depth cells with positive growth rate potential (GRP) from July 15–Oct 31 (i.e., 109 days $\times$ 48 depth cells = 5232 total cells per year). To estimate GRP, we used daily, depth-specific temperature and dissolved oxygen concentrations, which were simulated with the model of Rucinski et al. (2014) and the age- and species-specific bioenergetics models developed by Arend et al. (2011). Estimates of GRP were quantified across 3 years of contrasting temperature (cool = 1992; intermediate = 1990; warm = 2002; see Rucinski et al. (2014) and 19 nutrient loading levels (1997 base loads scaled by a factor of 0.1–1.9).

be necessary. Because of this and because increases in the frequency and magnitude of winter and spring storm events (Kling et al. 2003; Kunkel et al., 1999) will draw additional attention to non-point sources (Daloğlu et al., 2012), the following sections focus on the more difficult challenge of prioritizing actions for controlling non-point sources of nutrients.

Spatial distributions of loading sources

Phosphorus loads to Lake Erie are not distributed equally across the basin. The WB received approximately 60% of the 2003–2011 average TP loads; whereas the CB and EB received about 30% and 10%, respectively. The WB received 68% of the 2005, 2007–2011 average DRP loads; whereas the CB and EB received 24% and 8%, respectively. The loads from individual tributaries within each basin also vary considerably for both TP and DRP, with the largest contributions coming from the Maumee, Detroit, Sandusky, and Cuyahoga rivers (Fig. 12). Thus, it is clear that loads to the WB are a very important determinant of the WB and CB eutrophication response.

The sources and fates of watershed TP also vary considerably. As described previously, Han et al. (2012) quantified the net anthropogenic TP inputs for 18 U.S. watersheds from fertilizers, atmosphere, detergents, and the net exchange in food and feed. TP budgets were also constructed for the soil and water compartment of each watershed, and those are especially helpful for comparing inputs. Here, we re-categorize inputs and outputs as TP from fertilizers, animal manure, atmosphere, human loading, and net crop export (Fig. 13). While TP inputs to the Lake St. Clair, Clinton, Detroit, Huron, Cuyahoga, and Ashtabula watersheds (#2–4, 13, 14) are dominated by human sources, inputs to the St. Clair, Ottawa-Stony, Raisin, Maumee, Cedar-Portage, Sandusky, Huron-Vermilion, and Cedar Creek watersheds (#1, 6–11, 24) are dominated by fertilizer; and inputs to the Grand (Ont) and Thames watersheds (#19, 20) are dominated by manure.

Just as tributary loads are not evenly distributed among major watersheds, non-point sources within those watersheds vary considerably. To explore this heterogeneity, Bosch et al. (2013) applied calibrated SWAT models (Bosch et al., 2011) of the Huron, Raisin, Maumee, Sandusky, Cuyahoga, and Grand watersheds representing together 53%

of the binational Lake Erie basin. These authors simulated subwatershed average annual TP and DRP yields (Fig. 14) for 1998–2005. Their results indicate, for example, that the Maumee River subwatersheds with the highest DRP yield were located sporadically throughout the watershed; whereas, those yielding high TP loads were found primarily in its upper reaches. By contrast, high-yield subwatersheds for both DRP and TP were dispersed throughout the Sandusky River watershed; while subwatersheds in the upper reaches of the Cuyahoga River watershed were the greatest sources of both DRP and TP. Findings such as these led Bosch et al. (2013) to conclude that DRP and TP flux is not uniformly distributed within the watersheds. For example, 36% of DRP and 41% of TP come from ~25% of the agriculturally dominated Maumee River subwatersheds. Similar disproportionate contributions of DRP and TP were found for the Sandusky River watershed (33% and 38%, respectively) and Cuyahoga watershed (44% and 39%, respectively).

These collective results suggest that spatial targeting of management actions would be an effective P reduction strategy. However, it is important to note that these loads represent flux to the stream channels at the exit of each subwatershed, not P delivered to the lake. Thus, the maps of important contributing sources of TP and DRP to the lake could be different if flux to the lake were considered.

Agricultural BMPs

In addition to identifying potential sources of TP and DRP to the Lake Erie ecosystem, the EcoFore-Lake Erie program sought to evaluate how land-use practices could influence nutrient inputs that drive hypoxia formation. In the following sections, we review some of the available best management practices (BMPs) and use SWAT modeling to test their effectiveness in influencing nutrient flux.

McElmurry et al. (2013) reviewed the effectiveness of the current suite of urban and agricultural BMPs available for managing P loads to Lake Erie. Because of the dominance of agricultural non-point sources, we focus here on agricultural BMPs. The Ohio Lake Erie Phosphorus Task Force also recommended a suite of BMPs for reducing nutrient and sediment exports to Lake Erie (OH-EPA 2010).

Source BMPs (Sharpley et al., 2006) are designed to minimize P pollution at its source. *Efficient fertilizer management* is reflected in the “4R” stewardship framework, based mostly on Roberts (2007), which focuses on applying the right formulation at the right rate and right times in the right places. While the appropriate application method is determined by the crop, cropping systems, and soil properties, methods that place the fertilizer in contact with the soil (e.g. injection, in-row placement) and away from the surface are preferred. *Animal feed*

management controls the quantity and quality of available nutrients, feedstuffs, or additives in feed thereby improving efficiency; reducing nutrients and pathogens in manure; and reducing odor, particulate matter, and greenhouse gas emissions. *Manure management* minimizes manure loss during storage, and land application at agronomically appropriate amounts.

Transport BMPs are designed to reduce the runoff of P with water and sediments. *Conservation Tillage* leaves at least 30% of the soil surface covered with crop residue to reduce soil erosion through mulch-till, strip-till, no-till, and ridge-till techniques. However, recent studies suggest that the often-associated broadcast fertilization techniques may lead to elevated DRP loss (e.g., Daloğlu et al., 2012; Seo et al., 2005; Sweeney et al., 2012; Tiessen et al., 2010; Ulen et al., 2010). *Conservation Cropping and Buffers* are designed to reduce sediment and nutrient runoff, and in some cases, provide vegetative cover for natural resource protection. *Treatment Wetlands* treat runoff from agricultural processing and storm runoff and *grassed waterways* are designed to reduce gully erosion. Wetlands and grassed waterways are effective in reducing P loading, and grassed waterways are most effective in reducing erosion (Dermis et al., 2010; Fiener and Auerswald, 2003; Fisher and Acreman, 2004). *Drain Tiles* are designed to facilitate movement of water from the field, and if flow to the tile is through the soil matrix, sediment, particulate P (PP), and DRP losses are minimized. However, recent work has suggested that preferential flow through worm holes and soil cracks, for example, brings surface water and its constituents directly into the tiles (Gentry et al., 2007; Reid et al., 2012). So, *Drain Management* actions that slow down or retain water can reduce particulate nutrients, pathogen, and pesticide loading from drainage systems.

Focus on management of DRP

Given the dramatic increase in the proportion of TP that is delivered to Lake Erie from agricultural watersheds as DRP, differentiating between BMPs focused on particulate P (PP) vs. DRP is important. While TP is generally considered to be only partially bioavailable (Baker, 2010), most of DRP is bioavailable. The combination of movement toward no-till and associated broadcast application appears to have exacerbated loss of DRP from no-till lands. Seo et al. (2005) reported DRP as 70% of TP in runoff from a no-till/broadcast fertilized field, and Ulen et al. (2010) reported that DRP losses increased by a factor of four in a no-till compared to conventional-till systems. Likewise, Tiessen et al. (2010) reported that conversion to conservation tillage increased P concentrations and exports, mostly as soluble P, especially during snowmelt. Kleinman et al. (2011) showed that while PP decreased by 37% in a no-till vs. conventional-till watershed, TP increased by 12%, with that increase attributed to dissolved P mediated by high concentrations of surface soil P. BMPs that lower the accumulation of P at the soil surface should be considered in areas where DRP is a major concern (Tiessen et al., 2010). A summary of BMPs that focused on controlling DRP (Crumrine, 2011) outlines their potential effectiveness, costs, and likelihood of use.

Evaluating watershed-scale effectiveness of traditional agricultural BMPs

Bosch et al. (2013) explored the impacts of expanding the current use of filter strips, cover crops, and no-till BMPs in controlling runoff. When implemented singly and in combinations at levels currently considered feasible by farm experts, these BMPs reduced sediment and nutrient yields by only 0–11% relative to current values (Fig. 15). Yield reduction was greater for sediments and the greatest reduction was found when all three BMPs were implemented simultaneously. They also found that targeting BMPs in high source locations (see above), rather than randomly, decreased nutrient yields more; whereas, reduction in sediment yields was greatest when BMPs were located near the river outlet. A more detailed analysis of increased BMP implementation

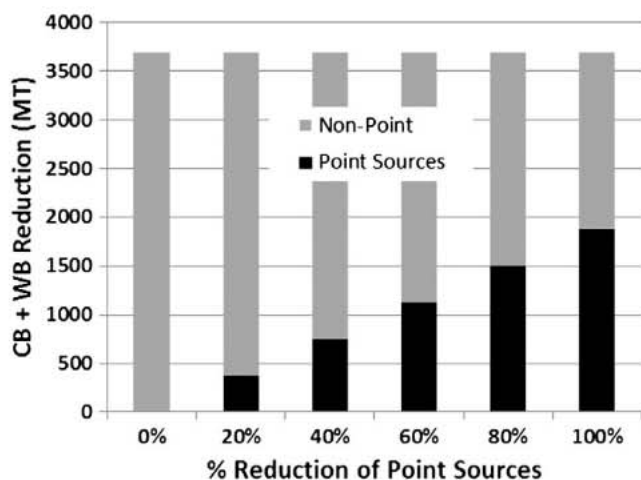


Fig. 11. Hypothetical allocations of the 3689 MT/year load reduction needed to achieve 2000 km² hypoxic area between point and non-point sources in Lake Erie's western and central basins.

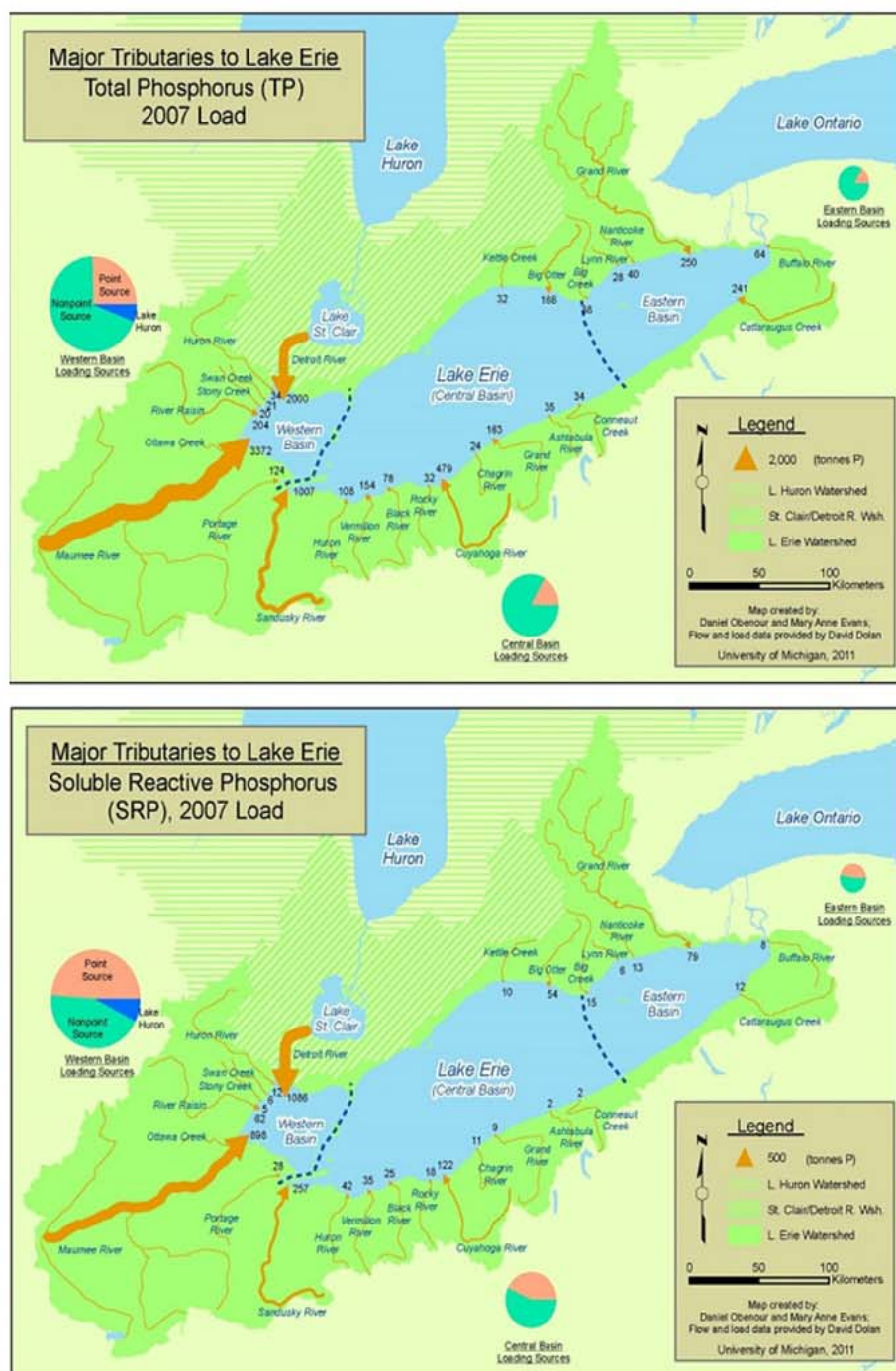


Fig. 12. Relative total phosphorus (TP) and dissolved reactive phosphorus (DRP) loads for the major tributaries of Lake Erie in 2007. Loads are proportional to the drawn river widths. Data source: David Dolan, unpublished.

strategies for the Maumee watershed (Fig. 16) pointed to the need for more aggressive implementation of multiple BMPs to reduce loads substantially. For example, a 20% reduction in TP or DRP load requires implementing the BMPs on more than 50% of the agricultural land.

Climate change implications

Meteorological conditions, including both temperature and precipitation, have changed appreciably during the past century in the Great Lakes basin, with increased temperature and winter/spring

precipitation expected into the future (Hayhoe et al., 2010; Kling et al. 2003). Thus, establishing loading targets to control Lake Erie hypoxia should consider how potential climate change might impact loads, processes that lead to hypoxia formation, fish, and BMP effectiveness.

Watershed impacts

While uncertainty surrounding the projected future regional precipitation is greater than for temperatures, confidence is increasing that future precipitation patterns will continue to trend toward more intense

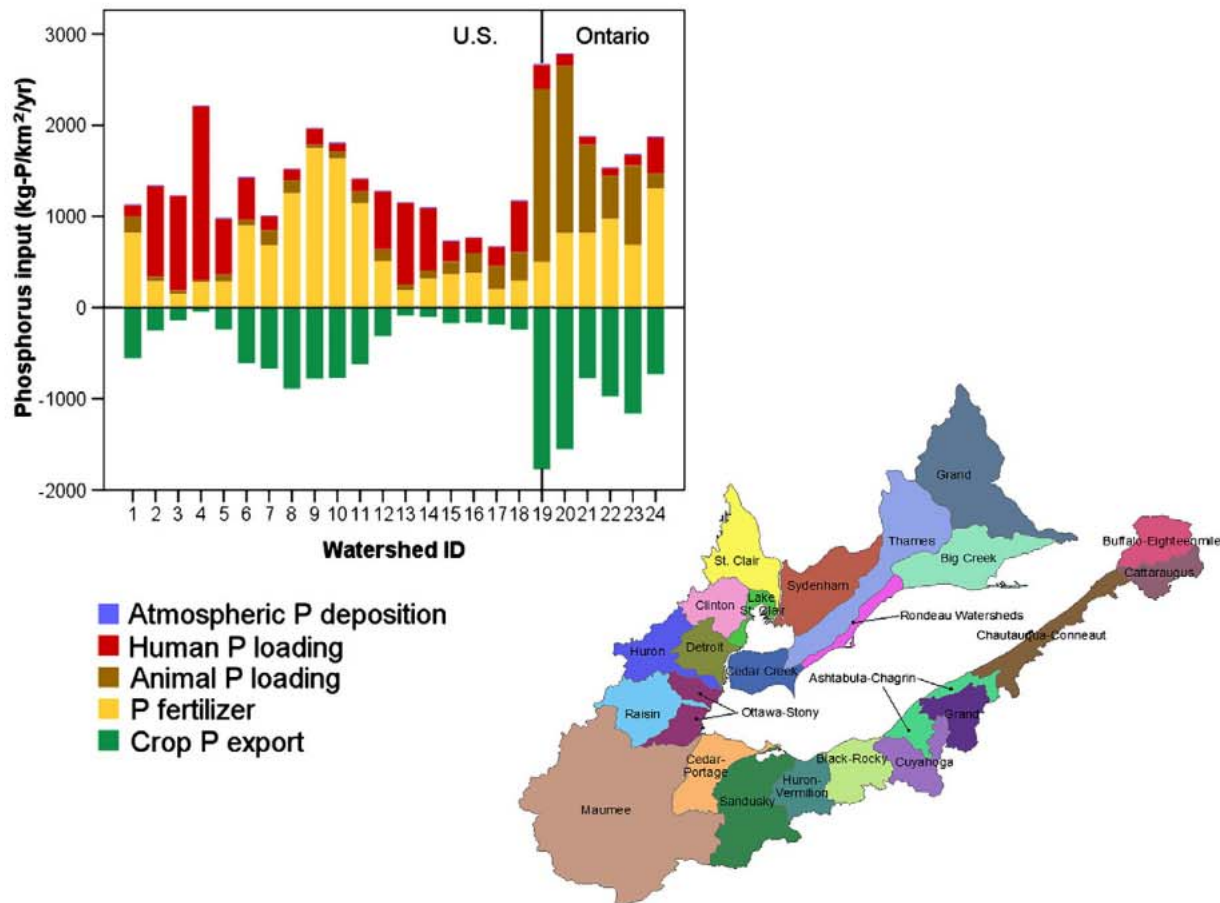


Fig. 13. Inputs and outputs of total phosphorus (TP) for 24 Lake Erie watersheds during 2002. Watersheds are numbered counter-clockwise starting with the St. Clair River. Source: Han et al., 2012.

late-winter and early spring precipitation events (Hayhoe et al., 2010). Such intense events could lead to higher nutrient runoff, and in the absence of dramatic changes in land use, could increase overall nutrient loads because 60–75% of P inputs are delivered during precipitation-driven river discharge events (Baker and Richards, 2002; Dolan and McGunagle, 2005; Richards et al., 2001). A preliminary study of the impact of climate change on the Maumee River (DeMarchi et al., 2011) suggested a 10–30% increase in sediment load, depending on the general circulation model (GCM) and greenhouse gas emission scenario. In fact, these changes have already been happening. Daloğlu et al. (2012) showed through modeling efforts that higher frequency intense storms of today's climate is a key driver of elevated DRP loads from the Sandusky River watershed. Similarly, Michalak et al. (2013) showed that such extreme precipitation events in 2011 drove substantially higher P loads, resulting in massive WB and CB cyanobacteria (*Microcystis*) blooms.

Hypoxia formation impacts

Lower water levels predicted by some climate models (Angel and Kunkel, 2010) would lead to a thinner hypolimnion (Lam et al., 1987a, 1987b) and increase in DO depletion (Bouffard et al., 2013). Warmer future temperatures (Hayhoe et al., 2010; Kling et al., 2003) should lead to a longer summer stratified period, with thermal stratification developing earlier in the year and turnover occurring later in the year (Austin and Coleman, 2008). A longer stratified period would allow hypolimnetic oxygen to be depleted over a longer time period and warmer hypolimnetic temperatures could lead to higher respiration rates and more rapid DO depletion (Bouffard et al., 2013). Changes in the wind regime (Pryor et al., 2009) will have important effects on lake stratification (Huang et al., 2012), impacting hypoxia formation

as well. Climate models predict an almost negligible increase in the mean wind speed in the next 50 years (Pryor and Barthelmie, 2011), although the frequency of extreme storms is expected to increase (Meehl et al., 2000). The result of increased strong winds will be a deeper thermocline (thinner hypolimnion) and likely increased rate of DO depletion (Conroy et al., 2011). Adding uncertainty to predictions of future hypolimnion thickness are potential changes in wind vorticity that controls thermocline depth through the Ekman pumping mechanism (Beletsky et al., 2013).

Fish impacts

Previous modeling has indicated that warm-water, cool-water, and even some cold-water fishes could benefit from climate change in the Great Lakes basin due to increased temperature-dependent growth (Minns, 1995; Stefan et al., 2001), lengthened growing seasons (Brandt et al., 2011; Cline et al., 2013), and increased over-winter survival of juveniles (Johnson and Evans, 1990; Shuter and Post, 1990). However, these expectations may not hold for cool- and cold-water fishes in the CB under increased intensity and duration of hypoxia. For example, by using a bioenergetics-based GRP model to compare a relatively warm year with prolonged hypoxia extending far above the lake bottom (e.g., 1988, a type of year that we would expect to become more frequent with continued climate change) to a relatively cool year with a thin hypoxic layer persisting for a short time (e.g., 1994, a type of year that we would expect to become less frequent in the future), we explored how climate change might influence fish habitat availability. The results of this analysis (also see Arend et al., 2011), suggest that climate warming can cause preferred habitat to be squeezed both from above (by warmer temperatures) and from below (via

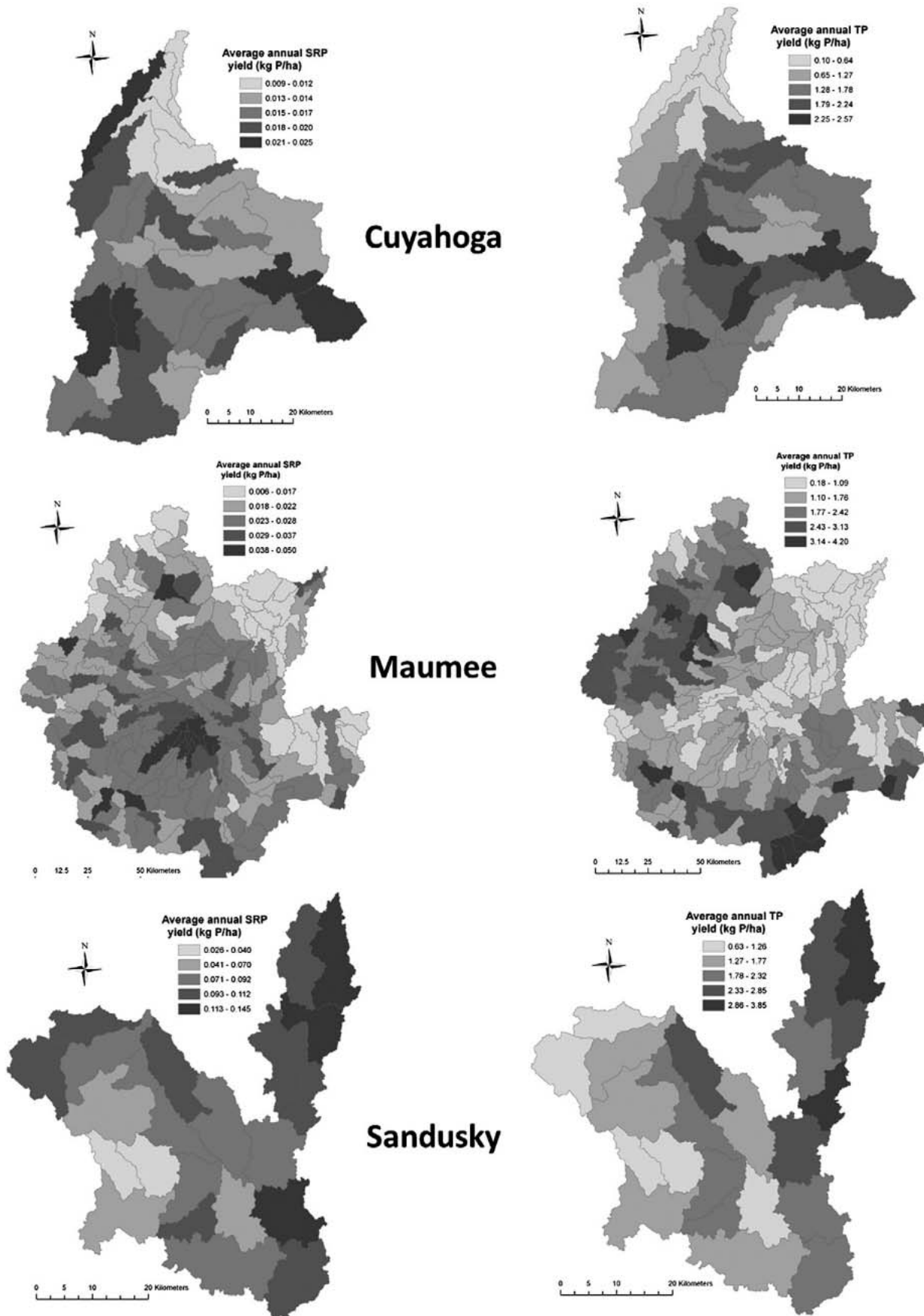


Fig. 14. Average annual dissolved reactive phosphorus (DRP) and total phosphorus (TP) yields from sub-basins of three major Lake Erie watersheds. Yields represent loss from the land, not delivery to the lake. Source: Bosch et al. (2013).

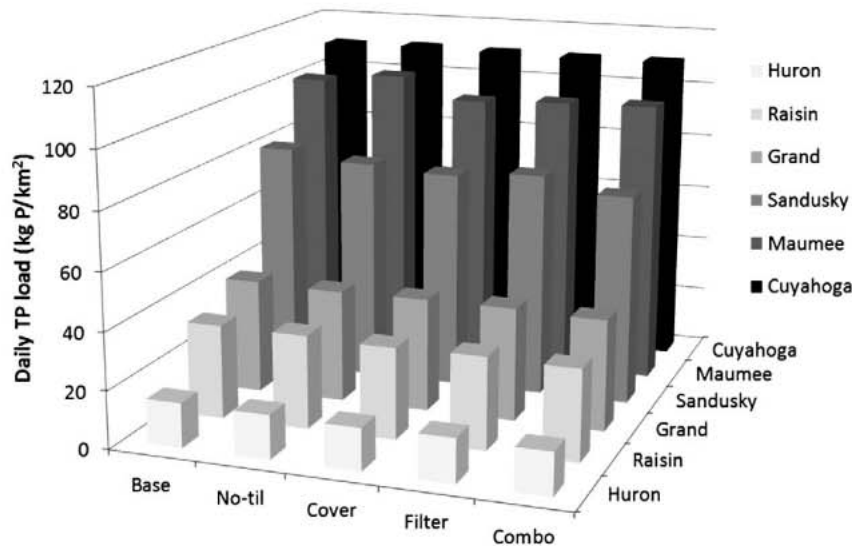


Fig. 15. Comparison of reduction in daily TP yield from implementing “feasible” best management practices, including no-till, cover crop, filter strips and a combination of all three, for six Lake Erie Watersheds as predicted by SWAT model scenarios. Source: Bosch et al. (2013).

increased hypoxia (Fig. 17). In fact, the influence of inter-annual variation in water temperature may have a stronger effect on fish habitat quality than nutrient loading (Fig. 10). Under a warmer climate, we may need to reduce loading levels even more dramatically to have meaningful positive effects on habitat quality and Lake Erie fish stocks (Shimoda et al., 2011).

Climate impacts on BMP effectiveness

Bosch et al. (in revision) assessed climate impacts on a range of BMPs with the SWAT model. They projected water flow, sediment yields, and nutrient yields (Figs. 18, 19), based on simple characterizations of future climates (Table 3) consistent with those projected from climate models (Hayhoe et al., 2010). These watersheds showed consistent increases in sediment yield, with increases being larger under more pronounced climate scenarios. They also found that under a warmer climate, sediment and nutrient yields would be greater from agricultural (e.g., Maumee and Sandusky) vs. forested watersheds (e.g., Grand in Ohio). Total annual discharge increased 9–17% under the more pronounced climate scenario and 4–9% under the moderate scenario. Stream sediment yields increased by 9% and 23% for moderate and pronounced climate scenarios, respectively. DRP yields decreased (□ 2% on average) under the moderate

climate scenario and increased slightly (3%) in response to more pronounced climate change. TP yields increased 4% under moderate climate change and 6% under pronounced climate change. Importantly, while agricultural BMPs might be less effective under future climates, higher BMP implementation rates could still substantially offset anticipated increases in sediment and nutrient yields (Fig. 19).

Implications for policy and management action

If “acceptable levels” (or goals) for hypoxia were set, the above-described response curves could be used to establish P loading targets. Given the emergence of DRP as a significant and increasing component of the total phosphorus load, the research presented above supports considering both TP and DRP targets. In addition, because the results of management actions aimed at addressing non-point sources tend to occur on the scale of years to decades, potential impacts of a changing climate need to be taken into consideration for effective action. The indications we have discussed suggest that climate change will not only exacerbate existing problems, but also make reducing loads more difficult.

Whole-lake targets alone may no longer be appropriate due to differences in temporal and spatial scales of loading on hypoxia and other environmental stressors. For example, CB hypoxia evolves over a longer seasonal time frame in response to loads distributed over wider spatial and temporal scales as evidenced by gradual oxygen depletion and the dependence on total lake loads (e.g. Burns et al., 2005; Rosa and Burns, 1987; Rucinski et al., 2010, 2014). Whereas, WB cyanobacteria blooms appear to be driven by relatively short-term loads of immediately available P (Michalak et al., 2013; Stumpf et al., 2012; Wynne et al., 2013). Thus, while a recent assessment demonstrated that the Detroit River had little impact on the massive 2011 cyanobacteria bloom (Michalak et al., 2013), it does not mean that the river is not an important driver for hypoxia; hypoxia development is a cumulative process that can be influenced by longer term loads of both immediately available DRP and P that is made available through internal recycling mechanisms over the summer. Thus, a new loading target aimed at reducing or eliminating cyanobacteria blooms might be insufficient in both magnitude and geographic proximity to reduce hypoxia. Because the major components of the P load are now from non-point sources, and because resources available to address those sources will always be limited, management efforts will be most cost effective if placed on sub-watersheds that deliver the most P. We now have the ability to identify not only the most important contributing watersheds (e.g., Detroit, Maumee, Sandusky), but also the regions within those tributary

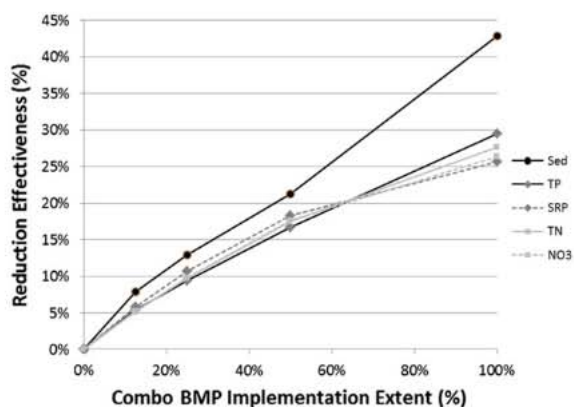


Fig. 16. Average annual percent reduction in riverine yields for the Maumee watershed under various implementation extents (% of agricultural land area) of combined BMP conditions as predicted by SWAT model scenarios. Source: Bosch et al. (2013).

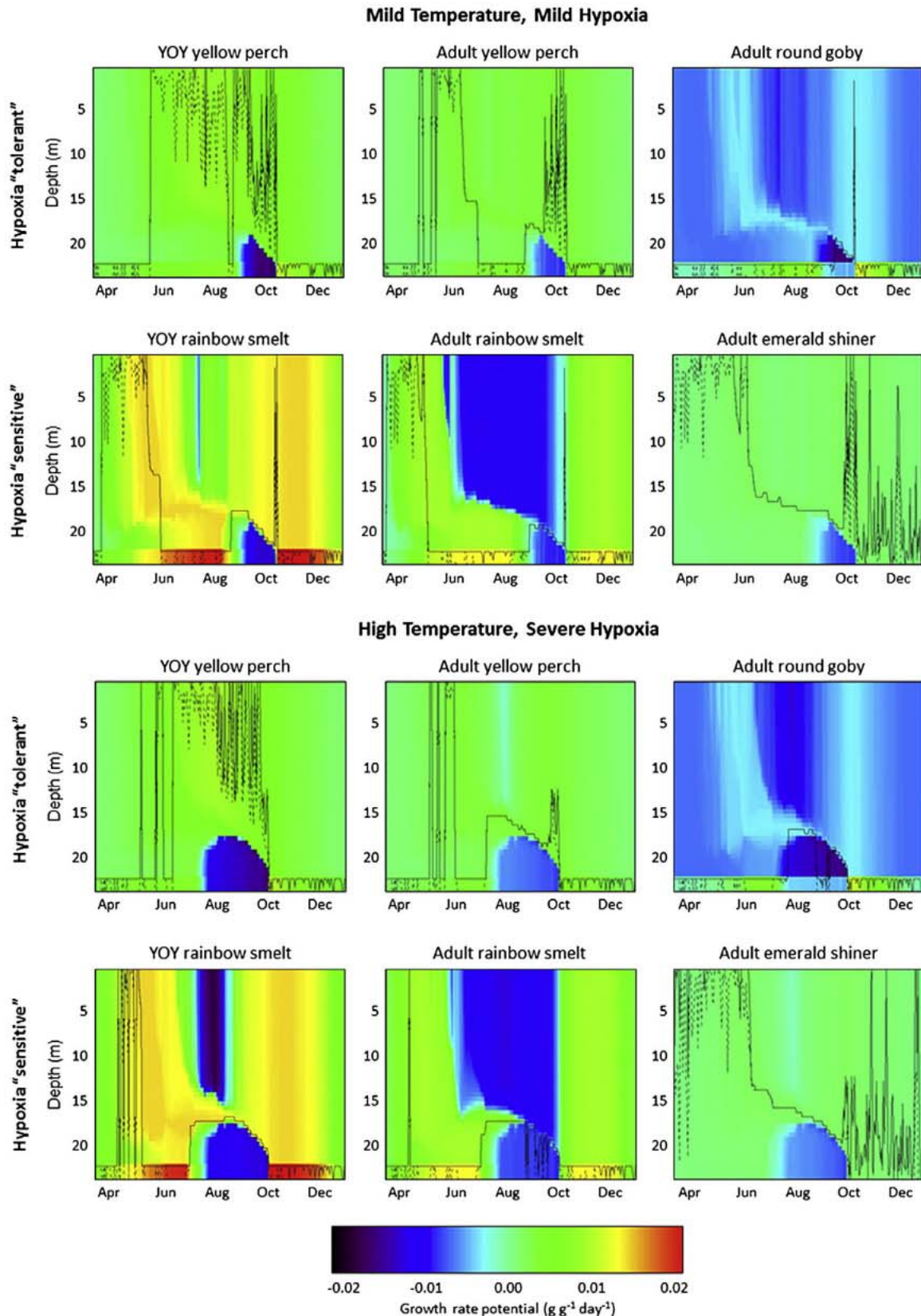


Fig. 17. One-dimensional habitat quality for six representative fishes in central Lake Erie as indexed by bioenergetic growth rate potential (GRP). This index of habitat quality is an integration of vertical temperature and dissolved oxygen daily hindcasts (from Rucinski et al., 2010) and is based on the assumption that fish feed at 50% of their maximum daily rate. Colors depict habitat quality (GRP) and the black line tracks the vertical position of daily greatest habitat quality. Note the difference in habitat quality between a cool year, with brief hypoxia, 1994 (top panel), as compared to a warm year with a long duration of hypoxia, 1988 (bottom panel). Model details about this modeling approach are presented in Arend et al. (2011).

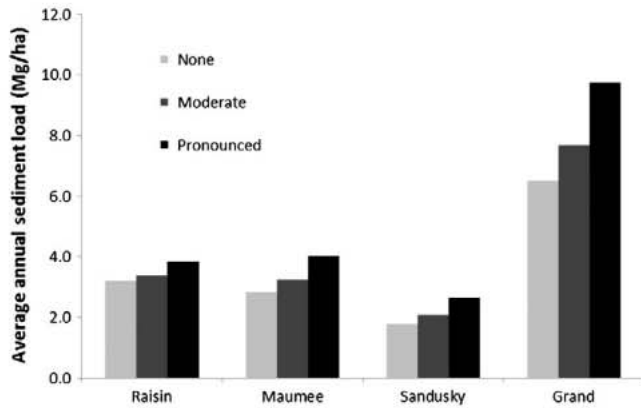


Fig. 18. Predicted average annual sediment load (Mg/ha) from land to stream channel for the Raisin, Maumee, Sandusky, and Grand (OH) rivers under three different climate conditions, increasing in severity (no change, moderate change and pronounced change in climatic conditions). Source: Bosch et al. (in revision).

watersheds that release the most P. This knowledge should allow for more effective targeting of BMPs to high-load subwatersheds, assuming that the stakeholders in those regions are open to these options. For this reason, research that identifies factors that drive land-use decision-making behavior and how these motivations and behaviors vary across

Table 3

Future climate scenarios (moderate and pronounced) for temperature and precipitation used in the SWAT model. Seasons were defined as Winter (December–February), Spring (March–May), Summer (June–August), and Fall (September–November). From Bosch et al. (in revision).

Season	Moderate		Pronounced	
	Temperature (°C)	Precipitation (%)	Temperature (°C)	Precipitation (%)
Winter	+2		+5	
Spring		+11		+29
Summer	+4		+7	
Fall				+7

the watershed will be essential to help policy-makers determine the ability to meet any newly developed loading targets through implementation of spatially-targeted BMPs.

For example, current farm policy is based on volunteer, incentive-based adoption of BMPs. The 2014 U.S. Farm Bill includes a focus on special areas and replacing subsidies with revenue insurance, providing opportunities to employ more targeted approaches. Daloğlu et al. (in press) point out that farmer adoption will be critical, and their analysis suggests that coupling revenue insurance to conservation practices reduces unintended consequences. For example, using a social-ecological-system modeling framework that synthesizes social,

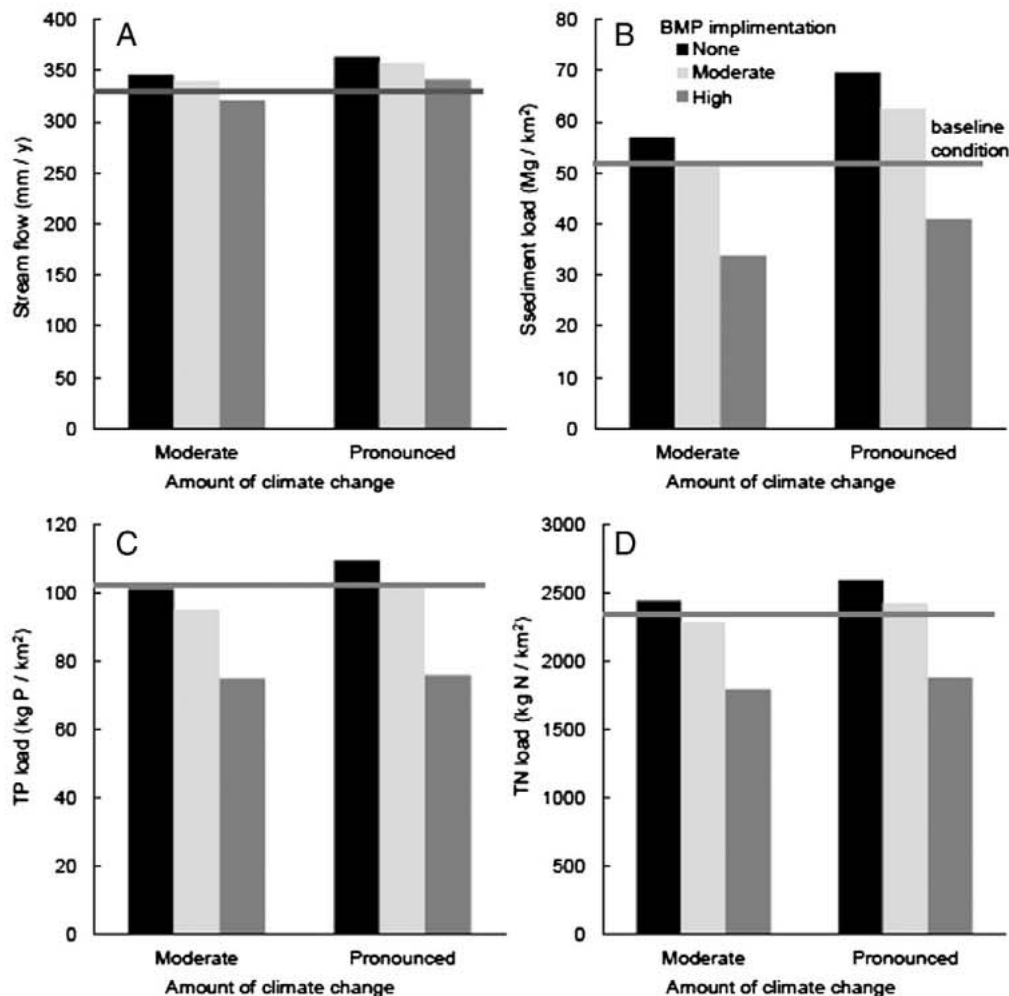


Fig. 19. Predicted average annual stream flow (panel A), sediment load (panel B), total phosphorus (TP) load (panel C), and total nitrogen (TN) load (panel D) for the Maumee watershed under various climate change and best management practice (BMP) conditions. In each panel, a horizontal line marks the baseline (no climate change and no BMPs) condition for flow or load. Source: Bosch et al. (in revision).

economic, and ecological aspects of landscape change under different agricultural policy scenarios, Daloğlu (2013) and Daloğlu et al. (in press) evaluated how different policies, land management preferences, and land ownership affect landscape pattern and subsequently downstream water quality. This framework linked an agent-based model of farmers' conservation practice adoption decisions with SWAT to simulate the influence of changing land tenure dynamics and the crop revenue insurance in lieu of commodity payments on water quality over 41 years (1970–2010) for the predominantly agricultural Sandusky River watershed. The results showed that non-operator owner involvement in land management decisions yielded the highest reduction in sediment and nutrient loads and that crop revenue insurance tended to create a homogeneous conservation landscape with slight increases in sediment and nutrient loads. However, it also suggested that linking crop insurance to conservation compliance and strengthening and expanding conservation compliance provisions could reduce nutrient loads. Daloğlu (2013) and Daloğlu et al. (in press) demonstrated, for example, that DRP load decreased by 6% with conservation compliance that included structural BMPs, as compared to an increase of 8% without compliance. The relatively small percent changes, however, reinforce the recommendation of Bosch et al. (2013) that significantly more BMP implementation is needed.

Experiences in other large regions with nutrient problems (e.g., Chesapeake Bay, Gulf of Mexico/Mississippi River) have shown that significantly reducing non-point source loads is difficult. Not only are the sources spatially distributed, but the methods used are primarily voluntary and incentive based and thus difficult to target and track. Reducing non-point inputs of sediments and nutrients is also difficult because the response time between action and result can be many years or longer, and the results can only be measured cumulatively in space and through time. For these reasons, we recommend the use of an adaptive management approach that sets “directionally correct” interim targets, evaluating the results both in loads and lake response on appropriate time-scales (e.g., 5-year running averages), and then adjusting management actions or loading targets, if necessary. Lake Erie is a good candidate for such an approach because its short water residence time (2.6 years) reduces one common time-lag in system response. Such an approach would also allow for more effective testing and post-audits of the ability of models to project the ecosystem's response and thus improve subsequent assessments and projections. We see this iteration of research and analysis, management-focused model development and application, management action, and monitoring of results as a particularly effective way to manage large, spatially complex ecosystems. If the monitored results are not as anticipated, returning to research and model refinement establishes a learning cycle that can lead to better informed decisions and improved outcomes.

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is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Dedication

This paper is dedicated to the memory of Dr. David Dolan, one of the authors. His untimely death is a great loss to the entire Great Lakes community. We will miss his friendship, insights, important and continuing contributions to the International Association of Great Lakes Research, and unfailing dedication to ensure that our community and the world both understand and have access to the changing sediment and nutrient loads to the Great Lakes. Dave was truly a “Great Lakes Man”.

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