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Commentary

Needed: Early-term adjustments for Lake Erie phosphorus target loads to address western basin cyanobacterial blooms

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ABSTRACT

For Lake Erie, it is already time to revise the phosphorus target loads set to address the problem of cyanobacterial blooms in the Western Basin. Current targets were proposed by the Annex 4 task group in 2015, adopted by U.S. and Canadian governments in 2016, and set as objectives of domestic action plans in 2017. These targets, applicable to all spring discharges below the 90th percentile, set a maximum load for both total phosphorus (TP) and dissolved reactive phosphorus (DRP) equivalent to 60% of their 2008 spring loads. This essentially mandates 40% reductions in both particulate phosphorus (PP) and DRP loading relative to 2008 loads. These targets do not explicitly incorporate the difference in bioavailability between DRP (~100% bioavailable) and PP (~25% bioavailable). From 2008 to 2017, DRP comprised 24% of the spring TP load and over half (~56%) of the total bioavailable phosphorus (TBAP) load, while PP comprised 76% of the TP load but only ~44% of the TBAP load. Subsequent deposition of PP in the estuarine and nearshore zones further reduces its significance in bloom development. By ignoring differences in bioavailability, the current targets provide no guidance for choosing among practices based on their relative effectiveness in reducing DRP or PP and their combined reductions in TBAP loading. Current targets place more emphasis on PP than needed to efficiently reach targeted cyanobacterial bloom reductions. To clarify appropriate management approaches and lead to greater success in reducing cyanobacterial blooms, target loads should be based on TBAP.

Introduction

To address problems associated with the re-eutrophication of Lake Erie, an Objectives and Targets Task Team (subsequently referred to as Task Team) was established under Annex 4 of the 2012 Amendments to the U.S.-Canadian Great Lakes Water Quality Agreement (IJC, 2012). The Task Team was charged with developing new phosphorus target loads for Lake Erie, taking into account the bioavailability of various forms of phosphorus. They issued their Final Report in May 2015 (Annex 4, 2015). To reduce the cyanobacterial blooms that had again become common in the western basin, that report recommended targets wherein the maximum spring loads of total phosphorus (TP) and dissolved reactive phosphorus (DRP) from the Maumee River, only to be exceeded at discharges greater than the 90th percentile, would be equivalent to 60% of their respective 2008 loads (i.e., a 40% reduction from the 2008 loads). Those loading targets were accepted by the governments of Canada and the United States (IJC, 2016) and endorsed by the U.S. EPA's Science Advisory Board (2017). They are now serving as the basis for state, provincial and federal domestic action plans for meeting the target loads (Ohio Lake Erie Commission, 2018; U.S. Environmental Protection Agency, 2018).

The Task Team's Final Report also called for an active adaptive management process that would include review of (1) reduction targets in view of ongoing nutrient loading-algal bloom model development and additional monitoring data, (2) progress in adoption of best management practices (BMPs), and (3) information on river loading responses to BMP adoption. The report also recommended that such reviews take place in connection with the 5-year cycle of the Coordinated Science and Monitoring Initiative, which is due for Lake Erie in 2019. In view of changes in model calibrations and conclusions since the 2014 deliberations of the Task Team that have already been published (Bertani et al., 2016; Stumpf et al., 2016; Verhamme et al., 2016), together with ongoing research and monitoring studies, we believe that target loads should be revised to place more emphases on those forms of bioavailable phosphorus that most directly support cyanobacterial blooms. This commentary summarizes information on phosphorus bioavailability and loading from the Maumee River that supports such a revision.

Phosphorus bioavailability and agricultural nonpoint pollution

Phosphorus management for Lake Erie has occurred within the context of detailed mass balance studies that were initiated in 1974 as part

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of the U.S. Army Corps of Engineers' Lake Erie Wastewater Management Study (U.S. Army Corps of Engineers, 1979) and that continue to the present day (Maccoux et al., 2016). These studies support quantification of point sources, nonpoint sources, atmospheric deposition and Lake Huron outflows as summarized in Fig. 1. These data are most often presented as annual, stacked bar graphs including both direct and indirect point sources and monitored and unmonitored nonpoint sources (Maccoux et al., 2016; Scavia et al., 2014). In Fig. 1, we have plotted line graphs of 5-yr running averages of total point sources and total nonpoint sources, as well as atmospheric deposition and Lake Huron outflows, to emphasize the relative magnitudes of each major source as well as the timing of the large downward trend in point source loading and the minimal overall changes in nonpoint source loading (Maccoux et al., 2016).

The charge to the Annex 4 Task Team to consider the bioavailability of various forms of phosphorus stems from studies initiated in the late 1970s and early 1980s as agricultural nonpoint pollution was becoming the dominant source of phosphorus entering the lake (Fig. 1). Within the agricultural community, it was well known that crop growth depended on the uptake of orthophosphate from the soil solution (IPNI, 2012) and that various forms of soil phosphorus had very different equilibrium orthophosphate concentrations. Phosphorus soil tests were developed in the late 1940s to quantify the concentrations of soil phosphorus in forms that correlated with crop growth (CAST, 2000). Agricultural fertilizer management utilizes measurements of this bioavailable phosphorus in the soil (i.e., phosphorus soil test values) rather than the TP content of the soil, which is seldom even measured.

In contrast with agricultural phosphorus management, the environmental community has traditionally used measurements of TP for water quality management, even though orthophosphate is also the form of phosphorus taken up by aquatic plants, including algae and cyanobacteria (Lee et al., 1980). When point sources dominated phosphorus loading to Lake Erie, as in the 1960s and early 1970s (Fig. 1), TP measurements were adequate because, for those point sources lacking advanced phosphorus removal, most (70%–100%) of the phosphorus loading was either orthophosphate, and directly bioavailable to algae, or could readily be converted into orthophosphate (Lee et al., 1980; Sonzogni et al., 1982). In the 1970s, as agricultural nonpoint sources became the dominant source, particulate phosphorus (PP) became the dominant form of phosphorus entering the lake. Several studies using algal bioassays were conducted to assess the bioavailability of the PP associated with suspended sediments (SS) derived from cropland erosion (DePinto et al., 1981; IJC, 1979; Logan et al., 1979; Young and DePinto, 1982). For the agricultural tributaries to western Lake Erie, these studies suggested that only 20–30% of the PP was bioavailable particulate phosphorus (BAPP) and that chemical extractions of PP with NaOH provided useful indications of BAPP (IJC, 1979; Young et al., 1985).

Within aquatic environments, a second aspect of bioavailability was recognized – that of positional bioavailability (Sonzogni, 1983; Sonzogni et al., 1982). BAPP can settle out of the water column prior to releasing orthophosphate, and thus not directly support algal or cyanobacterial growth apart from subsequent contributions to internal phosphorus loading (Lee et al., 1980; Matisoff et al., 2016). The dissolved portion of agricultural nonpoint loading, as measured by DRP, consisted primarily of orthophosphate, or easily convertible forms, and, as such, was considered 100% bioavailable (Lee et al., 1980; Sonzogni et al., 1982). Because the dissolved portions of the phosphorus load are defined as forms passing through a 0.45 μm membrane filter and consist of either dissolved or colloidal materials, these forms were not subject to the same positional bioavailability limitations as BAPP.

The 11,000 metric ton year^{-1} (MTA) target phosphorus load set for Lake Erie under the 1972 GLWQA and its 1978 amendments was based on modeling studies using the relationships between TP loading, resulting TP concentrations in the lake, and the severity of hypoxia in the central basin. Issues of neither chemical nor positional bioavailability were incorporated into the targets, and instead, studies of bioavailability were identified as future research needs (IJC, 1980). In the 1980s, Annex 3 of the United States/Canada Water Quality Agreement called for an additional 2000 MTA reduction in phosphorus loading to Lake Erie, primarily from agricultural nonpoint sources. Again, the issue of bioavailability was placed in the category of “future research needs” (Great Lakes Phosphorus Task Force, 1986). The ongoing phosphorus mass balance studies for the lake were showing reductions in TP loading through the mid-1980s (DePinto et al., 1986; Dolan and Chapra, 2012), and the problems of eutrophication were becoming less evident. In fact, the lake was viewed as a “Poster Child” for lake recovery (Matisoff and Ciborowski, 2005). Unfortunately, the improved conditions were short-lived, and symptoms of re-eutrophication began to appear in the mid- and late 1990s, (Kane et al., 2014; Matisoff and Ciborowski, 2005; Scavia et al., 2014) even though the 11,000 metric tons per annum (MTA) target load was generally being met (Lake Erie LaMP, 2009). While multiple factors have been identified as possible contributors to the re-eutrophication of Lake Erie (Matisoff and Ciborowski, 2005; Smith et al., 2015), most studies concluded that large increases in DRP loading beginning in the 1990s in the Maumee River (Fig. 2) and other agricultural tributaries to the lake were the primary cause of re-eutrophication (Annex 4, 2015; IJC, 2014; Ohio EPA, 2010).

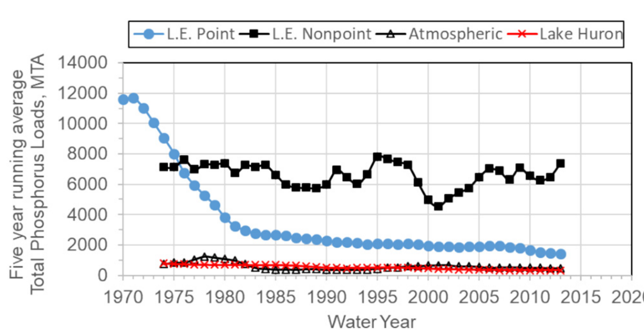

Fig. 1. Trends in total phosphorus loading to Lake Erie from point sources, nonpoint sources, atmospheric deposition and Lake Huron outflows from 1970 to 2013, as reflected in 5-year running average values plotted in year 3 of each 5-year period. Data as presented in Maccoux et al. (2016). MTA = metric tons per annum.

Trends in bioavailable phosphorus export from the Maumee River

The Maumee River, which drains into the southwestern corner of the western basin, has the largest watershed of any river draining into the Great Lakes. Cropland occupies about 73% of the 16,388-km^2 watershed upstream from the Waterville gaging station (Baker et al., 2014a), and it is the largest source of phosphorus entering not only Lake Erie but also the entire Great Lakes system (Kreis et al., 2014). The cyanobacterial blooms that affect the western basin of Lake Erie develop primarily within storm water discharges of the Maumee River with little impact from the Detroit River (Annex 4, 2015). For major Ohio tributaries, the monitoring programs of Heidelberg University's National Center for Water Quality Research (NCWQR) that support long-term TP mass balance studies for Lake Erie include analyses of DRP (https://ncwqr.org/monitoring/). The DRP data, together with periodic studies of the bioavailability of PP using NaOH extractions (Baker et al., 2014a), allow calculations of total bioavailable phosphorus and its dissolved and particulate fractions. The long-term trends (5-year running averages) in spring flow-weighted mean concentrations (FWMCs) and loadings of DRP, BAPP, and total bioavailable phosphorus (TBAP), as well as spring discharge for the Maumee River, are shown in Fig. 2 where it should be noted that:

1. Peak discharges, FWMCs and loads of all phosphorus fractions occurred in the 1979–1980 period.

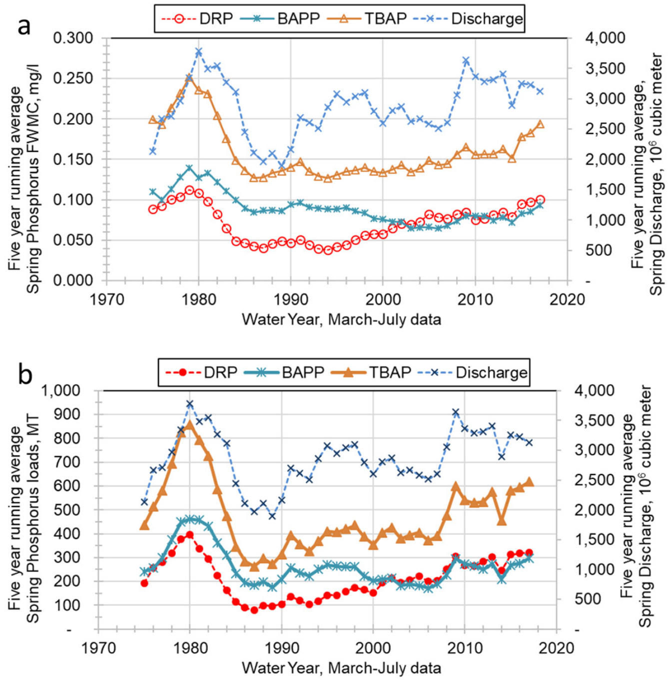

Fig. 2. Trends in spring discharge and flow weighted mean concentrations (A) and loads (B) of dissolved reactive phosphorus (DRP), bioavailable particulate phosphorus (BAPP) and total bioavailable phosphorus (TBAP) for the Maumee River at Waterville, as reflected in 5-year running average values. BAPP calculated as PP × 0.25 and TBAP calculated as BAPP + DRP. MT = metric tons.

2. Between 1979 and 1987, FWMCs of DRP dropped much more than BAPP and these declines occurred after the large declines in point source loads (Fig. 1).
3. Because of the large declines in discharge between 1980 and 1987, loads for all phosphorus forms dropped proportionally more than FWMCs (Richards and Baker, 2002). These declines were accompanied by major decreases in cyanobacterial blooms.
4. From 1994 to 2005, FWMCs of DRP were increasing while those of BAPP were decreasing. During this period, discharges were decreasing and TBAP remained relatively constant.
5. From 2005 through 2014, FWMCs of DRP and BAPP remained relatively constant while discharge increased markedly between 2007 and 2009, accompanied by increases in both DRP and BAPP loads.
6. From 2014 to 2017, FWMCs of DRP and BAPP both increased while discharge decreased slightly and loads of DRP and BAPP increased slightly.

Patterns in discharge, FWMCs, and loads are closely linked (Fig. 2). As a multiplication factor in load calculation, discharge directly affects loads. It also indirectly affects FWMCs in agricultural watersheds through altering the proportions of runoff water, with its high nutrient and sediment concentrations, to baseflow water with its lower nutrient and sediment concentrations (see Electronic Supplementary Material Figs. S1–S4). Agricultural management practices largely affect loads by altering FWMCs of nutrients and sediments exported from cropland. The Task Group recommended that FWMCs be used to track progress toward achievement of target loads because FWMCs are less variable, in terms of relative standard deviations, than either discharge or loads.

The methods and rationale underlying the Annex 4 Task Team target loads

A common procedure for setting target loads is to use an ensemble modeling approach wherein a group of models is used to analyze dose-response curves between the doses of an independent variable (e.g., spring nutrient load) and a dependent variable (e.g., cyanobacteria bloom size) (Annex 4, 2015; LimnoTech, 2013). Input for such curves was provided by the large, weather-induced, annual variations (rather than long-term trends) in spring nutrient loads from the Maumee River (Fig. 3) and the large annual variations in cyanobacteria bloom size in the western basin (Fig. 3). The nutrient load for which the related algal bloom was the largest "acceptable" size was selected as the target load.

For setting target loads related to cyanobacteria blooms in the western basin, the Task Team selected two empirical models – the NOAA Western Lake Erie HAB Model (Stumpf et al., 2012) and the U-M/GLERL Western Lake Erie HAB Model (Obenour et al., 2014), and one process model – the Western Lake Erie Ecosystem Model (WLEEM) (DePinto et al., 2016; Verhamme et al., 2016). For the empirical models, monitored data for both nutrient loads and cyanobacterial bloom sizes were used to calibrate the models and establish targets. For the process model, monitored nutrient inputs were combined with the inner workings of the model to estimate bloom size, following model calibration using monitored dose and response data. To maintain consistency with previous Lake Erie modeling efforts, TP loads were used as the independent variable (x-axis) in all three models. Two major data sets were used as the response variables – a satellite image data set (Stumpf et al., 2012) and fixed-site cyanobacteria sampling program (Bridgeman et al., 2013). Data from the modeling programs converged on a TP load equivalent to a 40% reduction relative to the base year of 2008 load as enough to prevent unacceptable bloom size at discharges up to the 90th percentile. The linkage of the target to a maximum discharge rate recognized that in extremely wet years, it may not currently be feasible for the agricultural community to achieve the target load. Loading during the 2008 water year was chosen as the base load because that year had a high discharge (near the 90th percentile), a Lake Erie TP load near 11,000 MTA, which was the previous target load for Lake Erie, and also had a significant cyanobacterial bloom.

The WLEEM studies included analyses of reducing just the spring DRP load and retaining the PP loads at 2008 levels (Annex 4, 2015). These model runs indicated that reducing the DRP to zero would be insufficient to achieve bloom levels in the mild bloom category associated with the 40% TP load reduction and that additional reductions in PP loading would be required. The U-M/GLERL model-fitting procedures indicated that PP was 50% bioavailable, and the U-M/GLERL modelers also concluded that reducing DRP to zero would be insufficient to attain a mild bloom target (Annex 4 Modeling Subgroup, 2016). These observations were the major factors leading to the Task Team's decision to apply the 40% reduction target to both TP and DRP.

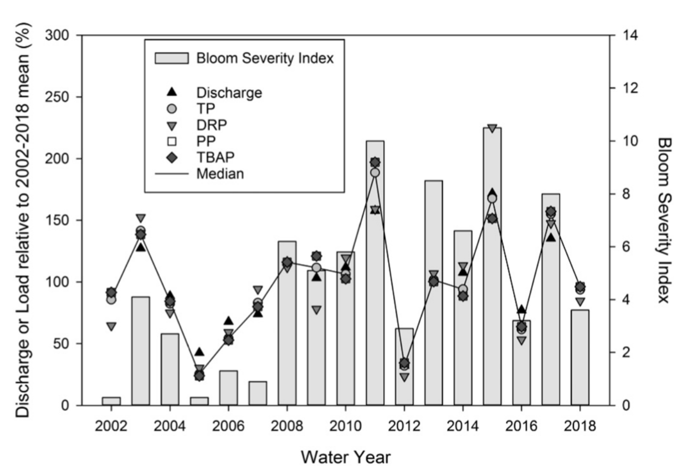

Fig. 3. Correspondence between annual variability of western basin bloom severity (bars) and relative spring discharge and loads of TP, DRP, PP and TBAP (points) when expressed as percentage of average 2002–2016 values for each parameter. Data for the loading and discharge for the Maumee River from 2002 to 2018.

Reasons for revising the target loads

Differential impacts of DRP and PP load reductions on TP and TBAP loads

Various combinations of DRP and PP can reach the 40% reduction in TP but have vastly different effects on TBAP (Table 1). Given the 2008 base year spring loads of TP (1433 MT) and DRP (310 MT) (Annex 4, 2015), the base year loads for PP, BAPP and TBAP were calculated as follows: PP is TP minus DRP, BAPP is 25% of PP, and TBAP is taken as the sum of DRP and BAPP. DRP was considered to be 100% bioavailable and required no further calculation. While subtraction of DRP from TP gives the combined concentrations of PP and dissolved nonreactive phosphorus (DNRP), the DNRP fraction represents only about 5% of the difference (Baker et al., 2014a) and is not considered in this paper. Inclusion of DNRP would result in a slight decrease in PP loads and a slight increase in TBAP loads because DNRP is generally considered to be less bioavailable than DRP but more bioavailable than PP (Sonzogni et al., 1982). The calculated load of TBAP (591 MT) represents only 41% of the TP moving past the Maumee River monitoring station during the spring of 2008.

The load reductions to meet the TP and DRP targets under base year hydrological conditions are equivalent to 40% of their base year loads (Table 1). If both TP and DRP are reduced by 40%, PP, BAPP and TBAP must also be reduced by 40% (Table 1). The 40% load reductions for TP and TBAP are 573 metric ton (MT) and 236 MT, respectively. In 2014, the WLEEM and U-M/GLERL models both indicated that reducing the entire 2008 WY DRP load of 310 MT to zero is insufficient to achieve the mild bloom status at the 90th percentile discharge without additional reductions of PP because the required TP load reduction was 573 MT. However, reducing the DRP load to zero is more than enough to reach the 40% reduction in TBAP load of 236 MT, which should achieve the mild bloom conditions (Table 1). These results suggest that the models were not properly accounting for the differences in bioavailability of DRP and PP.

Multiple combinations of DRP and PP reductions can meet the 40% TP reduction (573 MT). These range from 0% DRP reduction and 51% PP reduction, which results in a 24% reduction in TBAP loads, to 100% DRP reduction and 23% PP reduction, which leads to a 64% reduction in TBAP loads (Table 1). Similarly, multiple combinations of DRP and PP reductions can meet the 40% reduction for TBAP (236 MT). These range from 0% DRP reduction to 84% PP reduction, which combine for a 66% reduction in TP loads, to a 76% reduction in DRP and a 0% reduction in PP, which combine for a 16% reduction in TP loads (Table 1). Clearly meeting a required reduction in TBAP loading could be achieved with widely varying combinations of DRP and PP reductions, with the associated reductions in TP loading ranging from 66% to 16%. The agricultural community should be given the opportunity to select the most economical combinations of DRP and PP reductions for meeting the needed reduction in TBAP loading. That opportunity is largely precluded by the current formulation of target loads where both TP and DRP are given target loads of 40% reductions from 2008 base year loads.

The calculations illustrated in Table 1 assume that the bloom responses within the western basin are proportional to the spring TBAP loads as measured at the Waterville transport station. As such, the calculations assume that there are no “positional” losses of BAPP along the transport pathways to the western basin. If such losses of BAPP do occur, the loads of DRP become even more important drivers of bloom development.

Selection of the independent variable for dose-response curves

In 2014, the Task Team modelers chose to continue the use of TP as the independent variable for the dose-response curves, just as the modelers of the 1970s had used TP for setting an 11,000 MTA target load at that time. Whereas point sources dominated TP loading when the initial targets were set (Fig. 1), nonpoint sources dominate loading now. The issue of bioavailability was far less important then, because point source phosphorus discharges were dominated by DRP.

Years with high loads of TP do tend to have large bloom severity indices while years of low TP loads have smaller bloom severity indices (Fig. 3). There is a strong correlation between bloom severity and TP loads (Stumpf et al., 2016). However, similar correlations occur for bloom severity and loads of DRP, PP and TBAP as well as discharge (Stumpf et al., 2016). The usefulness of dose response models to set targets hinges on causal correlations between doses and responses. At an annual or seasonal scale, the variability in loads of all forms of phosphorus is primarily driven by variations in discharge (see ESM Figs. S1–S4). Because discharge affects loads of all the forms of phosphorus equally in a year or season, there is considerable covariance between phosphorus forms when plotted as a percentage of their longer-term average loads over the study period (Fig. 3). Consequently, the dose response curves are similar for TP, DRP, PP, TBAP and discharge even though the x-axis values for P-forms are very different (Stumpf et al., 2016). If TBAP had been chosen for the dose variable by the Task Team Modeling group in 2014, DRP reductions to zero would have been more than enough to meet a suggested 40% reduction at the 90th percentile discharge (Table 1).

Recently published updates to the models used by the Task Team and a related modeling effort

The decisions of the Task Team were based on the results they received from the modeling subgroup during their deliberations of 2014, as summarized in their final report (Annex 4, 2015). Additional details

Table 1
Base year (2008) March–July loads of total phosphorus (TP), dissolved reactive phosphorus (DRP), particulate phosphorus (PP), bioavailable particulate phosphorus (BAPP), and total bioavailable phosphorus (TBAP) from the Maumee River at Waterville. The ranges of DRP and PP load reductions for meeting the TP target load and for meeting the associated TBAP target load are also shown.

Phosphorus loads	Total phosphorus	Dissolved reactive phosphorus	Particulate phosphorus[a]	Bioavailable particulate phosphorus[b]	Total bioavailable phosphorus[c]
	March–July, metric tons (% of base year load)				
2008 base year loads	1433	310	1123	281	591
Load reductions to meet targets (40% reduction of base year loads)	573	124	449	112	236
Target loads (60% of base year loads)	860	186	674	168	354
Ranges of DRP and PP load reductions meeting the TP target load of 860 MT					
0% DRP, 51% PP	573 (40%)	0 (0%)	573 (51%)	143 (51%)	143 (24%)
100% DRP, 23% PP	573 (40%)	310 (100%)	263 (23%)	66 (23%)	376 (64%)
Ranges of DRP and PP load reductions meeting associated TBAP target load of 354 MT					
0% DRP, 84% PP	944 (66%)	0 (0%)	944 (84%)	236 (84%)	236 (40%)
76% DRP, 0% PP	236 (16%)	236 (76%)	0 (0%)	0 (0%)	236 (40%)

[a] Particulate phosphorus calculated as total phosphorus minus dissolved reactive phosphorus.
[b] Bioavailable particulate phosphorus calculated as 0.25 × particulate phosphorus (Baker et al., 2014a).
[c] Total bioavailable phosphorus calculated as dissolved reactive phosphorus + bioavailable particulate phosphorus.

of the modeling efforts were presented in the Annex 4 Ensemble Modeling Report (Annex 4 Modeling Subgroup, 2016). The models were further developed and described in a special section of the Journal of Great Lakes Research (Scavia et al., 2016). There have been significant changes in the model inputs and outputs between versions available to the Task Team in 2014 and those described in the JGLR special section. Additional calibration years were added to the models. For the NOAA Model (Stumpf et al., 2016), TBAP loads replaced TP loads as the independent variable. TBAP was calculated as 100% of the DRP load plus 26% of the PP load to reflect the chemical bioavailability and an additional 70% reduction in BAPP to account for positional losses of BAPP through deposition to locations no longer accessible to overlying waters where it could support algal growth. These adjustments to the bioavailability of PP were based on the observations of Baker et al. (2014a, 2014b). The resulting TBAP provided a better fit to bloom size than did TP (Fig. 8 of Stumpf et al., 2016) and resulted in major changes to the x-axis such that DRP becomes a much larger driver of bloom size than for a TP-based model.

The 2016 version of the U-M/GLERL model (Bertani et al., 2016) incorporated bioavailability considerations for PP into the independent variables and concluded that such inclusion provided a better fit to bloom size than loading of DRP alone. In the version of the model used by the Task Team (Obenour et al., 2014), TP loads were found to provide better fits to the data than DRP. In the 2016 version, which included 2 additional years of calibration data, the statistical procedures used to estimate bioavailability indicated that 63% of the PP was bioavailable, although they noted that large uncertainty was associated with that estimate. Assuming 63% bioavailability of PP in the 2008 base year, PP would comprise about 2.3 times more loading of bioavailable P than would DRP. The Task Team had concluded that PP and DRP were responsible for roughly equal portions of bioavailable P loading (Annex 4, 2015). An important feature of the U-M/GLERL model is that it predicts an increasing sensitivity of the lake to bioavailable phosphorus loading, such that equal TBAP loads give larger cyanobacteria blooms under 2014 conditions than under 2008 conditions.

In the WLEEM model as presented by Verhamme et al. (2016), an additional year (2014) was added to the calibration/verification period. The graphs comparing dose-response relationships for the effects of reducing TP and DRP loads from the Maumee River on cyanobacteria biomass changed significantly between the report of the Annex 4 Modeling Subgroup (2016) and the Verhamme et al. (2016) paper. Verhamme et al. (2016) state, "… a 60% reduction of SRP (DRP) only is necessary to reach the "mild" bloom threshold, in comparison with a 25% reduction required for the Maumee River annual TP load." This statement differs greatly from the 2014 statements to the Task Team based on the WLEEM model that even 100% removal of DRP loading from the Maumee would not achieve the mild bloom threshold without additional PP removal. Relative to 2008 base year loads, a 60% reduction in DRP loads amounts to a 186 MT reduction while a 25% reduction in TP amounts to a 358 MT reduction. The allocation of the 25% reduction in TP among reductions in PP and DRP is not specified. These relationships are still presented as justification for requiring equal reductions in both DRP and TP, even though the authors state that reducing DRP gives more "bang for the buck" than reducing TP.

One new empirical modeling study has recently been completed (Ho and Michalak, 2017). It uses data from the Landsat satellite to extend existing satellite-based cyanobacteria bloom measurements from 2002 back to 1984, thereby adding 18 years to the calibration/analysis data sets (Ho et al., 2017). The model uses a combination of spring-based DRP loading with long-term (9-year) cumulative DRP loading to explain bloom size, with the cumulative load supporting internal loading rates. The authors conclude, "… we also do not find evidence of TP explaining bloom severity beyond its correlation with DRP." Their model results indicate that internal loading of P is larger than previously expected and that internal loading is proportional to cumulative effects of DRP on phytoplankton community size and related recycling of that biological P rather than on P released from inorganic sediments. Ho and Michalak (2017) also suggest that the cumulative effects of DRP loading explain the increasing sensitivity of the lake to phosphorus loading, as also suggested by the U-M/GLERL Model (Obenour et al., 2014; Bertani et al., 2016). These interpretations indicate that recovery of the lake will be delayed relative to decreases in external loading.

In summary, revision of two of the three original modeling efforts summarized above indicates that DRP reductions alone are sufficient to achieve a mild bloom condition. Only the U-M/GLERL model (Bertani et al., 2016) shows increased role of PP relative to DRP in the development of western basin cyanobacterial blooms. A fourth model (Ho and Michalak, 2017) explains the variability in blooms using current and past DRP loads. However, the models differ significantly regarding the role of internal loading in bloom development and temporal responses to reductions in tributary loading.

Role of internal loading relative to western basin cyanobacterial blooms

Relationships between recent bloom severity and phosphorus loading rates, as well as direct measurements of the diffusive flux of DRP from western basin sediments, do not support internal loading as a significant factor affecting cyanobacterial bloom size. Because of low discharge, the 2016 spring loads of DRP (144 MT) and TP (755 MT) met the 40% reduction target loads shown in Table 1, and the associated bloom fell within the targeted mild bloom category (Fig. 3). Also, the 2018 bloom was smaller than predicted by the 2018 loads (Fig. 3). Discharge volume, independent of its effects on loads and FWMCs, may be more important than loads themselves, since discharge volume affects the areal extent of storm runoff water distribution in the western basin (Stumpf et al., 2016) and typical DRP concentrations in storm runoff water exceed the 0.025–0.030 mg/L DRP concentrations considered necessary to support significant blooms (Christianson et al., 2016; King et al., 2015b).

Direct measurements of internal loading raise doubts about its significance in contributing to bloom size (Matisoff et al., 2016). Those studies suggest that the diffusive flux of DRP from bottom sediments could contribute sufficient DRP to sustain TP concentrations of 3.0–6.3 μg/L in waters flowing through the western basin. The authors note that this represents 20–42% of the 15 μg/L IJC target concentration for waters of the western basin as set in the 1970s. Cyanobacterial blooms develop in storm waters discharged by Maumee River. From 2001 to 2017, the average spring FWMC of DRP discharged from the Maumee River was 81 μg/L, and DRP concentrations during storm water discharges would have been even higher. The new target FWMC for DRP in the Maumee River is 50 μg/L. To assess the significance of internal loading relative to cyanobacterial bloom development, the contributions of internal loading should be compared with Maumee River storm water FWMCs of DRP, not target loads of TP in the western basin, which would reflect mixing of Detroit River and Maumee River flows.

Release of DRP from bottom sediments during wind or wave resuspension events is difficult to measure. Although the total amounts of sediment that are suspended and then redeposited have been estimated to be comparable to annual tributary loading of suspended sediments (Matisoff and Carson, 2014), much of the suspension-deposition may be recycling of the same sediment in contrast with newly introduced sediment from tributaries. The actual amounts of DRP released from the sediment while in suspension depend in part on the concentrations of the sediment in the water column as well as the BAPP content of that sediment and the ambient DRP concentration. Whatever DRP is released from the sediments will be subject to dilution by waters having much lower SS concentrations. In studies of the effects of open lake disposal of dredged sediments, sediment plumes associated with dumping were short-lived and did not cause elevated DRP or TP concentrations within the plumes (U.S. Army Corps of Engineers, 2014).

Trends in DRP and BAPP loading in relation to re-eutrophication of Lake Erie

From 1994 through 2005, FWMCs of BAPP were decreasing at the Maumee sampling station while those of DRP were increasing, leading to relatively equal FWMCs of both by 2002 and thereafter (Fig. 2A). Although loads of BAPP were decreasing and DRP were increasing, the TBAP loads, calculated as the sum of BAPP + DRP loads, remained relatively constant between 1995 and 2006 (Fig. 2B). During this same time, cyanobacterial blooms in the lake were increasing in magnitude (IJC, 2014; Michalak et al., 2013). This suggests that cyanobacterial growth in the lake is more responsive to DRP loads than to TBAP loads. An explanation for this difference may be found in the second aspect of bioavailability – the locational or positional considerations.

While tributary monitoring programs can provide information on the concentrations and loads of DRP and BAPP moving past monitoring stations, they cannot provide information on where the SS bearing the BAPP will settle out of the water column or how much DRP will be released from the BAPP into the water column prior to settling. The rate and extent of these processes depend on the nature of the downstream receiving waters (Boström et al., 1988; Sonzogni et al., 1982) and require detailed local studies. It is 38.3 km from the Maumee River sampling station at the Bowling Green Water Treatment Plant to the mouth of the Maumee River, with the lower 20.3 km representing an estuarine segment of the river. The river empties into Maumee Bay that has an area of 58.6 km^2, average depth of 3.7 m and volume of 0.22 km^3. Large storm events from the Maumee River can discharge more than 1 km^3 of runoff water into the bay and lake, thus refilling Maumee Bay almost five times and sending storm water far into the shallow western basin.

In 2010, the NCWQR conducted a Lagrangian study and analysis of the movement of storm water from the Waterville sampling station to the mouth of the Maumee River (38.3 km), through Maumee Bay, and into the western basin of Lake Erie (Baker et al., 2014b). That study indicated that considerable portions of the SS and PP settle out of the river water prior to reaching the mouth of the river. During low flows, the mixing zone between river and lake water, with its low concentrations of DRP, was located at the river mouth. However, the Maumee River storm water discharges are so high that the storm runoff water displaces the pre-existing Maumee Bay and adjacent western basin waters lakeward during large storms, replacing it with storm runoff water having DRP concentrations of 0.040 to 0.120 mg/L. Mixing zones between lake and storm runoff water were limited to the margins of storm water plumes. The runoff waters inside the plumes retained the same DRP and nitrate concentrations present when those waters passed the Waterville sampling station. In contrast, the SS and PP concentrations were much lower, reflecting their downward movement in the water column and deposition along the flow paths (Baker et al., 2014b).

Because this movement to the river, bay and lake bottom occurs in an aqueous environment with very high DRP concentrations; little of the BAPP is likely to move from the particulate phase into the water column as orthophosphate prior to the BAPP reaching the river or lake bottom. Upon arriving at the bay and lake bottom, the BAPP would contribute to the pool of bioavailable phosphorus that supports internal loading.

It is unlikely that the 2014 version of the WLEEM model was accurately tracking all the above processes that deliver orthophosphate to the successive algal and cyanobacterial communities of the western basin. Inputs into the WLEEM model include the instantaneous concentrations of PP and DRP along with instantaneous discharge at the Waterville sampling station. The internal workings of the model are aimed at determining the amounts of PP that are bioavailable and the amount of orthophosphate that will be released to the water column prior to the settling of the clay particles that convey the PP to the lake. In the modeling report (DePinto et al., 2016), it is noted that the WLEEM model underestimated the deposition of PP between Waterville and the river mouth in both 2012 (modeling report, B7–17) and 2008 (modeling report, B7–22). Thus, the WLEEM model can overestimate the delivery of SS and PP through the lower Maumee River. An orthophosphate release rate from BAPP of 10% per day, based on dual chamber algal bioassays, is mentioned in the Annex 4 report as a model input. That release rate of DRP from BAPP associated with storm event SS was measured with orthophosphate concentration gradients established and maintained by phosphorus-starved algal cells where orthophosphate concentrations were 0.001 mg P/L or less (DePinto et al., 1981; Young et al., 1985). For storm water runoff in the Maumee River, BAPP is suspended in waters having DRP concentrations from 0.040 to 0.120 mg/L (Baker et al., 2014b). A release rate of 10% per day, as observed in bioassays with phosphorus-starved algae and apparently used in the WLEEM model, likely far exceeds the release rate that would occur as sediment settles out of the storm runoff water that has moved into the lake environment. Depletion of DRP concentrations by growing algal communities is likely minimal until turbidity lessens as SS settles out of the water column. Dilution of storm runoff water by pre-existing lake water is limited to the margins of storm water plumes.

Some challenges for DRP load reduction programs

There are numerous challenges associated with reducing DRP export from the agricultural watersheds draining into the western basin of Lake Erie (Kleinman et al., 2011; Kleinman et al., 2015; Sekaluvu et al., 2018; Smith et al., 2018; Wilson et al., 2018). These include:

1. The large loads of DRP moving from cropland through the Maumee River to the western basin represent the cumulative effects of many small DRP losses at the field scale. These losses are typically less than 2% of the maintenance levels of phosphorus fertilizer applied to area cropland (Christianson et al., 2016; King et al., 2015b). As such, these losses are of little economic concern for individual farmers.
2. Subsurface drainage via tile systems is an essential component of crop production in much of the Maumee watershed. Recent research has confirmed that these tile systems represent major pathways for DRP export from cropland (Christianson et al., 2016; King et al., 2015a). Because of their economic benefits to farmers, intensification of these tile drainage systems is continuing, resulting in increased water yields from cropland, greater connectivity between cropland and stream systems, and increased DRP runoff (Kleinman et al., 2015).
3. Utilization of no-till and reduced till cropping systems offer economic advantages to farmers, as well as erosion control benefits. These systems are often accompanied by increased runoff concentrations and loads of DRP because of a build-up of phosphorus soil test levels in the upper layers of the soil (Baker et al., 2017) and development of soil macropores, which support preferential flow of water from the surface to tile systems (Kleinman et al., 2015). In part, increased DRP runoff is an unintended consequence of erosion control practices aimed at reducing particulate phosphorus loading (Jarvie et al., 2017; Smith et al., 2018).
4. Winter cover crops, which are an integral part of the soil health movement and erosion control programs, can increase DRP runoff due to freeze-thaw breakdown of crop residues at the soil surface and related release of DRP (Elliott, 2013; Liu et al., 2013).
5. Past fertilizer application practices have increased the content of plant-available phosphorus in the soil such that current reserves, even when soil test levels are in the maintenance range, could sustain current crop yields for multiple years without addition of fertilizers (King et al., 2017). These legacy pools of phosphorus could sustain DRP runoff at current levels well into the future.
6. Hydrology is a major driver of phosphorus runoff from cropland, as reflected in the large annual variability in DRP and TP export associated with stream discharge. Annual precipitation and spring discharge have significantly increased between 1966 and 2015

(Sekaluvu et al., 2018). Climate change models predict that this region will see increased storm intensities and frequencies, such that DRP loading will increase, given current management practices (Michalak et al., 2013).

Fortunately, these and other challenges are recognized, and systematic efforts are underway to assess and address them, as evident in the Domestic Action Plans.

Domestic Action Plans

Following adoption of the target loads recommended by the Task Team, federal, state and provincial governments have developed Domestic Action Plans (DAPs) that identify appropriate BMPs for reducing DRP and PP loading, as well as the organizations responsible for overseeing their adoption (Ohio Lake Erie Commission, 2018; U.S. Environmental Protection Agency, 2017, 2018). By including a specific target for DRP loading, the Task Team did advance phosphorus management toward its bioavailable forms. Now nutrient management BMPs, such as those included in the 4-R program (Right rate, Right time, Right place, and Right form; IPNI, 2012), occur side-by-side with the erosion control BMPs that were the focus of the nonpoint control programs triggered by the 1984 amendments to the Great Lakes Water Quality Agreement (Great Lakes Phosphorus Task Force, 1986). Additional practices for reducing DRP runoff include the use of: soil amendments such as gypsum, to reduce the solubility of fertilizer phosphorus and improve soil infiltration; blind inlets to reduce phosphorus concentrations in tile drainage; various types of end-of-tile treatment systems to trap nutrients; modifications of drainage ditches to trap sediments and nutrients; and wetland constructions at selected sites along streams and rivers.

The U.S. DAP does note that a paradigm shift will be required for shifting agricultural programs from addressing particulate phosphorus through erosion control programs to dissolved phosphorus (U.S. Environmental Protection Agency, 2018). To bring about such a paradigm shift, the need for it must be evident to all sectors including policy makers, the research and extension communities, the fertilizer industry, farm and environmental organizations, individual farmers, and consumers. Support for that shift is not present in the current phosphorus targets that, in effect, require equal percentage reductions of both DRP and PP. The term "bioavailable" is totally absent in the Ohio DAP and occurs only twice in the U.S. DAP, once where it is linked to the requirement for DRP analyses in sewage effluents from municipal wastewater treatment plants and once in describing the NOAA HAB forecasting program for Lake Erie. In contrast with the U.S. and Ohio DAPs, the Canadian/Ontario DAP does include discussions of bioavailable phosphorus and gives priority to DRP reduction programs (Canada-Ontario Agreement Partners, 2017). The above DAPs, as well as those for Michigan, Indiana, and Pennsylvania, can all be accessed at https://binational.net.

To offer guidance to the implementation programs, the modeling community has used multiple versions (5) of the Soil and Water Assessment Tool (SWAT) model to evaluate sets of BMPs relative to achievement of the target loads (Scavia et al., 2017). Once again, consideration of bioavailability is totally absent from the SWAT assessments of BMP effectiveness which focus on TP and DRP load reductions. Although the SWAT model outputs could be used to calculate separate DRP and PP loads and associated bioavailable phosphorus loads, that step is not included in the modeling summaries (Scavia et al., 2017). Furthermore, the modelers compared 10-year average loads based on weather conditions from 2005 to 2014 with targets applicable to the 90th percentile discharge and, as such, underestimated the challenges of meeting the target loads. Their results do show that individual sets of BMPs have differing effects on DRP and PP reductions, although PP reductions must be inferred through comparisons of TP and DRP reductions. Listed TP reductions do not indicate the proportions of PP and DRP associated with given TP reductions.

Fortunately, state and local organizations, businesses and individuals have been moving forward with renewed efforts to reduce agricultural runoff to Lake Erie as part of their ongoing programs (Wilson et al., 2018). These efforts include utilization of many of the BMPs noted in the DAPs for reducing DRP loading, such as 4R Nutrient Stewardship. Thus, momentum is building for these programs. However, they lack the coordination and funding support that could come from prioritization of DRP reduction within the target loads.

Recommendations for refocusing Lake Erie phosphorus reduction programs and related research

In view of the information summarized in earlier sections of this commentary, we recommend that an Annex 4 Task Team reformulate the target loads so that they specifically consider issues of both chemical and positional bioavailability. For example, a single target could be set for TBAP with specific guidance for its calculation, in terms of weighting factors for DRP and PP. Ideally, the agricultural economics associated with various DRP and PP reductions would be considered in relation to the economic impacts of Lake Erie eutrophication. Assessment of the agricultural economics associated with various DRP control measures is being developed for the Nutrient Tracking Tool (http://ntt.tiaer.tarleton.edu; https://ncwqr.files.wordpress.com/2017/06/saleh-ntt_presentation_saleh_3_7_17.pdf), which has been calibrated with hydrologic and water quality data from USDA-ARS paired edge-of-field studies across Northwestern Ohio (R. Confesor, personal communication, 10 January 2019). Both current and historical research stresses the economic importance of including bioavailability in the design of programs to manage eutrophication (DePinto et al., 1986; Iho et al., 2017; Lee et al., 1980; Sonzogni et al., 1982).

Educational programs regarding bioavailable phosphorus from the viewpoint of eutrophication are warranted. We recommend that future efforts of an Annex 4 Task Team (or perhaps an Adaptive Management Task Team) include input from representatives of all constituencies because all need to be aware of and/or contribute to ongoing adjustments of programs to restore Lake Erie. Additionally, the Lake Erie Basin is the site of many edge-of-field studies that are evaluating BMP effectiveness relative to nutrient and soil losses (King et al., 2018; Williams et al., 2016). We recommend that the effectiveness of these sets of BMPs be evaluated in terms of their impacts on bioavailable phosphorus losses. It is vital that the scientific community associated with the various constituencies speaks with one voice to the multiple communities impacting and benefiting from Lake Erie water quality.

We believe that it is especially important that more information regarding what is known about phosphorus bioavailability relative to eutrophication be conveyed to both agricultural agencies and farmers. Nutrient management by farmers is geared to the content of bioavailable phosphorus in their soils, as measured by soil tests, rather than the total phosphorus content of their soils. The same should be true for nutrient management of our rivers and lakes. While DRP is certainly the focus of much of the discussion of BMPs (Wilson et al., 2018; U.S. Environmental Protection Agency, 2018), the reason it is so important needs to be emphasized to farmers. Soil erosion preferentially removes finer soil particles having higher PP/SS ratios than the field soil (i.e., enrichment ratio concepts); but much of the phosphorus attached to sediment particles is simply not available to either crops or algae. In contrast, DRP losses in runoff water are removing the most bioavailable phosphorus present in the soil. This DRP also turns out to be the most bioavailable phosphorus that moves into the lake to support excessive algal and cyanobacterial growth. Understanding how to reduce losses of bioavailable forms of phosphorus would be beneficial for farmers at an agronomic level with concurrent benefits for downstream water quality.

Together, our hope is that Lake Erie can again become an example of how we can successfully restore a large aquatic ecosystem using the most appropriate, innovative and modern metrics.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jglr.2019.01.011.

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