Western Lake Erie Basin: Soft-Data-Constrained, NHDPlus Resolution Watershed Modeling and Exploration of Applicable Conservation Scenarios

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Abstract

Complex watershed simulation models are powerful tools that can help scientists and policy-makers address challenging topics, such as land use management and water security. In the Western Lake Erie Basin (WLEB), complex hydrological models have been applied at various scales to help describe relationships between land use and water, nutrient, and sediment dynamics. This manuscript evaluated the capacity of the current Soil and Water Assessment Tool (SWAT) to predict hydrological and water quality processes within WLEB at the finest resolution watershed boundary unit (NHDPlus) along with the current conditions and conservation scenarios. The process based SWAT model was capable of the fine-scale computation and complex routing used in this project, as indicated by measured data at five gaging stations. The level of detail required for fine-scale spatial simulation made the use of both hard and soft data necessary in model calibration, alongside other model adaptations. Limitations to the model's predictive capacity were due to a paucity of data in the region at the NHDPlus scale rather than due to SWAT functionality. Results of treatment scenarios demonstrate variable effects of structural practices and nutrient management on sediment and nutrient loss dynamics. Targeting treatment to acres with critical outstanding conservation needs provides the largest return on investment in terms of nutrient loss reduction per dollar spent, relative to treating acres with lower inherent nutrient loss vulnerabilities. Importantly, this research raises considerations about use of models to guide land management decisions at very fine spatial scales. Decision makers using these results should be aware of data limitations that hinder fine-scale model interpretation.

Keywords: Lake Erie; NHDPlus model; Soft Data; Conservation Practice; SWAT-SAS; Model calibration

1. Introduction

For the past two decades, complex watershed simulation models such as the Soil and Water Assessment Tool (SWAT, Arnold et al., 2012), Agricultural Policy/Environmental eXtender (APEX, Williams et al., 2012), and Hydrological Simulation Program-Fortran (HSPF, Bicknell et al., 1997) have been implemented for water resources assessment. These and similar advanced technologies enable modelers to investigate challenges associated with water supply, water quality, pollutant control, and other ecosystem services (Chou and Wu, 2014; Jha and Gassman, 2014; Gebremariam et al., 2014). Model outputs are used to inform land management decision-making and policy development at local to national scales (White et al., 2015a, b; Johnson et al., 2015).

As modeling technologies have been further refined to simulate more complex systems in finer spatial and temporal resolution, parameter inputs required to calibrate the models have also increased in complexity (Haan et al., 1995; Ajami et al., 2007). Model simulations of watershed responses to various agricultural management scenarios in terms of hydrological, sediment, and nutrient processes require a large number of physically/empirically based functions, but monitoring efforts in agricultural systems that could provide those inputs are limited. To address the challenges of providing sufficient input data and calibrating complex models (e.g., SWAT) at a fine spatial scale of resolution, efficient optimization techniques and uncertainty analysis need to be applied to ensure that statistically acceptable results are generated affordably and with limited computational time (Duan et al., 1992; Tolson and Shoemaker, 2007; Vrugt et al., 2009; Yen et al., 2014a). Because of the lack of monitored data to calibrate the model, assumptions are often necessary during fine-scale model development. Caveats associated with these assumptions should be given due consideration when interpreting modeled results.

Although models have traditionally been calibrated with hard data, it is increasingly necessary to use both hard and soft data to achieve model calibration. Non-temporal model outputs and the associated predictive uncertainty that can be stated as aggregated indices of watershed characteristics are considered soft data (Seibert and McDonnell, 2002). Soft data are typically associated with intra-watershed processes, which have been demonstrated to impact model predictions during model calibration (Yen et al., 2014b, c). It is possible to generate model runs that perform well in terms of hard data outputs (model performance can be tentatively categorized by statistical thresholds; Moriasi et al., 2007), but simultaneously produce unreasonable soft data outputs (e.g. a simulated annual denitrification rate twice the rate of measured data in the field; Yen et al., 2014b, c). Soft and hard data should both be carefully considered to ensure that the calibrated model is appropriate for application in further analysis (Efstratiadis and Koutsoyiannis, 2010).

Trade-offs in the relationship between model resolution and model performance (accuracy) are of keen interest in the field of water resources analyses, because these models increasingly serve as land management and policy development decision support tools (Chaubey et al., 2005). This is particularly true in the Western Lake Erie Basin (WLEB), where nutrient management strategies and water quality impacts have been a focus of modeling for decades (Di Toro et al., 1973). Model calibration strategies have necessarily kept pace with model development, enabling enhanced spatial and temporal resolution, so that model predictions are more relevant to field-scale and local policy level decision-making processes (Cotter et al., 2003). However, efforts related to widespread and long-term monitoring of water quality in WLEB have not kept pace with model development. Robertson and Saad (2011) noted that the reduction in tributary monitoring efforts in this region makes evaluation of current nutrient and sediment loadings

difficult. Current monitoring data in the Lake Erie drainage basin is inadequate to measure impacts of agricultural management on water quality (Betanzo et al., 2015) and thus insufficient to support the scales at which models simulating these impacts are currently being applied. This is one reason that the use of soft data is necessary for model calibration and one of the major challenges to model validation.

The SWAT model continues to provide informative analyses at increasingly more refined spatial scales. For example, the National Cropland reports published by the Conservation Effects Assessment Project (CEAP) evaluate the impacts of agricultural management and conservation practice adoption with the APEX (simulation under field scale level) and SWAT (combining outputs from APEX and routing among subwatersheds) models, based on data valid at the 8-digit Hydrologic Unit Code (HUC-8) scale of resolution for cropland in the contiguous USA (USDA-NRCS, 2011; Johnson et al., 2015). The CEAP-Cropland reports inform conservation policy decision making, including the Farm Bill's conservation spending budget. The USEPA (U.S. Environmental Protection Agency) supported Hydrologic and Water Quality System (HAWQS) is a user-friendly online system that applies SWAT to perform scenario analyses in multiple resolution delineation formats, including the HUC-8, -10, and -12 digit scales (USEPA, 2015).

The NHDPlus dataset (National Hydrography Dataset) is the finest resolution watershed boundary unit dataset currently available for large-scale modeling applications; the average size of subwatersheds is 2.6 km². There is only one large-scale watershed modeling implementation of NHDPlus dataset reported to date. Daggupati et al. (2015) applied the SWAT model to compare accuracy of simulation of hydrologic processes between HUC-12 and NHDPlus scales in the Western Lake Erie Basin (WLEB). Building upon the work performed by Daggupati et al. (2015), this study uses both hard and soft data (annual denitrification rate and nitrate (NO₃) loads

contributed from tile drainage system) to develop SWAT models to describe sediment and nutrient (total phosphorus and total nitrogen) dynamics at the NHDPlus scale in WLEB.

In 2002, the Farm Bill passed by the United States Congress included an 80% increase in allocation of federal funds for agricultural conservation relative to the funding amount in the previous Farm Bill (Johnson et al., 2015). CEAP was initiated by the USDA to conduct credible scientific evaluation of the benefits derived from the increased use of federal financial resources for agricultural conservation (Mausbach and Dedrick, 2004). A major goal of CEAP is to quantify the impact of agricultural conservation practices on water quality (e.g. sediment, nitrogen, phosphorus, and pesticide residues). The focused and scientifically credible outcomes of these analyses can be used to help inform future conservation practice and program investments, and may serve to guide conservation research inquiry and investments (White et al., 2014).

The impetus behind the development of this high resolution modeling project was a need expressed by the Conservation Effects Assessment Project (CEAP) - Wildlife Component for a product capable of accurate predictions of nutrient and sediment dynamics in the many very small streams systems across WLEB in order to conduct aquatic stream health analyses. SWAT model outputs developed from simulating the watershed at this scale could be used as data inputs in biological models, such as those used to assess stream health and inform stream conservation decisions (Keitzer et al., 2016). A principal caveat associated with this work is that while the model may be capable of producing appropriately scaled and reasonable-seeming results, insufficient quantitative data exist against which to compare the various model outputs at this scale in order to determine whether the simulated spatial specificity is reflective of reality across

the entire simulated WLEB. More monitoring is necessary in order to provide sufficient data for complete model validation.

This implementation of the SWAT model in a sophisticated watershed analysis of WLEB at the finest available, NHDPlus spatial resolution enhances both model performance and the predictive capacity associated with cross-field investigations, enabling better modeling results in the future. In considering the increasing interests in water quality issues in WLEB, this study reports on a unique, novel, and timely application of the SWAT model, in which NHDPlus resolution data are used to: (1) explore soft/hard data applications on NHDPlus scale model development and subsequent predictions of streamflow, and sediment and nutrient dynamics associated with agriculture in the WLEB; (2) demonstrate the utility of modeling at this scale to describe the possible costs and benefits of various conservation practice adoption scenarios in the WLEB. The most important reason that the NHDPlus stream network was adopted in this CEAP-Wildlife study is because the fish community research associated with potential/current contaminants requires detailed information such as simulated flow, sediment, and nutrients at the stream reach scale. In addition, the approach conducted in this study that pairs conservation scenarios with projected additional costs is particularly rare in literature, since that requires mutual work among modelers (engineers), biologists, and economists. Spatially explicit simulations like these provide important information to consider when managing agricultural lands - knowledge gaps could be exposed that focused research efforts could help resolve.

2. Materials and Methods

2.1. The SWAT Model

The SWAT model was used to simulate streamflow and water quality in WLEB (Figure 1). SWAT is a process/empirical-based, quasi-distributed, continuous time-step model developed by the United States Department of Agriculture – Agricultural Research Service (USDA-ARS) to perform large-scale watershed simulations based on specified soil, landuse, weather, and topographic data (Arnold et al., 2012). Potential impacts of current and changing land management and conservation practice adoption on ecological goals and concerns can be evaluated by simulating scenarios and analyzing predicted impacts on hydrologic, sediment, and nutrients processes (Daggupati et al., 2011; Keitzer et al., 2016). SWAT is one of the most broadly applied large-scale watershed simulation models in the field of water resource planning and management (Gassman et al., 2007) and is among the toolsets used to inform conservation practice policy within the USA (Johnson et al., 2015; Mausbach and Dedrick, 2005).

The publicly available version of SWAT is currently parameterized at three spatial scales: (1) basin level; (2) subbasin level; and (3) HRU (Hydrologic Response Unit) level. HRUs are a unique combination of slope, landuse, and soils; there can be many HRUs per subbasin whereas parameters for a HRU can be very specialized by user discretion. The HRU level parameters are specifically assigned to all HRU files so that the parameter values may be altered by the modeler for selected HRUs. The subbasin level parameters are assigned uniformly within a given subbasin, and the basin level parameters are uniformly applied to the entire basin defined by the SWAT project. It was stated that SPCON (linear parameter to adjust sediment load in channel sediment routing (Neitsch et al., 2011)) is one of the most sensitive parameters involved in sediment and sediment-associated phosphorus calibration and calculating transportable

maximum sediment load (Chu et al., 2004; White and Chaubey, 2005). However, SPCON is a parameter specified at the basin level, so it is possible for SWAT simulations to over- or underestimate the basin sediment and sediment-associated phosphorus loads, especially because most watersheds do not have homogeneously distributed topography (Neitsch et al., 2011). To improve model predictions of sediment and sediment-associated phosphorus dynamics in this study, a revised version of SWAT, named SWAT-SAS (Subregional Adjustment of Sediment) was developed, which allows SWAT users to specify SPCON values for each subbasin within the basin of interest. These subbasin specific adjustments in the *.rte files enhance the spatial accuracy of sediment and sediment-associated phosphorus predictions. More details of SWAT-SAS are described in Appendix A.

2.2. Soft-Data-Constrained Calibration

Large-scale watershed modeling efforts with sophisticated streamflow, sediment, and nutrient dynamic simulation processes necessarily require the incorporation of a large number of model parameters, some of which are difficult to populate with currently available datasets. Various parameter estimation (or, calibration) techniques have been proposed to solve challenging high-dimensional watershed calibration problems (Yen et al., 2014a). The parameter estimation process (e.g., automatic or manual calibration) is designed to minimize the error term between the observed and simulated data in a given time series (e.g., daily streamflow, monthly sediment load). The temporal data used to calculate error statistics during calibration process can be categorized as hard data (Seibert and McDonnell, 2002). Modelers can use manually defined statistical guidelines, such as the General Performance Ratings (Table 2), to evaluate the performance of error statistics (Moriiasi et al., 2007). However, it was stated in literature that

watershed simulation outputs may be inconsistent with real hydrological mechanisms and/or intra-watershed processes (Yen et al., 2014b).

In addition to hard data in the form of temporal series, soft data are defined as non-temporal measures of a watershed in reflecting intra-watershed processes such as denitrification, average annual sediment loading, or ratio of nitrate attributed from subsurface versus surface flow (Yen et al., 2016). Developing calibration routines without due consideration of soft data may produce excellent model outputs in terms of error statistics, but the simulations may violate actual watershed behavior. For example, in a case study at the Eagle Creek Watershed, Indiana, USA, an auto-calibrated SWAT model without consideration of the region's prevalent tile drainage system might provide accurate stream flow predictions, but would incorrectly attribute the majority of flow to surface runoff losses (Yen et al., 2014b). The presence of the tile drainage system requires the modeler to apply additional constraints to the model simulation rather than relying on the auto-calibration system to correctly designate flow to the appropriate loss pathways (Yen et al., 2014b).

In much of the Midwest region (MWR) of the USA, tile flow pathways contribute to significant subsurface flow pathways; if model results are used to inform conservation practice adoption decisions or conservation-related policy, overlooking the significance of this loss pathway could lead to inappropriate land management decisions. In addition, the associated predictive uncertainty may also be affected considerably (Yen et al., 2014c). It was indicated that the use of soft data may have direct impact on both parameter and predictive uncertainty. Some relevant parameters may not explore the full spectrum of given ranges because of the incorporation of additional soft-data-constraints (however, exploration of uncertainty of hard/soft data is not the primary goal in this study so it will not be fully investigated). In this study, soft-

data-developed constraints were added to the denitrification rate and amount of nitrate (NO_3) contributed from tile flow to help govern the simulated intra-watershed processes in the region. The denitrification rate is constrained to be less than 50 kg/ha (David et al., 2009) and the ratio of NO_3 contributed by tile flow is constrained to be no less than two-thirds of total NO_3 losses (Schilling, 2002). For auto-calibration routines in practice, projected (or proposed) candidate parameter sets are generated by the current best solution evaluated by statistical performance. By the incorporate soft-data-constrained calibration, simulated results which violate the two constraints are automatically rejected and will not be considered as the candidate parameter set. In this case, the finalized calibration results will follow the constrained watershed behavior automatically. The approach of soft-data-constrained calibration guarantees that quality of derived model outputs to be representative to the real world in terms of better reflecting hydrological and water quality processes. On the other hand, modelers and engineers can take advantage of this approach to avoid the risk of generating good modeling results for the wrong reasons.

2.3. Study Area Description

The WLEB is located in the Midwestern United States of Ohio (76%), Indiana (17%) and Michigan (7%), has a drainage area of 23,817 km^2 , and is bounded by the Raisin River in the north and the Sandusky River in the south (Figure 1). Elevation changes in the WLEB are fairly mild (elevation varies from 246 m to 387 m above sea level), with an average slope of around 2%. The mean winter temperature is -5 °C and temperatures can rise to 29 °C in summer (available from the National Oceanic and Atmospheric Administration (NOAA) Cooperative Observer network and Weather- Bureau-Army-Navy stations). Average annual precipitation ranges between 838 and 940 mm. The major land use in WLEB is cultivated cropland (70%),

dominated by corn-soybean rotations; forest (12%) and urban (12%) lands comprise the other dominant land uses in the region. A significant portion of the agricultural lands in WLEB, like most agricultural lands across the MWR, are installed with well-organized tile drainage systems, which enable agricultural activities (Kaatz, 1955) in a region previously dominated by a wetland (4,000 km²) called the Great Black Swamp (Mitsch et al., 2007). Although agricultural tile drainage is used in other regions of the globe, such as Canada and Europe, the extensiveness and level of sophistication of the MWR's large-scale agricultural tile drainage system is unique. Model simulations of the impacts of various land management scenarios on conservation concerns in this unconventional watershed must consider both natural and anthropogenic influences on the mechanisms of surface and subsurface flow (groundwater, tile drainage flow) in order to assess the actual watershed behavior.

2.4. Model Setup/ Incorporation using NHDPlus Data

Two SWAT models with different spatial resolutions were developed for the WLEB region: (1) a HUC-12 scale model; and (2) an NHDPlus scale model, at the 30-meter Digital Elevation Model (DEM) for topographical features (Figure 2). The HUC-12 model was built on the Watershed Boundary Dataset (WBD) (1:24,000 scale-level) with 391 subbasins, which is available online (<http://datagateway.nrcs.usda.gov>). The sizes of subbasins in the HUC-12 model ranged from 25 to 191 km² (average: 72 km²). The WBD dataset is conjointly synchronized by the United States Geological Survey (USGS), United States Department of Agriculture – Natural Resources Conservation Service (USDA-NRCS), and the United States Environmental Protection Agency (USEPA). The NHDPlus dataset is coordinated by USGS and USEPA; associated data (at 1:100,000 scale-level) is available online (<http://www.horizon-systems.com/NHDPlus/index.php>). The NHDPlus model includes 11,128 subbasins, which range in size from

0.001 to 80 km² (average: 2.6 km²). Daily streamflow calibrations in the NHDPlus version were developed by Daggupati et al. (2015).

SWAT models were set up using the Geographic Information System (GIS) interface of ArcSWAT2012 (rev593) (supported by ArcGIS10.1). Since it is time-consuming to execute a SWAT project in NHDPlus resolution (e.g. 10 hours to complete a single 13-year SWAT run using an Intel ® Core™ i5-2500k CPU @ 3.30 GHz, 64-bit operating system, Microsoft Windows 7 Professional), a HUC-12 resolution SWAT model of WLEB was built and calibrated for streamflow (each iteration consumes less than 2 hours for the same running period in NHDPlus). The calibrated streamflow parameters were then transferred from the HUC-12 designed model to the NHDPlus project. Details can also be found in Daggupati et al. (2015).

Soils data were acquired from the publicly available USDA-NRCS State Soil Geographic database for the Conterminous United States (STATSGO at 1:250,000 scale-level) (USDA-NRCS, 1995). The simulated land uses were derived from the 30-meter resolution Crop Data Layers (CDLs), which include major rotations of local crops (Srinivasan et al., 2010). The slope information is incorporated with the soil and landuse data by HRU, where three slope categories (0~2%, 2~5%, and >5%) were defined, with the threshold area of 50 hectares for each. In addition, row-crop agriculture was represented in HRUs without implementing pre-defined thresholds. The total number of HRUs for the HUC-12 and the NHDPlus models was 13,156 and 34,807, respectively.

Management practice data for agricultural activities were obtained from multiple sources. Cropland tillage practices were available from USGS survey data (Baker, 2011). Operation schedules designed for the RUSLE2.0 erosion model were acquired from USDA-ARS (Foster, 2005). Fertilization rates were obtained from National Agricultural Statistics Service (NASS)

reports (USDA-NASS, 2014) of average crop yield at the county level. The assembled data were organized in the format of SWAT management files by a software package in VB.net (code available upon request). The management operations for agricultural activities (e.g., harvest/kill operations) were defined by date, and heat units were applied to those without data. In SWAT operation, historical precipitation was used for the years being simulated (e.g., from 1990 to 1999). On the other hand, it is very difficult to obtain the corresponding management details in each and every year in practice. Therefore, the projected operation schedule was implemented to reflect the management operation during simulation. Since spatial data on tile drainage systems are not publicly available, it is not feasible to allocate tiles specifically to currently drained field sites. Therefore, to accommodate for the impacts of tile drainage, the existence of tile drainage was assumed on all agricultural soils with less than 1 percent slope and soils categorized as poorly drained per the Soil Survey Geographic database (http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/soils/_home). The HUC-12 and the NHDPlus SWAT models use the same temperature and precipitation data (from 1960 to 2010) inputs, available from the Weather Bureau Army Navy (WBAN) and Cooperative Observer Network Stations collected by National Oceanic and Atmosphere Administration (NOAA).

Available measured streamflow, sediment, and water quality record data were used to conduct model calibration (1990-1999) and validation (2000-2006) (Table 1). More details on streamflow calibration in the NHDPlus scale model can be found in Daggupati et al. (2015). Daily loads for total nitrogen, total phosphorus, and total suspended solids data used in calibration and validation of nutrient and sediment processes were estimated from measured USGS values with the Fluxmaster load estimation program (Schwarz et al., 2006; Robertson and Saad, 2011) (Table 1). The LOAD ESTimator (LOADEST) program (Runkel et al., 2004) was

used to estimate daily loads for soluble phosphorus and NO₃ based on measurements of grab samples taken at the gauging stations (Table 1).

2.5. Model Calibration/Validation

SWAT has been applied in WLEB at coarser scales than the analyses presented here, including 350 subbasins and 4,416 HRUs in total; (Bosch et al., 2011 and 2013) and 4-digit HUCs (USDA-NRCS, 2011). In this study SWAT simulated 391 subbasins and 13,156 HRUs at HUC-12 scale; 11,128 subbasins and 34,807 HRUs at NHDPlus scale.

The potential impacts of nitrogen, phosphorus, and sediment loadings on aquatic species can be evaluated through standardized indices such as IBI (Index of Biotic Integrity) and dissolved oxygen in a given water body. Excess nitrogen and phosphorous concentrations in freshwater can pose a serious threat to freshwater biodiversity and many of the valuable ecosystem services provided by freshwater ecosystems (Carpenter et al., 1999, Smith et al., 1999). For instance, nutrient loading from WLEB watersheds has contributed to eutrophication of Lake Erie and threatens commercial and recreational fisheries, tourism, and the supply of safe drinking water (Michalak et al., 2013, Kane et al., 2014). In the United States, nutrient enrichment is a major contributor to degraded stream conditions (USEPA 2006). Water quality (drinkability, fishability, swimability, etc.) issues in WLEB continue to concern municipalities and the general public. In 2014, potentially 500,000 water users faced a drinking water crisis in and around Toledo, Ohio, USA related to an algal bloom, a symptom of eutrophication (Dungjen and Patch, 2014). This study provides scientific reference to nitrogen, phosphorus, and sediment dynamics, which play a role in the development of harmful algal blooms (HABs) and toxicity of the blooms. Although the drivers of HABs and their toxicity are poorly understood, they are clearly an emerging threat to freshwater systems (Brooks et al., 2016.). This report and the

SWAT model development associated with it may help inform land management and policy decisions to help reduce the likelihood and/or frequency of recurrences of similar emergency situations.

As mentioned earlier, SWAT streamflow calibration and validation in WLEB was conducted in previous work (Daggupati et al., 2015). Weather data used for the calibration period for the nutrients and sediment dynamics was from 1990 to 1999, with a three-year additional simulation warm-up period (1987 to 1989); the validation was conducted on 2000 to 2006 data. Since the NHDPlus project was computationally demanding, the sediment and nutrients calibration of WLEB was conducted manually by expert judgement.

To evaluate the model performance under the developed calibrations, two quantitative error statistics were used: the Nash-Sutcliffe Efficiency Coefficient (NSE) and the Percent Bias (PBIAS). The NSE (Nash and Sutcliffe, 1970) has been implemented in a wide variety of watershed modeling topics (Servat and Dezetter, 1991; ASCE, 1993) and has proven effective at predicting reasonable temporal outputs, especially on seasonal peaks (Santhi et al., 2001). The PBIAS statistical measure, on the other hand, performs better at capturing trends in consistent watershed responses, such as average flow rate, which is important in maintaining stabilized fish populations (USDA-NRCS, 2014). The potential magnitudes of NSE range from $-\infty$ to 1. Perfect matches between model predictions and observation data are indicated when NSE equals one. On the other hand, the best PBIAS value is 0%; underestimation is apparent when $PBIAS > 0\%$, and overestimation is signified by $PBIAS < 0\%$. General performance ratings of NSE and PBIAS values follow the convention developed by Moriasi et al. (2007) (Table 2).

2.6. Conservation Scenarios

The CEAP-Croplands report on the Great Lakes region reports on conservation practices and agricultural practices in use from 2003-06 (USDA-NRCS, 2011). The report classifies all cropland acres into “needs classes” based on inherent vulnerabilities to leaching and or runoff and levels of conservation treatment. In addition to analyses of “current conditions” simulations and their impacts on conservation concerns, the CEAP-Croplands National reports contain analyses on modeled hypothetical conservation scenarios that estimate potential impacts of various conservation strategies associated with prioritizing treatment of acres according to “needs classes” (USDA-NRCS, 2011). Simulation models were used to estimate potential benefits and costs associated with applying prescribed conservation practices to any or all of the classes of acres. Per the CEAP-Croplands classification conventions, “Critical Needs” acres have a high need for additional conservation treatment; these acres include the most vulnerable of under-treated acres, those with the fewest conservation practices in place, where the highest losses of sediment and/or nutrients can be expected if no further conservation actions are taken. “Moderate Needs” acres have an intermediate need for additional conservation treatment; these acres are under-treated, but have lower inherent levels of vulnerability to losses or have some effective conservation practices in place. “Low Needs” acres have low inherent vulnerabilities to losses, or are adequately treated to address those vulnerabilities; these acres may still suffer annual nutrient and sediment losses that could be lessened with additional treatment. Further, current treatment levels on Low Needs acres must be maintained in order to ensure low nutrient and sediment losses in the future.

One of the objectives is to demonstrate the potential outcomes of watershed modeling at NHDPlus resolution by incorporating possible costs via different scenarios of conservation

practices. Therefore, settings of parameters in each scenario are the same in order to conduct less biased comparisons. The major differences in each scenario are the level of nutrient reduction by alternated conservation practices applied (e.g., erosion control, nutrient management) and resultant impacts on losses of nutrients and sediment from farm fields. The scenarios were selected based on scenarios that CEAP-Croplands had applied in previous reports (USDA-NRCS, 2014, 2015, and 2016). Using these scenarios broadens the potential implications of the conducted analyses, as it allows for future comparison across regions that were similarly simulated in CEAP-Croplands reports. In this study, eight simulations were performed with the SWAT model developed for these analyses. The calibrated SWAT model represents the Baseline condition (BL) and serves as the basis from which the other scenarios were developed. The conditions observed in the BL scenario are the result of extensive conservation practices already in use in the region. Every cultivated cropland acre in the WLEB region was treated with an average of 1.8 conservation practices in 2003-06 and 2.4 conservation practices in 2012. Conservation practices in place in WLEB in 2003-06 represented a \$208 million annual investment (\$43.39 per acre), whereas conservation practices in place in 2012 represented a \$277 million annual investment (\$56.98 per acre; USDA-NRCS, 2016). The data on practices and management used to develop the BL were primarily from the time period between 2003-06 and 2012.

Conservation treatment types were classified into two groups: erosion control practices alone or erosion control practices in conjunction with nutrient management practices. Simulation of six treatment scenarios explores the potential impacts of treating acres in particular needs classes with one or both treatment practices at the watershed scale. Acres in each of the three needs classes defined in the CEAP-Croplands report on the Great Lakes (NRCS-2011) were

361 treated with either structural practices or both structural and nutrient management practices
362 (Table 3).

363 An additional Grass Background (GBG) scenario was simulated to provide context for the
364 predicted impacts of “current” agricultural management. The GBG scenario provides an estimate
365 of sediment and nutrient losses in the absence of agriculture (replaced by grass cover). The eight
366 simulated scenarios were analyzed to provide estimations of annual financial costs and
367 conservation impacts of various watershed-scale conservation strategies (Table 3 and Figure 4).

368 The primary conservation practice cost data source was the 2010 official state USDA-NRCS
369 Payment Schedule database, augmented with cropland rental rates, commodity prices, and
370 fertilizer prices from other published sources (USDA-NRCS, 2012). The NRCS National
371 Conservation Plans (NCP) database was used to estimate the average number of units of practice
372 per protected acre by state and practice. Rates from the NRCS Technical Service Provider
373 database were used as a proxy for Technical Assistance (TA) costs. Since a farm field may have
374 a variety of practices applied, each with a different useful life span, a special Equivalent Net
375 Annual Value formula was used to amortize the cost of each practice, sum the costs across
376 practices for each sample point, and calculate the annualized cost per acre of the treatment
377 alternative (Boardman, et al., 2001). The costs used here include the full cost of planning,
378 installation, maintenance, and forgone income on land converted to conservation cover,
379 regardless of whether some or all of the cost would be partially reimbursed to the farmer or
380 incurred by a non-farm entity such as a federal agency. State average cropland rental rates were
381 used as the cost of land converted from active crop production to a conserving use such as filter
382 strips or buffers (USDA-NASS, 2011). All conservation practice costs were converted to units of
383 practice per protected acre, using data from the NCP database. For example, the NCP data may

show that on average 1.6 acres of buffer strip are used per 40 acres of cropland field, for a ratio of 0.04 acres of buffer practice per acre of protected cropland. If the buffer strip annualized cost per acre of buffer was \$100, then for this example, the modeled cost would be \$4 per protected acre ($\100×0.04).

2.7. Potential Biological Applications

The potential for conservation practices to improve stream biological conditions was explored by quantifying the amount of streams potentially degraded by excess nutrients in each conservation scenario. Excess nutrients are a pervasive threat to stream biodiversity (Richter et al., 1997; Dudgeon et al., 2006; Vörösmarty et al., 2010). For example, recent surveys suggest that excess nutrients are potentially degrading stream biological conditions in >40% of the rivers and streams in the United States (United States EPA, 2016). For the purposes of this study, a stream segment was classified as degraded if average annual total nitrogen (TN) or total phosphorus (TP) concentrations were above established nutrient criteria shown to increase primary production excessively and alter stream algae, aquatic insect, and fish communities (Evans-White et al., 2013). A stream segment's average concentration was calculated as the mean of median annual concentrations from the 21-year simulation for each conservation scenario. Outcomes for each conservation scenario were compared to the baseline scenario to assess the benefits provided by that scenario.

A number of nutrient criteria have been developed to assess stream health and while there is some agreement in the general magnitude of threshold values, the actual thresholds can vary by more than 6-fold (Evans-White et al., 2013). Nutrient criteria are particularly influenced by the statistical method (e.g., percentile approaches, 2DKS, regression tree), location (e.g., ecoregion), and biological response (e.g., algae, aquatic insects, or fish) used in their development (Evans-

White et al., 2013). To account for this variability explicitly, 43 different nutrient criteria from a recent review of stream nutrient criteria in the United States (Table 4 in Evans-White et al., 2013) were used to assess potential stream degradation. These nutrient criteria included thresholds for both TN and TP based on empirical observations and were developed using a variety of statistical methods and biological responses (Evans-White et al., 2013).

A linear mixed model was used to assess the return on investment of different conservation practice types (erosion control only vs. erosion control and nutrient management). This model included the different nutrient criteria as random intercepts, and the additional cost of a conservation scenario, practice types included in a conservation scenario, and their interaction as fixed effects. The model used was:

$$y_i = \alpha_{j(i)} + \beta_1 X_{i1} + \beta_2 X_{i2} + \beta_3 X_{i3} + \epsilon_i$$

$$\alpha_j \sim N(\mu_\alpha, \sigma_\alpha^2)$$

where in the first level of the model y_i is the percent of the watershed improved in conservation scenario i using nutrient criteria j as a function of practice types included X_1 , the amount of additional investment X_2 , and their interaction X_3 . In the second level of the model, nutrient criterion α_j was normally distributed with a mean of μ_α and residual variance of σ_α^2 across all nutrient criteria. The effects of cost, practice types included, and their interactions were considered significant if the bootstrapped 95% confidence intervals of the estimates did not include zero (replicates = 10,000). All models were fit using the lme4 package (Bates et al., 2014) in the R statistical environment (R Core Team 2014).

3. Results and Discussion

3.1. Results of Calibration/Validation

As it was stated previously that watershed simulation of the NHDPlus project is an extremely time-consuming task. Therefore, a HUC-12 model was initiated and calibrated for streamflow in advance of the development of the NHDPlus scale model (Daggupati et al., 2015). One of the most important conclusions was made by Daggupati et al. (2015) was that some subbasins are similar to other adjacent ones in terms of geographical differences and also modeling behavior. Therefore, a total of five major subregions were categorized to conduct the following computationally expensive NHDPlus project. Five representative gauge stations (Daggupati et al., 2015) selected for watershed calibration and validation are shown in Figure 1 and Table 4. According to the General Performance Ratings (Table 2), the daily streamflow calibration and validation can be considered between “Satisfactory” and “Good” (calibration: NSE: 0.54 ~ 0.87; PBIAS (%): -11.76 ~ 22.70; validation: NSE: 0.43 ~ 0.88; PBIAS (%): -26.07 ~ 25.25). In general, higher statistical standards are expected for streamflow calibration since flow is the fundamental medium of hydrologic, sediment, and nutrient processes in watersheds. However, streamflow calibration could be compensated for by sediment and nutrient mechanisms, especially in the NHDPlus resolution. It is extremely difficult to achieve “Very Good” simulation of streamflow processes while concurrently calibrating for other output variables to be within appropriate ranges in terms of statistical performance.

Statistical metrics for monthly sediment calibration and validation ranged from “Good” to “Very Good” (calibration: PBIAS (%): -10.43 ~ 18.63); validation: PBIAS (%): -35.01 ~ 35.46). It is important to have sediment processes well-calibrated because of the interaction between sediment and total phosphorus dynamics, due to sediment-associated phosphorus. Results for

monthly total phosphorus calibration and validation were “Very Good” (calibration: PBIAS (%): -12.78 ~ 8.42; validation: (PBIAS (%): 0.61 ~ -22.74). Model predictions for monthly total nitrogen ranged from “Satisfactory” (validation at St. Joseph) to “Very Good” (all other stations).

Model performance varied between individual gauging stations. The model provided better performance for the Maumee River gauge (“Very Good” results in all output variables) compared to the other stations (temporal results of streamflow, sediment, and nutrients are shown in Figure 3). Among the five selected gauge stations, the Maumee River Station is the most representative outlet, since it receives 68.3% of the whole drainage area. In addition to hard data calibration, soft data used in these analyses, can have substantial impact on model predictions (Seibert and McDonnell, 2002; Yen et al., 2014b). The biological analyses in this study required simulation results which mimic realistic watershed responses. To simulate actual watershed behavior correctly, denitrification rate (DENI) and NO_3 contributed from tile flow (SSQ_Ratio) were constrained within certain ranges ($\text{DENI} \leq 50 \text{ kg/ha}$; $\text{SSQ_Ratio} \geq 0.6$). The calibrated SWAT model had a DENI of 23.31 kg/ha and SSQ_Ratio of 0.72 (72% of NO_3 attributed to subsurface flow pathways) and both soft data outputs were reasonable (in quantity and in ratio) according to the available literature (Schilling, 2002; David et al., 2009).

As mentioned previously, the NHDPlus SWAT simulation of WLEB is exceptionally computationally expensive. It is not feasible to conduct thousands of model simulations during parameter estimation. Instead, expert judgement was used, especially for sediment and nutrient calibration. Therefore, necessary compensation (trade-off) has been made during the calibration process.

3.2. Applications of Conservation Scenarios

The conservation strategies simulated here explore potential conservation gains and associated financial costs that may be expected from the application of various suites of practices to various acres (Figure 4, Table 3). The application of structural practices that provide erosion control without complementary nutrient management practices (ECC, ECA, and FT; Table 3) is much less expensive than is inclusion of structural practices and nutrient management. Application of structural practices on Critical Needs, All Needed, or all cropland acres, (ECC, ECA, and FT scenarios) could be implemented at 54, 37, and 49 percent of the respective costs required to supplement structural practices with nutrient management practices on these same acres (NMC, NMA, and NM, respectively).

The NM scenario suggests adoption of erosion control and nutrient management on all cropland acres could reduce annual loads of sediment being lost from croplands by up to 45 percent, total nitrogen by up to 41 percent, nitrate nitrogen by up to 32 percent, total phosphorus by up to 54 percent, and dissolved phosphorus by up to 28 percent (Figure 4), but adoption of NM does not alleviate all of the sediment, nitrogen, or phosphorus conservation concerns in all of the streams. Even simulated elimination of agriculture from the landscape and simulated conversion all agricultural lands to vegetated easements (GBG scenario) did not reduce any nutrient or sediment losses by more than 80 percent, suggesting that other land uses currently contribute to sediment and nutrient loads (Figure 4). Restoring stream health across the region will likely require efforts from all stakeholders and not just the agricultural community.

Maintaining current conservation and applying nutrient and structural erosion treatment to all agricultural acres in WLEB would cost around half a billion dollars every year. These funds are not currently available; therefore, the idea of prioritizing conservation spending by targeting

Critical Needs acres for treatment before all other acres has been posited. However, an important and interesting finding of this work is that treating acres classified as Low Needs provides substantial benefits in almost all cases, relative to treating only Critical Needs or Critical and Moderate Needs acres. These simulations suggest that an effective conservation strategy for cropland acres in WLEB should consider implementation of improved conservation practices on *all* cropland acres in the region, including those that have a low inherent vulnerability to losses and those that are already well treated.

It should be noted that the conservation scenarios explored here were developed to reduce edge-of-field losses, not to meet the needs of the streams or the ultimate receiving water body, Lake Erie. Further, these scenarios were not developed to represent the most cost effective policies to reduce nutrient and sediment loads to the streams or to Lake Erie. Land managers and communities must identify their conservation goals and develop comprehensive plans to achieve those goals; these plans will likely include both on-field and off-field conservation practices.

As expected, targeting treatment to critical needs acres provides the largest conservation return per dollar investment. Critical Needs acres can be treated with erosion control (ECC) at 4 percent of the investment required to treat all acres with erosion control (FT) and provides 13 percent of FT's annual sediment conservation benefits. All Needed acres can be treated with erosion control (ECA) at 43 percent of the cost of FT and will provide 56 percent of FT's annual sediment loads reduction benefits. Therefore, although low needs acres have the potential to provide nearly half of all possible sediment reduction benefits, there is a higher cost per unit of sediment reduction benefit on these acres than on Critical Needs and Moderate Needs acres.

Estimations of the annual reductions in sediment loads lost from agricultural acres in WLEB under alternative scenarios are shown in Figure 4(A). As would be expected, structural erosion

control practices have a significant impact on reduction of sediment loads. Increasing treatment to include nutrient management practices drastically increases the cost of treatment, but provides no benefit to sediment loss reductions (Table 3, Figure 4). For example, all acres can be treated with erosion control practices (FT) at 49 percent of the cost of treating all acres with erosion control and nutrient management (NM), but both options reduce sediment loads being lost from agricultural acres by 45 percent, relative to baseline conditions.

Estimations of the annual reductions in total nitrogen loads being lost from agricultural acres in WLEB under alternative scenarios are shown in Figure 4(B). Inclusion of nutrient management with structural erosion control practices had a much greater impact on total nitrogen loss reduction than did structural practice adoption alone. The simulated benefits of enhanced nutrient management practices on annual nitrate reductions are dramatic, likely due to the fact that erosion control practices only address overland flow losses and nitrate may be lost to subsurface flow pathways (Figure 4D). The application of structural practices to control erosion (ECC, ECA, and FT) achieves only 56, 51, and 59 percent of the potential annual total nitrogen load reduction benefits achieved in scenarios that apply structural practices and nutrient management practices (NMC, NMA, and NM) on Critical Needs, All Needed, or all cropland acres, respectively (Figure 4B). However, ECC, ECA, and FT provide only 15, 5, and 4 percent of the nitrate load reduction benefits achieved in the NMC, NMA, and NM scenarios, respectively (Figure 4D). Treating all acres with erosion control practices (FT; \$128.3 million, annually) provides over three times the annual benefits in total nitrogen load reduction, but provides only 19 percent of the nitrate loss benefits relative to gains made with adoption of erosion control and nutrient management on Critical Needs acres only (NMC; \$8.4 million, annually). Thus, nutrient management is clearly necessary if nitrate load reductions are the

principal conservation goal. Investment decisions should carefully consider the resource concern of interest, as different management strategies are required to meet various conservation goals.

Estimations of the annual reductions in total phosphorus loads being lost from agricultural acres under alternative scenarios are shown in Figure 4(C). Phosphorus, unlike nitrogen, is commonly bound to sediment. Therefore, erosion control practices tend to have more impact on total phosphorus losses than on total nitrogen losses. However, application of nutrient management in conjunction with structural erosion control practices has a much greater impact on total phosphorus loss reduction than does structural practice adoption alone. This effect is markedly more dramatic when impacts on annual dissolved phosphorus load reductions are considered, likely due to the fact that erosion control practices only address overland flow losses, whereas dissolved phosphorus may also be lost to subsurface flow pathways (Figure 4E).

The application of structural practices to control erosion (ECC, ECA, and FT) achieves 93, 83, and 84 percent of the annual total phosphorus load reduction benefits achieved in scenarios that apply structural practices in conjunction with nutrient management practices (NMC, NMA, and NM), respectively (Figure 4C). However, ECC, ECA, and FT provide only 62, 23, and 22 percent of the potential dissolved phosphorus load reduction benefits that could be achieved with inclusion of nutrient management alongside erosion control (in NMC, NMA, and NM, respectively (Figure 4E)). This suggests that on Critical Needs acres a significant amount of the dissolved phosphorus being lost could be conserved through structural practice adoption, whereas on the moderate and low needs acres nutrient management practices are imperative in order to achieve comparable reductions in dissolved phosphorus losses. Model simulations suggest that treatment of all acres with structural erosion control practices (FT; \$128.3 million, annually) provides over 4.8 times the annual benefits in total phosphorus load reduction, but

provides only 1.3 times the dissolved phosphorus loss benefits relative to gains that could be made with adoption of structural erosion control and nutrient management practices on Critical Needs acres only (NMC; \$8.4 million, annually). As with nitrogen management decisions, phosphorus management strategies must take into account the importance of treating for total phosphorus reduction or targeting dissolved phosphorus reduction. The conservation goal informs the appropriate conservation strategy.

Beyond the general comparisons for each output variable, cross comparisons between total nitrogen and total phosphorus are also interesting. WLEB and most of the MWR are heavily tiled, so that most excess precipitation is transported through tile drainage systems. Rerouting water through tiles also reroutes nutrients to subsurface pathways. Tile drains are not only loss pathways for nitrate nitrogen and dissolved phosphorus, but have also been implicated as a loss pathway for sediment and sediment-associated phosphorus losses (Gaynord and Findlay, 1995; Molder et al., 2015). These results suggest that careful and comprehensive conservation planning that provides nutrient loss reductions in all loss pathways is needed to provide desired nitrogen and phosphorus loss reductions in this unique area (Figures 4B and 4C).

3.3. Model Responses Enclosed in Stream Order Levels

To examine additional information that can be obtained from the NHDPlus resolution SWAT model, model responses for spring and summer seasons by different stream order were examined (Figures 5 and 6). Larger loads are predicted during spring as compared to summer, which is consistent with regional weather. Greater flows are expected in WLEB during spring, when snow melt occurs. Snow melt and spring rains may mobilize nutrients applied during spring planting or in the previous fall.

Modeling at this scale exposed the fact that rivers with lower stream orders have a higher degree of uncertainty. There are multiple reasons for this uncertainty. One reason is that calibration at this scale is difficult due to lack of calibration data; there is uncertainty associated with spatial interpretations when data are lacking. Low stream order streams and rivers are generally smaller in size, so may be less resilient to disturbance, such as flooding events and or anthropogenic disturbances. Therefore, simulated nutrient and sediment quantities in low order streams may vary substantially from reality, which estimates in higher order streams should be better calibrated and have less uncertainty.

Because of lower order stream vulnerabilities to perturbation, conservation practices that benefit lower order streams are likely to be beneficial to stream health in the region. These lower order streams feed into higher order streams, which would then also benefit from the conservation practices that reduced loads to the lower order streams. The biological indicators used in this work suggest that increased focus should be placed on conservation practices that target loss reduction in the spring season instead of the entire year (Keitzer et al., 2016).

3.4. Potential Reductions in Degraded Biological Conditions

Anthropogenic disturbance, including nutrient enrichment is a serious threat to stream biodiversity (Richter et al., 1997; Dudgeon et al., 2006; Vörösmarty et al., 2010). Nutrient enrichment can cause excessive algal growth, alter algal community structure and food quality, and decrease dissolved oxygen levels, all of which can degrade stream biodiversity (Miltner and Rankin 1998, Dodds et al., 2006, Wang et al., 2007, Evans-White et al., 2009, Miltner 2010, Taylor et al., 2014). Understanding the effectiveness of conservation actions on mitigating nutrient enrichment of streams and rivers at the watershed-scale is therefore a key component of effective stream biodiversity conservation.

Averaged across all nutrient criteria, we estimate that 95% of WLEB streams have some level of nutrient enrichment that could degrade habitat and decrease biodiversity (about 19,256 km of streams). This includes nearly 73% of the streams in the watershed estimated to have annual nutrient levels above TP criteria, 93% above TN criteria, and nearly 68% of stream segments have both TP and TN concentrations above the nutrient criteria used for this study. These estimates suggest that while both are widespread in WLEB streams, streams are more likely to have excess nitrogen concentrations than excess phosphorus concentrations. However, high levels of both nutrients co-occur across a substantial portion of the watershed. Nutrient management strategies likely need to address both nitrogen and phosphorus losses through all potential loss pathways to be effective at providing benefits to stream biodiversity and health.

Our results suggest that the conservation practices simulated here are more effective at reducing stream degradation as the result of excess phosphorus than nitrogen (Fig. 7). We found that across all nutrient criteria and conservation scenarios, the percent of streams with improved water quality associated with scenario adoption ranged from < 1% to 34% based on TP criteria and from < 1% to 12% for TN criteria. Regardless of whether one or both nutrients are selected as the dominant conservation concern in the region, achieving large reductions will likely require treating a substantial portion of the watershed. For instance, treating only Critical Needs acres with erosion control and nutrient management provides at most a 3% reduction in the percent of streams impacted by phosphorus; treating all agricultural acres in the region with structural controls and nutrient management can provide up to a 34% reduction in streams impacted by phosphorus. Similar results from a variety of watersheds suggest that conservation practices must be widely implemented to achieve the large reductions in nutrient and sediment losses from

agricultural lands needed to improve stream biodiversity throughout the watershed (Einheuser et al., 2012, Bosch et al., 2013).

Our results also suggest that there are differences in potential return on investment of depending on the conservation concern being addressed (Table 5, Fig. 7). Erosion control practices alone represent a more cost effective strategy for reducing degradation by TP, while including nutrient management proves to be a more cost effective means for reducing degradation by TN (Table 5, Fig. 7). Because both nitrogen and phosphorus can negatively affect stream biodiversity (Miltner and Rankin 1998; Wang et al., 2007; Evans-White et al., 2009), it is likely necessary that both must be managed appropriately if stream biodiversity is to be restored across the watershed. This effort will require development of comprehensive conservation plans and investments in both structural erosion control and nutrient management.

This work only considered the benefits of conservation investments on in-field and edge-of-field practices as they relate to nutrient losses. However, conservation practices can also reduce sediment loading to streams, reduce the frequency of extreme flows (e.g., stream drying), and improve in-stream habitat, all of which should benefit stream biodiversity (Wang et al., 2002, Wang et al., 2006). We may therefore have underestimated the potential benefits of in-field and edge-of-field conservation practices, particularly if multiple stressors interact to affect stream biodiversity, which appears to be common (Townsend et al., 2008, Matthaei et al., 2010, Wagenhoff et al., 2011, Wagenhoff et al., 2012). In-stream practices were not explored here and may be valuable complements to on-farm conservation practices.

While nutrient enrichment is a pervasive threat to stream biodiversity (Richter et al., 1997; Dudgeon et al., 2006; Vörösmarty et al., 2010; US EPA 2015; US EPA 2016), it should be noted that it is not the only factor contributing to habitat degradation in the region. Among other

stressors (e.g., invasive species), changes to in-stream habitat quality (e.g., siltation), flow regimes, and riparian vegetation are also prevalent concerns in the WLEB (Trautman 1939, Trautman and Gartman 1974, Karr et al., 1985). Conservation practices simulated here are limited to in-field and edge-of-field practices and their impacts on overland and subsurface water flows and associated nutrients. Other stream restoration techniques, including in-stream practices, dam removal, road improvements, riparian restoration, improved floodplains connectivity, etc., were not considered here and may be essential to successful holistic agroecosystem planning and stream habitat improvement at the watershed scale. Results presented here should be considered with the caveat that the stressors and practices explored are limited in scope; these scenarios may under or overestimate the potential benefits of conservation practices in conjunction with other management across the landscape.

4. Conclusion

In this study, SWAT was applied at the NHDPlus resolution to explore the model's capacity to estimate potential environmental impacts of alternative conservation practices in terms of sediment and nutrient losses reduction with corresponding projected cost. An important finding is that SWAT is able to parse the varied impacts of conservation practices on total nitrogen, total phosphorus, nitrate, and dissolved phosphorus in response to the same conservation practice. However, without adequate monitored data, the fine-scale results cannot be appropriately validated. This work improves SWAT's capacity to serve as a decision support tool to determine conservation strategies associated with various conservation concerns and nutrient loss pathways. Structural erosion practices alone provide a sufficient remedy to sediment loss and could reduce current sediment loads lost from agricultural acres by up to 45 percent. The model

also demonstrated that nutrient loss reduction strategies, especially for nitrate and dissolved phosphorus, benefit from the inclusion of nutrient management plans. In the WLEB, dissolved phosphorus is a critical concern (Daloglu et al., 2012; Baker et al., 2014). Therefore, realistically predicting dynamic responses to management can be valuable to researchers from various disciplines, as well as land use planners.

The work presented here is not just another piece of a calibration/validation project in watershed modeling. Instead, it is the first time in history that the most advanced technology of delineating stream networks at the fine resolution (NHDPlus) was implemented on a large-scale watershed like the WLEB. The goal of this study was to determine if the SWAT model was capable of identifying environmental impacts caused by agricultural activities, including various conservation strategies, at a refined spatial scale by applying the state-of-the-art techniques, including soft-data calibration. The model proved capable to simulating seemingly reasonable statistical results. While this manuscript and the model developed to support it provide scientists, engineers, and stakeholders with increased science-based information to augment current decision support tools for determining appropriate land management and policy development, it should be noted that there is insufficient spatially explicit monitoring data in the region to support appropriate validation of the model's spatially explicit results. It is true that the proposed finest resolution model may produce uncertainty and artificial errors because of the limitation. However, potential errors could cause more negative impacts on model predictions if the associated biological analysis was conducted in larger scale such as HUC-12 (average 72 km² in unit size which is more than 27 times larger than average unit area of NHDPlus). Implementations of finer resolution data are not only to scale down the individual unit of

simulation, but it also can provide more information as indications that particular issues needed to improve in the future to enhance the accuracy of the modeling work.

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Table 1: General information of calibration/validation data sources

Station ID	HUC-8 Number	Drainage Area (km2)	Location	Data Source (Streamflow)	Data Source (Nutrients)
4176500	04100002 (Raisin)	2680 (11.3%)	River Raisin near Monroe, MI	Heidelberg College River Studies	Heidelberg University & USGS
4178000	04100003 (St. Joseph)	1541 (6.5%)	St. Joseph River near Newville, IN	United States Geologic Survey	USGS
LES040-0007	04100004 (St. Marys)	1311 (5.5%)	St. Marys River	Indiana Department of Environmental Management	Indiana Department of Environmental Management
4193500	04100009 (Maumee)	16270 (68.3%)	Maumee River at Waterville, OH	Heidelberg College River Studies	Heidelberg University, USGS & OH Environmental Protection Agency
4198000	04100011 (Sandusky)	2015 (8.5%)	Sandusky River near Fremont, OH	Heidelberg College River Studies	Heidelberg University, USGS & OH Environmental Protection Agency

The selected representative gauge stations are used in calibration/validation processes where details of the selection procedure can be found in Daggupati et al. (2015).

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Table 2: General performance ratings (Moriassi et al., 2007)

Categories of General Performance Rating	NSE	PBIAS(%)	
		Streamflow	Sediment & Nutrients
Very Good	$0.75 < \text{NSE} \leq 1.00$	$\text{PBIAS} < \pm 10$	$\text{PBIAS} < \pm 25$
Good	$0.65 < \text{NSE} \leq 0.75$	$\pm 10 \leq \text{PBIAS} < \pm 15$	$\pm 25 \leq \text{PBIAS} < \pm 40$
Satisfactory	$0.50 < \text{NSE} \leq 0.65$	$\pm 15 \leq \text{PBIAS} < \pm 25$	$\pm 40 \leq \text{PBIAS} < \pm 70$
Unsatisfactory	$\text{NSE} \leq 0.50$	$\text{PBIAS} \geq \pm 25$	$\text{PBIAS} \geq \pm 70$

NSE: Nash-Sutcliffe efficiency coefficient
PBIAS: Percent bias

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Table 3: Definitions of CEAP scenarios and the corresponding cost (USDA-NRCS, 2015 and 2016)

Conservation Practices	Content	Projected Additional Cost \$ (USD)	Treatment Needs Level		
			High	Moderate	Low
Baseline	BL	162,500,000 [†]	-	-	-
Erosion Control - Critical	ECC	4,504,349	EC	-	-
Erosion Control - All Needed	ECA	55,782,326	EC	EC	-
Full Treatment Erosion Control	FT	128,262,006	EC	EC	EC
Nutrient Management - Critical	NMC	8,394,761	EC + NM	-	-
Nutrient Management - All Needed	NMA	149,253,423	EC + NM	EC + NM	-
Full Nutrient Management	NM	263,393,414	EC + NM	EC + NM	EC + NM
Grass Background	GBG	-	-	-	-

- BL : The default scenario that incorporates current watershed information and the associated management practices in the field. The baseline scenario will be calibrated before conducting the following assessment in varying scenarios.
- ECC : The ECC scenario includes region in the watershed with high treatment need for the target of erosion control.
- ECA : The ECA scenario includes region in the watershed with high and moderate treatment need for the target of erosion control.
- FT : The FT scenario includes region in the watershed with high, moderate, and low treatment need for the target of erosion control.
- NMC : The NMC scenario includes region in the watershed with high treatment need for the target of erosion control and nutrient management.
- NMA : The NMA scenario includes region in the watershed with high and moderate treatment need for the target of erosion control and nutrient management.
- NM : The NM scenario includes region in the watershed with high, moderate, and low treatment need for the target of erosion control and nutrient management.
- GBG : The GBG scenario represents the assumption that all available croplands are forced to apply grassland practice with no exception. In this case, the expected cost could be the highest of all scenarios.
- [†] : Per-acre cost data for individual conservation practices were available for the 2010 crop year. The following four sources were used to compile baseline conservation cost estimates: 1. The NRCS payment schedule, 2. The NRCS conservation plans database, 3. The Technical Service Provider rate database, and 4. Average cropland rental rates.

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Table 4: Statistics of the calibration (1990-1999) and validation (2000-2006) periods

Station	Streamflow		Sediment	TP	TN
	NSE	PBIAS (%)	PBIAS (%)	PBIAS (%)	PBIAS (%)
Raisin	0.70 [†] /0.43 [‡]	-11.76/-26.07	16.71/35.46	-3.55/-22.74	14.66/3.59
St. Joseph	0.73/0.74	22.70/18.66	-10.43/-20.3	5.33/4.95	-25.39/-68.71
St. Marys	0.54/0.43	17.94/25.25	17.99/19.57	6.52/9.42	8.23/21.00
Maumee	0.87/0.88	18.03/13.56	10.07/-10.59	8.42/3.42	16.87/13.27
Sandusky	0.82/0.75	18.67/7.00	18.63/-35.01	-12.78/0.61	-15.26/-12.37

[†]: Quantitative measures on the left-hand side represent the statistics for the calibration period.

[‡]: Quantitative measures on the right-hand side represent the statistics for the validation period.

Table 5. Coefficient estimates (95% confidence interval) from multilevel models describing the effects of additional investment (Cost), including nutrient management (Nutrient management), and their interaction (Cost x nutrient management) on the amount of the watershed that is improved according to nutrient criteria for total phosphorus (TP) and total nitrogen (TN). Estimates were considered significant if bootstrapped (n = 10,000) confidence intervals did not include zero.

Nutrient	Intercept	Cost	Nutrient Management	Cost x Nutrient Management
TP	0.91 (0.27,1.57)	0.072 (0.063,0.079)	-0.15 (-1.07,0.78)	-0.02 (-0.029,-0.011)
TN	0.029 (-0.27,0.31)	0.008 (0.005,0.01)	0.40 (0.03,0.80)	0.004 (0.0005,0.008)
Both TN & NP	1.13 (0.56,1.68)	0.08 (0.075,0.089)	-0.17 (-0.9376,0.6494)	-0.02 (-0.03,-0.015)

Figure 1. Location of the Western Lake Erie Basin and the representative gauging stations.

Figure 2. The HUC-12 and the NHDPlus models in WLEB (Daggupati et al., 2015).

Figure 3. Calibration and validation results of temporal processes of Maumee Station: (A) streamflow; (B) sediment; (C) total phosphorus; and (D) total nitrogen. (OBS: observation data; SIM: simulation output; -C: calibration period; -V: validation period)

Figure 3. (continued)

Figure 4. Demonstration of percentage reduction of targeted output responses with alternative conservation scenarios in terms of projected additional cost (\$ million, USD): (A) Sediment; (B) Total nitrogen; (C) Total phosphorus; (D) NO_3 ; and (E) Dissolved phosphorus. (ECC: Erosion Control – Critical; ECA: Erosion Control – All Needed; FT: Full Treatment Erosion; NMC: Nutrient Management – Critical; NMA: Nutrient Management – All Needed; NM: Full Nutrient Management; GBG: Grass Background)

Figure 5. Model responses for spring and summer seasons by different stream order (Baseline scenario): (A) Sediment; (B) Total nitrogen; (C) Total phosphorus; (D) NO₃; and (E) Dissolved phosphorus.

Figure 6. Model responses for spring and summer seasons by different stream order (NM scenario): (A) Sediment; (B) Total nitrogen; (C) Total phosphorus; (D) NO₃; and (E) Dissolved phosphorus.

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Figure 7. Amount of the watershed in which stream biological conditions improved as a result of additional investment in conservation practices reducing nutrient enrichment: (A) improved percentage for streams that were above TP criteria; (B) improved percentage for streams that were above TN criteria; and (C) improved percentage for streams that were above both TP and TN. Boxplots represent the interquartile range of improvements across nutrient criteria with whiskers extending to the 10th and 90th percentiles. Lines show the predicted relationship between additional investment for conservation scenarios that included only erosion control practices (solid line) and scenarios that included erosion control and nutrient management practices (dashed line).

Appendix A. SWAT-SAS

In general SWAT application, SPCON is one of the important parameters that associated with the sediment transportation process. By default, users can only alter SPCON value for the entire watershed. However, a single value of SPCON may not be sufficiently representative if the targeted watershed is composed of wide variety of geographical characters (e.g., rapidly changing slope among subbasins). In this study, SWAT-SAS (Subregional Adjustment of Sediment) is developed to solve the issue so that not only the sediment loss can be simulated more accurately, nutrient load such as total phosphorus prediction can also be more representative. By adding an additional line at the end of each *.rte file, users can specify customized SPCON values at subbasin level.

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.rte file subbasin: 1 6/10/2014 12:00:00 AM ArcSWAT 2012.10_0.8
2.500 CHW2 : Main channel width [m]
0.200 CHD : Main channel depth [m]
0.00500 CH_S2 : Main channel slope [m/m]
1.700 CH_L2 : Main channel length [km]
0.020 CH_N2 : Manning's nvalue for main channel
0.500 CH_K2 : Effective hydraulic conductivity [mm/hr]
2.000 CH_COV1 : Channel erodibility factor
2.000 CH_COV2 : Channel cover factor
10.000 CH_WDR : Channel width:depth ratio [m/m]
0.000 ALPHA_BNK : Baseflow alpha factor for bank storage [days]
0.00 ICANAL : Code for irrigation canal
0.00 CH_ONCO : Organic nitrogen concentration in the channel [ppm]
0.00 CH_OPCO : Organic phosphorus concentration in the channel [ppm]
0.00 CH_SIDE : Change in horizontal distance per unit vertical distance
0.00 CH_BNK_BD : Bulk density of channel bank sediment (g/cc)
0.00 CH_BED_BD : Bulk density of channel bed sediment (g/cc)
0.00 CH_BNK_KD : Erodibility of channel bank sediment by jet test (cm3/N-s)
0.00 CH_BED_KD : Erodibility of channel bed sediment by jet test (cm3/N-s)
0.00 CH_BNK_D50 : D50 Median particle size diameter of channel bank sediment (μm)
0.00 CH_BED_D50 : D50 Median particle size diameter of channel bed sediment (μm)
0.00 CH_BNK_TC : Critical shear stress of channel bank (N/m2)
0.00 CH_BED_TC : Critical shear stress of channel bed (N/m2)
0.00 0.00 0.00 0.00 0.00 0.00 0.00 0.00 0.00 0.00
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