Chapter 5

Watershed Modeling: National Institute of Food and Agriculture–Conservation Effects Assessment Project

M. Arabi, D.W. Meals, and D.LK. Hoag

Simulation modeling is arguably one of the most powerful scientific tools available to address questions, assess alternatives, and support decision making for environmental management. Watershed models are used to describe and understand the behavior of land and water systems under prevailing and projected conditions. They are increasingly used to evaluate potential water quality impacts of pollution control strategies, including agricultural practices. Modeling efforts constituted a major component of watershed-scale National Institute of Food and Agriculture–Conservation Effects Assessment Project (NIFA–CEAP). Chapter 5 describes the usefulness and challenges of watershed modeling and then summarizes activities, overall findings, and lessons learned from the modeling components of these studies. It also presents a set of recommendations for further enhancement and proper application of watershed modeling technologies for future conservation effects assessment at the watershed scale.

Appraisal of individual and cumulative effects of conservation practices on water quality using monitoring data is ideal but can be costly and, in some cases, impractical (especially for retrospective studies). Controlled watershed experiments, particularly paired-watershed analysis, have been used to characterize how long-term changes in land use or implementation of pollution control actions alter hydrologic and water quality fluxes (Bishop et al. 2005). The wider use of these approaches, however, faces challenging issues, including (1) identifying control landscapes; (2) characterizing the role of type, location, and timing of individual practices when multiple practices are implemented over time; (3) overcoming lag time or delay in water quality response to changes in management due to large pools of legacy sediments and nutrients in soils and streams; and (4) addressing the uncertainty of methods for extension of results to other spatial and temporal scales. Watershed modeling in combination with strategic monitoring can address these issues (Easton et al. 2008b; Tomer and Locke 2011).

Process-based distributed models facilitate dynamic hydrologic and water quality assessments under scenarios that cannot be investigated practically using actual watershed experiments. They can simulate what-if scenarios at varying spatial scales from fields to watershed outlets and at varying temporal scales from daily to decadal time steps. Watershed models are embodiments of scientific hypotheses. As such, they can be used within formal statistical hypothesis testing frameworks to answer questions relevant to watershed assessment and planning. For example, modeling scenarios can provide necessary information to examine the hypothesis that a targeted approach to implementation of conservation practices is significantly more effective

(c) SWCS. For Individual Use Only

in reducing pollutants than a random approach. Similarly, the likelihood that the assimilative capacity of a given water body will significantly decrease under projected climate change scenarios can be tested with models.

Over the past three decades, a number of models with varying levels of complexity have been developed to address a wide range of environmental issues. Substantial advances in modeling technologies have been made to study generation, fate, and transport of pollutants in various environmental compartments. Process-based models, such as the Soil and Water Assessment Tool (SWAT) (Arnold et al. 1998), Annualized AGricultural Non-Point Source (AnnAGNPS) (Bingner and Theurer 2003), Hydrological Simulation Program (Fortran) (Bicknell et al. 1996), Generalized Watershed Loading Function (GWLF) (Haith et al. 1992), Agricultural Policy/Environmental eXtender (APEX) (Williams and Izaurralde 2006), and the Water Erosion Prediction Project (WEPP) model (Flanagan et al. 2007), are commonly used for total maximum daily load analysis and for watershed-scale assessment of conservation practice effects. Selection, development, and application of watershed models must be conducted according to the purposes they are supposed to serve. Modeling can serve several distinct purposes, including prediction, exploratory analysis, knowledge building, and collaborative learning (Brugnach and Pahl-Wostl 2007). Whatever the purpose, the key issue is establishing reasonable practical credibility in the model-based analysis.

Watershed Models for Prediction

The purpose of predictive watershed modeling is to closely mimic physical, chemical, and biological responses of a system under current or future climate and land-use conditions. Predictive tools are also used to assess outcomes of certain actions, including the option to take no action. Current trends in development and application of predictive models for watershed management include the following:

- Predicting fluxes of water and contaminants in surface water and groundwater
- Identifying critical pollutant source areas within watersheds
- Quantifying water quality effects of management alternatives, e.g., agricultural practices
- Allocating pollutant loads among point and nonpoint sources for total maximum daily load development
- Identifying implications of long-term changes in land use on water quantity and quality
- Predicting consequences of future climate change on water quantity and quality

Numerous studies have applied models to determine water quality impacts of agricultural conservation practices. Perhaps the most visible of these studies are the national CEAP cropland studies in the Upper Mississippi River Basin (USDA NRCS 2010), the Chesapeake Bay region (USDA NRCS 2011a), and the Great Lakes system (USDA NRCS 2011b). These studies used the SWAT, APEX, and/or Environment Policy Integrated Climate models for two main reasons. First, the models were applied to quantify the effects of conservation practices currently present on the landscape in the regions. Second, potential benefits that could be gained by implementation of additional conservation treatment in under-treated agricultural acres were projected. The outputs of these studies were expressed deterministically in terms of numeric reduction estimates. For example, the Upper Mississippi River Basin study for a stream location at Grafton, Illinois, projected that implemented conservation practices to date have resulted in a 37% reduction of sediment loads, a 21% reduction in total nitrogen (N) loads, a 40% reduction in total phosphorus (P) loads, and a 51% reduction in atrazine loads (USDA NRCS 2010).

(c) SWCS. For Individual Use Only

Despite best efforts to calibrate the modeling component of the study, the projected reductions of pollutants were typically within the range of modeling errors. Therefore, the credibility of these predictions remains to be investigated. Future studies could establish confidence in modeling outcomes by reducing uncertainties to ensure that projected reduction of pollutants can be unambiguously attributed to implementation of practices and that they are not the artifact of modeling uncertainties.

A complementary and arguably more realistic approach to the assessment of conservation practices at the watershed-scale has been ongoing for a considerable period of time in 14 USDA Agricultural Research Service benchmark watersheds. The primary goals of these studies are to advance the science of process-based models used for simulating agricultural pollutants, conduct strategic monitoring to collect and analyze data for conservation effect assessment, and assist with validation of models used for the national assessment component of CEAP. The selected models for these studies are SWAT and AnnAGNPS (Harmel et al. 2008; Heathman et al. 2008; Richardson et al. 2008; Yuan et al. 2008; Cho et al. 2010). Simultaneous application of the two models in the benchmark watersheds presents the opportunity to evaluate the role of model structure uncertainty on the simulated effects of practices. When available, data collected from USDA Agricultural Research Service benchmark watersheds could be used to develop methods for enhancing the degree of confidence in modeling environmental benefits of conservation efforts.

Other studies have predicted water quality impacts of agricultural practices in different regions of the world using SWAT (Behera and Panda 2006; Bracmort et al. 2006; Gassman et al. 2007; Lam et al. 2011), AnnAGNPS (Mostaghimi et al. 1997; Yuan et al. 2002; Licciardello et al. 2007; Polyakov et al. 2007; Kuhnle et al. 2008; Zhou et al. 2009), WEPP (Renschler and Lee 2005; Flanagan et al. 2007; Pandey et al. 2009), GWLF (Lee et al. 2000; Schneiderman et al. 2007; Easton et al. 2008b), and APEX (Tuppad et al. 2010). One important and often neglected aspect of assessing conservation benefits arises from modeling uncertainties due to the stochastic nature of watershed processes and our incomplete knowledge of the system. When models are used in the predictive mode, errors must be minimized, uncertainties ought to be reduced, and confidence in model-based analysis should be explicitly documented. Several studies have demonstrated that the estimated benefits of pollution control strategies could markedly differ as a function of uncertainty in model parameters, incomplete representation of critical processes within the model structure, and more importantly, the numerical procedure for representation of management actions (Harmel et al. 2006; Harmel and Smith 2007; Arabi et al. 2007b, 2008).

Watershed Models for Exploratory Analysis

Exploratory models are used to provide insights about potential outcomes, opportunities, and risks associated with alternative management strategies. They are useful for designing effective policy instruments and exploring the attainability of various policy objectives. Examples of the use of watershed models for exploratory purposes include modeling the following:

- Critical natural processes and key management actions that influence fate and transport of pollutants at the watershed scale
- Geospatial and temporal factors that control effectiveness of conservation practices
- The likelihood that desired water quality targets can be attained
- Tradeoffs and targets associated with alternative pollution control strategies
- Possible outcomes of new regulations on land use and cropping patterns within a watershed system

(c) SWCS. For Individual Use Only

- Whether adoption of conservation practices will increase in response to regulatory policy instruments
- Possible outcomes of large-scale policy decisions, e.g., biofuel energy production, cap and trade, and on land-use patterns and cropping systems

In the context of watershed management and conservation planning, it is often desired to expose tradeoffs and targets associated with various management options. Integrated modeling-optimization systems can facilitate incorporation of environmental, socioeconomic, and institutional factors in the planning process. Over the past decade, spatially explicit multiobjective optimization methods have been developed and demonstrated to explore optimal cropping systems or cost-effective agricultural practices within watersheds to meet water quality targets (Srivastava et al. 2002; Veith et al. 2003; Whittaker et al. 2003; Bekele and Nicklow 2005; Rabotyagov et al. 2010b; Rodriguez et al. 2011). For example, Arabi et al. (2006a) applied SWAT within a genetic-algorithm based optimization framework in two small (<10 km^2 [2.92 mi^2]) agricultural watersheds in Indiana and showed that optimal allocation of practices would yield three-fold higher environmental benefits in terms of sediment and nutrient reduction at the same cost compared to an existing suite of practices.

Other important factors can be also included in the analysis. While previous optimization studies have considered pollutant loads and costs in the optimal design of practices, the next logical step is to include social acceptability expressed in terms of probability of adoption. Figure 5.1 depicts a hypothetical Pareto optimal surface for allocation of conservation practices. Each point on the surface reflects a spatially explicit suite of conservation options for the watershed. Because pollutant loads, costs, and probability of adoption are conflicting objectives, improving one objective will result in degradation of other objectives. Minimizing pollutant loads requires higher conservation expenditure. At the same time, some practices are socially less acceptable and have lower chance of adoption. The analyst can choose a solution most consistent with priorities of stakeholders in the region from the Pareto optimal surface using a multicriteria decision analysis approach. Points that are not on the Pareto optimal surface (depicted with a gray cross symbol) represent suboptimal solutions.

The use of models for exploring optimal conservation options is an example of model application for exploring the possibility of certain outcomes without assigning a given probability to them. In this context, modeling uncertainties could serve as a source of innovation and could unveil possibilities that are consistent with internal system behavior. Therefore, modeling uncertainties for exploratory purposes does not necessarily need to be eliminated, but uncertainty must be elucidated and clearly communicated.

Watershed Models for Building Knowledge

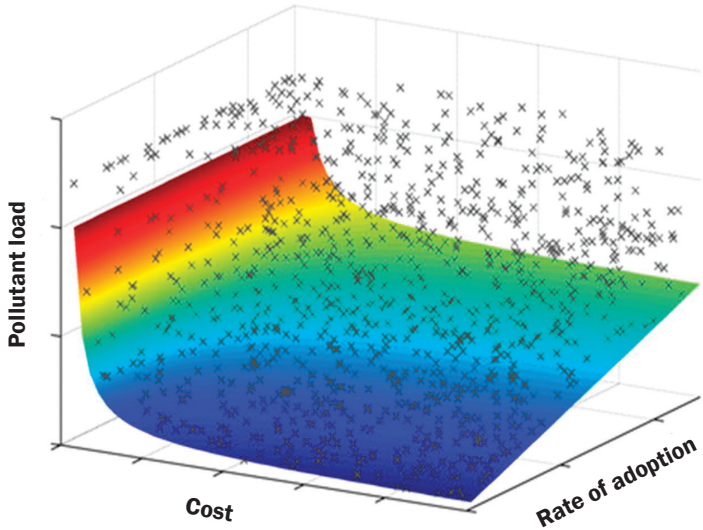
Watershed models may be applied to build knowledge of critical natural processes and key management actions that control hydrologic and water quality responses of a system. Examples of model application for knowledge-building purposes include the following:

- Assessing impacts of spatial and temporal scales on modeling watershed processes
- Assessing appropriate spatial and temporal scales for assessment of conservation practices
- Assessing appropriateness of different algorithms for representation of conservation practices
- Assessing suitability of watershed models for representation of agricultural practices

Models have been used extensively to develop knowledge about important issues that affect assessment, planning, and cost-effective implementation of conservation practices. For exam-

(c) SWCS. For Individual Use Only

Figure 5.1
Schematic of a hypothetical Pareto optimal surface for exploring tradeoff and targets associated with conservation options at the watershed scale. The surface can be calculated using an integrated watershed modeling-optimization system. Each point on the Pareto optimal surface represents an optimal suite of conservation practices. Points that are not on the surface reflect suboptimal solutions.



ple, models have been used to determine critical areas within a watershed that are major sources of nonpoint source pollution.

Targeting critical areas for implementation of conservation practices is considered an efficient approach to nonpoint source pollution control at the watershed scale. Determining landscape position of these areas for a pollutant of concern depends on the magnitude of sources and transport pathways from the source to a stream location or a water body of concern. Alexander et al. (2000) used the SPARROW (SPAtially Referenced Regressions on Watershed Attributes) model to unveil the importance of in-stream transport processes on removal of contaminants within the Mississippi River Basin. They concluded that major sources of N that drain into large tributaries (i.e., higher order streams) contribute at significantly higher magnitudes to total N delivered to the Gulf of Mexico than other major sources on small tributaries. The roles of other transport pathways, such as subsurface drainage systems, wetlands, and potholes, on fate and transport of nutrients have been investigated using process-based models.

Other studies used models to identify and communicate the correlation between critical areas and watershed attributes obtained from readily available data. Ghidey et al. (2010) showed that indices comprising topographic, hydrologic, and agronomic factors reveal critical areas for runoff and atrazine losses in accord with their findings from applications of the SWAT and APEX models. Schneiderman et al. (2007) and Easton et al. (2008a) used different versions of the GWLF and SWAT models, respectively, to demonstrate the temporal variability of source areas with the extent of soil saturation, particularly for pollutants that are carried with runoff in dissolved form (e.g., dissolved P).

(c) SWCS. For Individual Use Only

Watershed Models for Collaborative Learning

Application of models for collaborative learning aims to change attitudes and beliefs toward environmental management and to engage stakeholders in the decision-making process. The model-building process and model outcome evaluation may facilitate collaborative learning about a system. Watershed management involves stakeholders with multiple, and often, conflicting objectives. Therefore, participatory approaches that integrate stakeholder values and preferences in the planning process are critical for successful implementation of cost-effective solutions. In this context, the modeling process can be used to help mangers, decision makers, and stakeholders exchange ideas, set goals and objectives, learn about their systems of interest, evaluate management scenarios, and identify impacts of their decisions on the environment.

Stakeholder participation in watershed management is necessary for several reasons (Korfmacher 2001). First, it is the civic duty of involved citizens to participate in matters that affect them. Second, the information and knowledge imparted by stakeholders can substantively improve the decision-making process. This particularly holds true for conservation planning. Land owners and farmers gather invaluable knowledge of the system through direct observation of impacts of different agricultural practices on crop productivity, topsoil erosion, soil moisture, flooding, and other issues. Incorporation of this knowledge can establish practical credibility in model-based analysis. Third, the rate of adoption and social acceptability of conservation practices are likely to increase when stakeholders are involved in the process. In particular, social barriers to targeted implementation of conservation practices could be reduced by securing public support through stakeholder education and engagement in the decision-making process.

Scientific knowledge, including model outcomes, must be "socially robust" and should be seen by stakeholders as "transparent and participative" (Gibbons 1999). Current process-based modeling tools for watershed management and conservation planning require significant expertise and resources. Outputs created by these models are often overwhelming and difficult to process. Watershed models and their outputs are not generally accessible to broader stakeholders. However, the advent of Web-based tools has provided an impetus for engaging stakeholders in the modeling process and enhancing the use of watershed models for learning purposes.

Establishing Credibility for Model Application

Processes that control the generation, fate, and transport of water and contaminants within watershed systems are complex and tend to interact in a highly nonlinear fashion. Development of watershed models require making pragmatic assumptions and simplification of the real system. Hence, a simulation model is a mere representation of reality and will always include aleatory and epistemic uncertainties. When models are used to make decisions that affect stakeholders, factors that could conspire against credible model outcomes must be considered. Establishing credibility for the development, selection, and application of models is an essential component of modeling studies.

Advances in computer technology have enhanced our capacity to measure and model watershed processes at finer spatial and temporal resolutions. However, any watershed modeling process entails uncertainties from both model abstractions and input data. To this uncertainty, we can add the uncertainty that exists with the decision-making process. Consequently, it may be tempting to postpone decisions until more data of higher quality become available or our knowledge of watershed processes is more complete. But failing to act comes at a cost to the health of

(c) SWCS. For Individual Use Only

the ecosystem and can lead to unacceptable environmental degradation. Therefore, some levels of uncertainty in our scientific knowledge, including insights wrested from model applications, must be tolerated but should be recognized. Watershed modeling is increasingly embedded in the decision-making process, but establishing confidence in a model prior to its application is vital for the acceptance of the model and its outcomes in support of environmental decisions.

Credibility of a model can be defined as "a sufficient degree of belief in the validity of the model" (Rykiel 1996). Attributes of a valid model can be described by the following from Beck et al. (1997):

- Soundness of mathematical representation of processes
- Sufficient correspondence between model outputs and past observations
- Fulfillment of the designated task

Peer review is the primary mechanism for appraisal of the soundness of model algorithms. For example, Van Liew et al. (2007) evaluated the suitability of the SWAT model for the CEAP in several watersheds across the United States. The study concluded that the model is better suited for CEAP investigations in the more humid, permanently cultivated watersheds in the eastern part of the country than the arid and semiarid regions of the west. This assessment was also confirmed by Veith et al. (2010). Other studies have compared soundness and applicability of different models for simulating hydrologic and water quality processes at the watershed scale, total maximum daily load analysis, and conservation assessment (Borah and Bera 2003; Saleh and Du 2004; Shoemaker et al. 2005; Sadeghi et al. 2007; Parajuli et al. 2009).

Model calibration, also referred to as parameter estimation, is conducted to establish the performance validity of a model. When model simulations fail to match past observations according to a set of predefined criteria, model outcomes for future conditions or management options bear little credibility. Process-based watershed models typically include hundreds of parameters that represent various hydrologic and water quality processes. Many of these parameters do not have a direct physical interpretation and cannot be directly measured in the field. Instead, they are estimated using manual or automatic calibration procedures whereby model parameters are altered until an acceptable level of agreement between model predictions and observed behavior of the system is obtained. The set of model parameters that produces the best performance is called the calibrated or optimal set and is used for future predictions.

The model calibration process can also expose deficiencies in conceptual elements of a model. Then the analyst must consider enhancing the structure of the model. Revising the model structure calls for collection of more observational data for calibration and testing of new algorithms. This cycle of data collection and model refinement is terminated at a reasonable point, where essential characteristics of the watershed system under study are captured. The term essential characteristics encompasses both critical natural processes and key anthropogenic activities and is, of course, subject to professional judgment and interpretation. For example, subsurface drainage systems substantially alter movement of water and agrichemicals in groundwater. Thus, model development and application in areas where subsurface drainage systems are prevalent (e.g., Midwestern United States) must consider their impacts.

Another example of model refinement is the role of claypan soil horizons on movement of water and pollutants. The Missouri NIFA–CEAP revealed that the presence of naturally formed claypan soil horizons could impede vertical subsurface flow and could facilitate lateral movement of soil water. Claypan soils were shown to increase runoff potential and play a role in prolonged high atrazine concentrations in surface water. Models used for prediction of

(c) SWCS. For Individual Use Only

hydrologic and water quality fluxes in the study watershed were modified to incorporate the hydrologic and water quality impacts of claypan soils.

Challenges in Model Application for Watershed-Scale Conservation Assessment

Credible application of watershed models for pollutant source identification and assessment of management options is often hampered by lack of sufficient watershed data, challenges in determination of model parameters, and unavailability of widely accepted modeling algorithms for explicit representation of conservation practices.

Data Availability

Sufficient data must be available to properly develop a model for a watershed system of interest, estimate model parameters, and corroborate its performance. Soil, land cover, elevation, and hydrography datasets have become easily accessible owing to the advancement of geographic information systems and remote sensing technology. Long-term measurements of climate, including precipitation and temperature as well as streamflow are generally available for many watersheds in the United States. However, other important data are difficult, if not impossible, to collect. These data include the following:

- Locations of existing conservation practices within the watershed and their characteristics
- Farm-level information about fertilizer and pesticide application rates, timing, and methods
- Locations and characteristics of structures, such as surface and subsurface drainage systems, reservoirs, diversions, and irrigation systems
- Long-term water quality data with a sufficient frequency and spatial coverage before and after implementation of practices
- Flow and water quality data from point sources, including wastewater treatment plants and septic systems
- Concentrations of nutrients in soils
- Information on legacy sediments and nutrients in the channel network

In particular, access to spatially explicit information on conservation practices is essential for evaluation of their impacts but is rarely available. Performance of practices could vary substantially according to their landscape positions, soil properties, and other geospatial factors. Practices must be represented in the model at the precise locations where they are implemented, otherwise modeling their impacts on hydrologic and water quality processes is fraught with uncertainty.

Load Estimation

Water quality components of watershed models are often calibrated and tested against monthly nutrient data. Water quality monitoring data represent only a snapshot of the system behavior. Sampling location, duration, and frequency, along with accuracy of measuring instrumentation could affect development, enhancement, selection, and parameterization of watershed models (Johnes 2007). In the United States, daily or more frequent discharge measurements on many rivers and streams are available from the US Geological Survey. Conversely, nutrient concentrations are often measured by local watershed groups on less frequent (e.g., weekly or monthly) time steps. A variety of load estimation techniques have been developed to estimate monthly or annual loads from intermittent, instantaneous water quality concentration samples

(c) SWCS. For Individual Use Only

(Cohn et al. 1992; Stenback et al. 2011). The Load Estimator (LOADEST) model (USGS 2004) developed by the US Geological Survey is one of the commonly used tools for estimations of constituent loads, given a time series of streamflow and constituent concentrations (Goolsby et al. 2000; Goolsby and Battaglin 2001; Hooper et al. 2001; Maret et al. 2008; Eshleman et al. 2008). LOADEST includes several statistical estimation methods for development of an appropriate regression model for load estimation.

The validity of LOADEST and other techniques depends greatly on the adequacy of the monitoring design for capturing the temporal variability and spatial heterogeneity of water quality fluxes in the system under study. Estimation of annual P and N loads have shown errors of up to 35% and 65%, respectively (Robertson and Roerish 1999; Moatar and Meybeck 2005; Ullrich and Volk 2010). When estimated monthly nutrient loads are used to calibrate a watershed model, load estimation errors contribute to the prediction uncertainty. The analyst must select a proper load estimation technique, but assessing the accuracy of these methods in the absence of long-term, site-specific water quality data is challenging.

Nonuniqueness of Calibrated Parameter Values

The agreement between model outputs and observed data is typically evaluated using some predefined criteria. Many different goodness of fit measures have been proposed to provide guidelines for proper model calibration (Gupta et al. 1998; Legates and McCabe 1999; Moriasi et al. 2007). It is widely accepted that watershed models cannot be proven or validated but only tested and invalidated (Konikow and Bredehoeft 1992). As such, it cannot be ascertained with complete certainty that optimal parameter values are unique or can be unambiguously identified. Literature is replete with studies attesting to the feasibility of obtaining several sets of model parameters that produce equally good fits between model simulations and observations (Beven and Binley 1992; Beven 1993; Spear et al. 1994). This consideration is particularly important for application of complex watershed models with a large number of parameters (Wagener et al. 2001; Tolson and Shoemaker 2007).

In the context of watershed modeling, the nonuniqueness of optimal parameter values poses a challenge. Model parameters are surrogates for the watershed processes that they represent. Therefore, different parameter sets indicate the importance of different processes, identify different critical runoff and pollutant source areas, and highlight different key pathways for movement of water, sediments, and contaminants. A subsequent question then arises: which set of parameter values should be used to identify pollutant sources and to select cost-effective control strategies? The choice of cost-effective pollution control strategies could substantially vary depending on the choice of calibrated parameter values. Therefore, identification of critical source areas, assessment of conservation practice impacts, and targeting of effective abatement strategies remains elusive.

Application of probabilistic approaches may present a practical solution for coping with the nonuniqueness issue. For example, Arabi et al. (2007b) used a computational framework to propagate parameter uncertainty forward into hydrologic and water quality outputs of SWAT. The study showed that the suggested range of most SWAT parameters, especially parameters that are used to determine the transport capacity of streams and initial concentration of nutrients in soils, required site-specific adjustment. It was evident that uncertainties associated with sediment and nutrient outputs of the model were too large, perhaps limiting its application for

(c) SWCS. For Individual Use Only

determination of design quantities. However, the estimated effectiveness of conservation practices after reduction of parameter uncertainties varied by less than 10%. This suggested that practice effectiveness could be ascertained with good confidence using models when uncertainty is reduced.

Algorithms for Numerical Representation of Conservation Practices

The numerical procedure used to represent conservation practices can influence their estimated benefits. Unfortunately, most comprehensive watershed models do not include explicit options for representation of agricultural practices, particularly structural practices. Therefore, relevant model parameters are altered to mimic the impacts of practices on hydrologic process. This approach is highly subjective and results in inconsistent evaluation of management options, even when within the same geographic region. Recent studies have focused on development of standard protocols for representation of practices with models, such as SWAT (Gassman et al. 2007; Arabi et al. 2008). However, until these methods are well established and commonly accepted, numerical representation of conservation practices will be a source of considerable uncertainty.

The Choice of a Model for Watershed-Scale Conservation Assessment and Planning

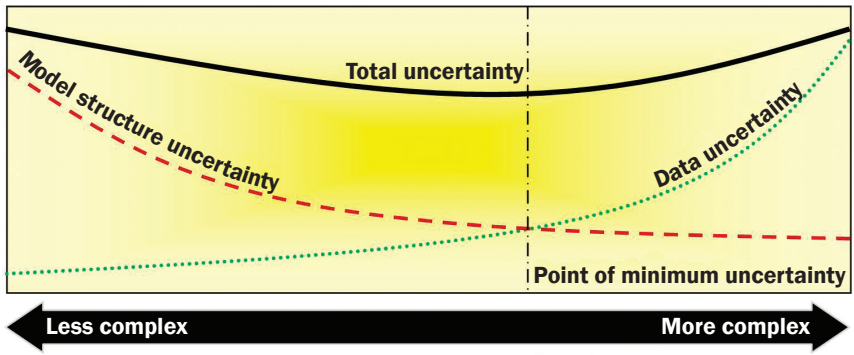
Selection of an existing watershed model of appropriate complexity or development of a new model structure should be a function of the modeling purpose, characteristics of the watershed system under study, and data availability (Wagener et al. 2001). Due to inherent trade-offs between data uncertainty and model structure uncertainty, an optimal level of complexity exists for watershed models (figure 5.2). With the temptation to incorporate more processes in watershed models comes an increase in their structural complexities. To the contrary, data availability and identifiability among other pragmatic considerations tend to favor adopting simple model structures. The analyst must select a proper level of complexity that can be supported by available data from watershed.

Model complexity is characterized by the level of details in the processes that constitute the model. Watershed processes and their nonlinear relationships are captured by state variables and parameters. Sufficient data must be available to estimate model parameters and adjust initial conditions for each state variable. Therefore, data requirements increase as the number of model parameters and state variables increase.

Models suitable for conservation assessment and planning must have certain characteristics. First, the model should be process based to be able to represent essential characteristics of the system and capture the impacts of practices on different hydrologic and water quality processes. Second, its parameters should be distributed in space in order to represent practices in a spatially explicit fashion. Third, continuous simulations are desired for assessing long-term benefits of management alternatives. Daily or subdaily time steps could facilitate simulation of the fate and transport of contaminants that move within the watershed in response to discrete events. Finally, capabilities for routing water and contaminants overland and within streams are essential when the aim is to characterize interactions among management practices.

(c) SWCS. For Individual Use Only

Figure 5.2
A simple model complexity continuum (adapted from Hanna 1988). Note that total uncertainty represents the summation of data and model structure uncertainties.



Empirical/statistical
Less data intensive
Rigid
Trend analysis
Example: SPARROW, GWLF

Physically-based/mechanistic
More data intensive
Flexible
Process specific analysis
Example: SWAT, AnnAGNPS

Role of Watershed Modeling in the National Institute of Food and Agriculture–Conservation Effects Assessment Project

In the absence of strategic monitoring experiments for directly measuring conservation effects of agricultural practices (see Chapter 4), watershed modeling was the primary approach encouraged to address the following questions required by the NIFA–CEAP program request for applications (USDA CSREES 2004, 2005, and 2006):

- Within the hydrologic and geomorphic setting of a watershed, how do the timing, location, and suite of implemented agricultural conservation practices affect surface and/or groundwater quality at the watershed scale?
- What are the relationships among conservation practices implemented in a given watershed with respect to their impact on water quality? Are the effects additive, contradictory, or independent?
- What is the optimal set of conservation practices and what is their optimal placement within the watershed in order to achieve water quality goals or to provide acceptable reductions in water quality impairments?

Most, but not all, of the NIFA–CEAP studies used watershed modeling for predictive, exploratory, and/or knowledge-building purposes to address questions posed in the request for proposals. In the predictive mode, application of models hinged on the following goals:

- Predicting fluxes of water and contaminants under historic land use, climatic, and land treatment conditions at varying spatial and temporal scales
- Identifying pollutant sources

(c) SWCS. For Individual Use Only

- Quantifying the effects of existing and potential land treatment on sediment, P, N, and atrazine loads at the watershed scale
- Determining the optimal suite of practices to achieve water quality goals at the lowest cost

In the exploratory mode, models supported examining the following:

- Geospatial and temporal factors that determine effectiveness of existing or potential practices
- Tradeoffs between water quality and socioeconomic criteria for implementation of practices at the watershed-scale
- The role of climate variability on projected effectiveness of practices
- The influence of parameter, structural, and measurement uncertainties on projected effectiveness of practices

A few studies also used models to learn about the behavior of their watershed systems. The application of watershed modeling for knowledge-building investigated the following:

- Critical hydrological processes that control the variability of runoff and pollutant source areas at varying spatial and temporal scales
- Proper standard procedures for numerical representation of practices
- How projected effectiveness of practices varies with the resolution of soil and land-use data used for model development
- How spatial scale of representation of practices affects their projected effectiveness

The use of models for collaborative learning was not exploited fully in any of the projects, but extension of scientific knowledge gained from models served as a mechanism for promotion of effective land treatment options.

Summary of Water Quality Modeling Activities in the National Institute of Food and Agriculture–Conservation Effects Assessment Project

In the absence of established guidelines for application of models to support conservation assessment and planning at the watershed-scale, modeling activities among the NIFA–CEAP watershed studies were not necessarily consistent. However, all modeling studies included the following activities:

- Data collection: geospatial data, e.g., soils, land cover, elevation, hydrography; watershed data about chemical use, conservation practices, and point sources; climate data; and discharge and ambient water quality data
- Monthly/annual load estimation from intermittent, instantaneous concentration data
- Model selection and development: SWAT in nine studies, AnnAGNPS in two studies, WEPP in one study, GWLF in one study, European Sediment Network (SedNet) in one study, and Conservational Channel Evolution Pollutant Transport System (CONCEPTS) in one study
- Model calibration and testing
- Scenario analysis

A few studies carried out other activities related to modeling, including the following:

- Modification of existing algorithms or incorporation of new ones in the selected models: SWAT in three studies, WEPP in one study, and GWLF in one study
- Sensitivity analysis for identification of important model parameters: six studies

(c) SWCS. For Individual Use Only

- Uncertainty analysis for exploring the role of input (climate and geospatial data), parameter, and structural uncertainties on the estimated environmental benefits of practices: two studies
- Genetic-algorithm based multiobjective optimization for cost-effective allocation of practices to meet water quality targets at the lowest cost: two studies

The principal modeling activities for each of the 13 NIFA–CEAP watershed studies are summarized in table 5.1. Specific details of modeling activities are included in Part II: Chapters 9 to 21.

Results: Contributions to Knowledge/Science from Watershed Modeling

Projected Environmental Benefits of Conservation Practices

All modeling studies reported some quantitative benefits for current and/or potential conservation practices in terms of reductions in sediment, nutrients, or pesticide loadings from fields or yields at the watershed outlet. Table 5.2 provides a summary of the results reported for effects of current or potential conservation practices. Diligent efforts were devoted toward calibration and testing of the selected models. Most of the studies were able to obtain satisfactory calibration results for flow and nutrient predictions according to the performance criteria suggested by Santhi et al. (2001) and Moriasi et al. (2007).

Although calibration provided circumstantial evidence of model validity for predicting hydrologic and water quality fluxes according to available flow and water quality data, additional efforts must be focused on key uncertainties. None of the NIFA–CEAP watershed studies, with the exception of ongoing efforts in the Eagle Creek Watershed in Indiana, reported the level of confidence associated with the projected effectiveness of practices. The role of uncertainties in model predictions ought not to be discounted. For example, the Lincoln Lake Watershed in Arkansas modeling study concluded that variability in weather has significant implications for sediment, N, and P losses, and performance of conservation practices could markedly vary under different climatic conditions (Gitau et al. 2011). Other studies revealed the influence of watershed subdivision and model configuration (e.g., number of subbasins and hydrologic response units), along with model parameter uncertainties on evaluation of practices (Arabi et al. 2006b, 2007b, 2008).

Key Geospatial and Temporal Factors and Critical Source Areas

Modeling showed that several factors influence water quality benefits of conservation practices. The hydrologic and water quality responses of a field plot in the Goodwater Creek Watershed in Missouri to changes in configuration and location of the backslope were simulated using a calibrated APEX model for the field. Runoff volume and atrazine concentrations at the field outlet increased with increasing length of backslope, while the steepness was kept constant. The proximity of the backslope to the field outlet was also identified as an important geospatial factor. Conversely, foot slope characteristics had minimal effects on simulated atrazine losses. This information could improve the performance of conservation practices installed on fields to reduce atrazine losses.

(c) SWCS. For Individual Use Only

Table 5.1
Summary of principal water quality modeling activities of the National Institute of Food and Agriculture–Conservation Effects Assessment Project.

Project	Principal activities
Arkansas	• Prediction: (1) quantitative estimates for effects of 171 potential land treatment options on sediment, total N, and total P loadings; (2) the optimal set of land treatment options for nutrient reduction. • Exploration: (1) impacts of the resolution of geospatial data (soils, land use, and elevation) and climate (precipitation) variability on estimated benefits of potential land treatment options; (2) tradeoffs between nutrient (total N and total P) loadings and costs; (3) roles of conservation practice type and interactions between practices on nutrients.
Georgia	• Prediction: quantitative estimates for effects of current and potential conservation practices on total N and total P loads. • Exploration: (1) priority subwatersheds for targeted implantation of practices and (2) roles of first and second order streams on nutrient transport.
Idaho	• Exploration: (1) roles of channel processes and legacy sediments on sediment loads and (2) geospatial/temporal factors that determine effectiveness of practices.
Indiana	• Prediction: quantitative estimates for effects of current and potential conservation practices on total N, nitrate, and atrazine loads. • Exploration: (1) uncertainty of projected effectiveness of practices due to input, parameter, structural, and measurement uncertainties; (2) critical natural processes and key management actions (subsurface drainage systems) that control movement of N and atrazine; (3) how availability of multivariable multisite water quality data impacts model parameter estimation.
Iowa	• Prediction: quantitative estimates for effects of current and potential conservation practices on nitrate loads.
Kansas	• Prediction: (1) quantitative estimates for effects of current and potential conservation practices on reduction of sediment and total P loads and (2) critical sediment source areas. • Knowledge building: comparative study of the suitability of SWAT and AnnAGNPS models for representation of flow, sediment, and P process.
Missouri	• Prediction: (1) quantitative estimates for effects of current and potential conservation practices on sediment, N, P, and atrazine loads; and (2) critical atrazine and nitrate source areas. • Exploration: role of landscape position on the effectiveness of practices. • Knowledge building: (1) suitability of SWAT and APEX models for conservation effects assessment at the field and watershed scales, (2) role of claypan soil horizons on hydrologic and water quality responses, and (3) modification of SWAT algorithms for capturing claypan soil effects on water table and shallow groundwater return. • Collaborative learning: participatory scenario building in cooperation with the steering committee and the farm panel.
Nebraska	• No modeling activities were carried out.
New York	• Prediction: (1) quantitative estimates for effects of current and potential conservation practices on dissolved P and total P loads and (2) critical dissolved P source areas. • Exploration: role of landscape position on the effectiveness of manure management. • Knowledge building: (1) suitability of SWAT, GWLF, a reconceptualized SWAT model, and a reconceptualized GWLF model for critical source identification and (2) role of soil saturation conditions on variable source areas. • Collaborative learning: cooperative assessment of exclusion of hydrologically sensitive areas from manure application.
Ohio	• Modeling activities were aborted due to difficulties in application of AnnAGNPS.
Oregon	• A SWAT model had been developed for the watershed, but the model and its outcomes were not applied to address the objectives of this project.

Continued

(c) SWCS. For Individual Use Only

Table 5.1 Continued

Project	Principal activities
Pennsylvania	• Exploration: sediment source identification (hillslope, gully, and steam bed and bank erosion) and importance of upland and within-channel nonpoint source pollution control measures on sediment loads.
Utah	• No modeling activities were carried out.

Notes: N = nitrogen. P = phosphorus.

Table 5.2
Summary of projected conservation practice effectiveness determined by modeling studies in the National Institute of Food and Agriculture–Conservation Effects Assessment Project. Reduction values for each scenario were computed compared to the baseline scenario. Negative reduction values indicate an increase in the pollutant of interest as a result of implementation practices.

Conservation practice (scenario)	Pollutant	Reduction (%)	Watershed
No-tillage on 50% watershed area (current)	Soil erosion	34	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 100% watershed area (future)	Soil erosion	61	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 50% watershed area (current)	Sediment yield	19	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 100% watershed area (future)	Sediment yield	43	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 50% watershed area (current)	Total P yield	14	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 100% watershed area (future)	Total P yield	27	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 50% watershed area (current)	Dissolved P yield	–13	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage on 100% watershed area (future)	Dissolved P yield	–30	North Fork Ninnenscah Watershed, Kansas (Chapter 14)
No-tillage (current)	Soil erosion	33	Red Rock Creek Watershed, Kansas (Chapter 14)
No-tillage (current)	Sediment yield	20	Red Rock Creek Watershed, Kansas (Chapter 14)
No-tillage (current)	Total P yield	15	Red Rock Creek Watershed, Kansas (Chapter 14)
Grassed waterways (current)	April atrazine concentration	19	Goodwater Creek Watershed, Missouri (Chapter 15)
Grassed waterways (current)	May atrazine concentration	26	Goodwater Creek Watershed, Missouri (Chapter 15)
Grassed waterways (current)	June atrazine concentration	18	Goodwater Creek Watershed, Missouri (Chapter 15)

Continued

(c) SWCS. For Individual Use Only

Table 5.2 Continued

Conservation practice (scenario)	Pollutant	Reduction (%)	Watershed
Increase in corn acres (1992 to current)	Atrazine load	57	Goodwater Creek Watershed, Missouri (Chapter 15)
Increase in corn acres (1992 to current)	Atrazine concentration	85	Goodwater Creek Watershed, Missouri (Chapter 15)
Grassed waterways (current) and increase in corn acres (1992 to current)	Atrazine load	16	Goodwater Creek Watershed, Missouri (Chapter 15)
Grassed waterways (current) and increase in corn acres (1992 to current)	Atrazine concentration	34	Goodwater Creek Watershed, Missouri (Chapter 15)
Increase in conservation tillage (1992 to current)	Atrazine concentration	6	Goodwater Creek Watershed, Missouri (Chapter 15)
Suite of practices: protection of vegetated buffers along streams, reduction of P application, prevention of manure and fertilizer application in areas prone to runoff generation (current)	Total P concentration	44	Cannonsville Watershed, New York (Chapter 17)
Suite of practices: protection of vegetated buffers along streams, reduction of P application, prevention of manure and fertilizer application in areas prone to runoff generation (current)	Total dissolved P yield	42	Cannonsville Watershed, New York (Chapter 17)
Changes in land use, mostly from cropland and pastureland to forest (1992 to current)	Total dissolved P yield	20	Cannonsville Watershed, New York (Chapter 17)
Suite of practices: protection of vegetated buffers along streams, reduction of P application, prevention of manure and fertilizer application in areas prone to runoff generation (current)	Dissolved P yield	37	Dairy Farm Watershed, New York (Chapter 17)
Suite of practices: protection of vegetated buffers along streams, reduction of P application, prevention of manure and fertilizer application in areas prone to runoff generation, all manure P exported from the watershed (current)	Dissolved P yield	> 70	Dairy Farm Watershed, New York (Chapter 17)
Conversion of all crop land to forest (potential)	Sediment load	90	Spring Creek Watershed, Pennsylvania (Chapter 20)
Conversion of all crop land to developed (potential)	Sediment load	–26	Spring Creek Watershed, Pennsylvania (Chapter 20)
Buffer strips (potential)	Total N loading	46	Lincoln Lake Watershed, Arkansas (Chapter 9)
Litter application timing and amount (potential)	Total N loading	–4 to 33	Lincoln Lake Watershed, Arkansas (Chapter 9)

Continued

(c) SWCS. For Individual Use Only

Table 5.2 Continued

Conservation practice (scenario)	Pollutant	Reduction (%)	Watershed
Optimal grazing management (potential)	Total N loading	35	Lincoln Lake Watershed, Arkansas (Chapter 9)
Suite of practices (potential)	Total N loading	–8 to 46	Lincoln Lake Watershed, Arkansas (Chapter 9)
Buffer strips (potential)	Total P loading	66	Lincoln Lake Watershed, Arkansas (Chapter 9)
Litter application timing and amount (potential)	Total P loading	–6 to 28	Lincoln Lake Watershed, Arkansas (Chapter 9)
Optimal grazing management (potential)	Total P loading	–13 to 25	Lincoln Lake Watershed, Arkansas (Chapter 9)
Suite of practices (potential)	Total N loading	–13 to 66	Lincoln Lake Watershed, Arkansas (Chapter 9)
Buffer strips (potential)	Mineral P loading	71	Lincoln Lake Watershed, Arkansas (Chapter 9)
Litter application timing and amount (potential)	Mineral P loading	0 to 56	Lincoln Lake Watershed, Arkansas (Chapter 9)
Optimal grazing management (potential)	Mineral P loading	49	Lincoln Lake Watershed, Arkansas (Chapter 9)
Suite of practices (potential)	Total N loading	–4 to 71	Lincoln Lake Watershed, Arkansas (Chapter 9)
Suite of practices (current)	Soil erosion	22	Little River Experimental Watershed, Georgia (Chapter 10)
Suite of practices (current)	Total N loading	6	Little River Experimental Watershed, Georgia (Chapter 10)
Suite of practices (current)	Total P loading	22	Little River Experimental Watershed, Georgia (Chapter 10)
Suite of practices (current)	Sediment yield	20	Little River Experimental Watershed, Georgia (Chapter 10)
Suite of practices (current)	Total N yield	4	Little River Experimental Watershed, Georgia (Chapter 10)
Suite of practices (current)	Total P yield	18	Little River Experimental Watershed, Georgia (Chapter 10)
Grassed waterways, terraces, conservation tillage, and contour farming on 100% of agricultural areas on cropland with no land treatment (potential)	Sediment loading	55	Little River Experimental Watershed, Georgia (Chapter 10)

Continued

(c) SWCS. For Individual Use Only

Table 5.2 Continued

Conservation practice (scenario)	Pollutant	Reduction (%)	Watershed
Grassed waterways, terraces, conservation tillage, and contour farming on 100% of agricultural areas on cropland with no land treatment (potential)	Total N loading	1	Little River Experimental Watershed, Georgia (Chapter 10)
Grassed waterways, terraces, conservation tillage, and contour farming on 100% of agricultural areas on cropland with no land treatment (potential)	Total P loading	56	Little River Experimental Watershed, Georgia (Chapter 10)
Nutrient management, 30% reduction of nutrient application rate (potential)	Total N loading	10	Little River Experimental Watershed, Georgia (Chapter 10)
Nutrient management, 30% reduction of nutrient application rate (potential)	Total P loading	4	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (current)	Sediment loading	75	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (current)	Total N loading	32	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (current)	Total P loading	76	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (potential)	Sediment loading	80	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (potential)	Total N loading	37	Little River Experimental Watershed, Georgia (Chapter 10)
Riparian forest buffer (potential)	Total P loading	80	Little River Experimental Watershed, Georgia (Chapter 10)
Conversion of all highly erodible lands to grassland	Nitrate loading	47	Squaw Creek Watershed, Iowa (Chapter 11)
Conversion of all row crops to grassland in upper basin subbasins	Nitrate loading	16	Squaw Creek Watershed, Iowa (Chapter 11)
Conversion of row crops in floodplain subbasins to grassland	Nitrate loading	8	Squaw Creek Watershed, Iowa (Chapter 11)
Conservation Reserve Program to cropland (1990 to 2005)	Nitrate yield	30	Squaw Creek Watershed, Iowa (Chapter 11)

Notes: N = nitrogen. P = phosphorus.

(c) SWCS. For Individual Use Only

Soil properties, in particular, saturated hydraulic conductivity, depth to restrictive soil layers (e.g., claypan soil horizons), and topographic characteristics of fields were identified as primary controls for generation of runoff and movement of chemicals in runoff in the Goodwater Creek Watershed in Missouri, the Paradise Creek Watershed in Idaho, and the Cannonsville Reservoir in New York NIFA–CEAP watershed studies. Indices, similar to the soil topographic index, were developed by combining these factors to identify hydrologically sensitive areas that are prone to runoff generation. These serve as critical sources of pollutants, particularly for chemicals that move primarily in dissolved form in runoff. Pollutant source identification using these simple indices agreed overall with results from SWAT, APEX, and WEPP models.

Application of reconceptualized SWAT and GWLF models in the Cannonsville Reservoir Watershed in New York clearly demonstrated how the extent of runoff generation in the watershed is controlled by soil saturation conditions. Therefore, critical source areas within the watershed vary spatially and temporally. The project was able to recommend that targeted implementation of conservation practices should focus on variable source areas where the majority of runoff is likely to be generated and can be supported by identification of variable source areas using these enhanced modeling tools.

The Arkansas, Missouri, and New York NIFA–CEAP modeling studies revealed that the performance of conservation practices can vary as a result of their interactions. For example, analysis of 171 conservation scenarios in the Lincoln Lake Watershed in Arkansas showed that buffer strips were more effective in reducing total N, mineral P, and total P losses when combined with optimal grazing scenarios rather than overgrazing scenarios. Also interactions between timing of litter application and buffer strips influenced the effectiveness of practices in removing total N. The Missouri Project found that fate and transport of atrazine in the Goodwater Creek Watershed were dominated by the interaction between weather and applications dates. The New York Project modeling results confirmed that the fate and transport of dissolved P was determined by the interactions between manure application date and weather conditions.

Other modeling studies highlighted the importance of proximity of fields to first and second order streams, along with characteristics of riparian buffers along these streams. The SWAT simulations in the Eagle Creek Watershed in Indiana underscored the high efficiency of first order streams in removing N and atrazine. These findings are supported by findings from application of the SPARROW model in the Mississippi River Basin (Alexander et al. 2000, 2007). The implication of these findings is that implementation of conservation practices in close proximity to larger perennial streams could provide higher water quality benefits than implementation in areas close to intermittent streams with shallow water depths. The SWAT model application in the Little River Experimental Watershed in Georgia also indicated the effectiveness of riparian forest buffers along first and second order streams at removing nutrients.

Modeling in the Little River Experimental Watershed in Georgia suggested targeting conservation practices would result in meeting sediment and nutrient targets at lower costs. Results indicated that targeted implementation of practices in critical source areas (identified from model application) would be more effective than targeting according to the stream order. However, prioritization of first and second order stream subwatersheds for targeted implementation practices was substantially more effective than a random allocation of practices.

(c) SWCS. For Individual Use Only

Impacts of Land-Use Change and Conservation Practices on Channel Erosion

Several NIFA–CEAP modeling studies evaluated the role of channel processes on sediment yield at the watershed outlet. According to long-term simulations of channel erosion processes with the CONCEPTS model in the Paradise Creek Watershed in Idaho, bed/bank erosion from the channel network is a major source of watershed sediment loads. The modeling study concluded that flushing legacy sediments in the channel network could take nearly 19 years (Newson 2007).

More importantly, analysis of monitoring data and modeling results in Iowa, Idaho, Kansas, Missouri, and Pennsylvania exposed the vulnerability of streams to erosion in the study watersheds after implementation of soil erosion control practices and conversion of agricultural fields to natural prairie or forest. Reduction of soil erosion due to implementation of conservation practices or land-use changes resulted in lower sediment concentrations in stream water. Without substantial reductions in discharge, however, streamflow maintains its transport capacity. The deficit in the balance of transport capacity of streamflow and sediment concentration in steam water is compensated by erosion of channel bed and banks. These findings highlight the importance of a holistic system approach to pollution control rather than a field by field, pollutant by pollutant approach. Modeling is a powerful tool for conducting a holistic analysis of a complex system containing many variables.

Optimal Set of Conservation Practices for Achieving Water Quality Targets

Assessment of the cost-effectiveness of all possible targeting scenarios could be cumbersome, if not infeasible. Two of the NIFA–CEAP watershed studies used integrated simulation-optimization frameworks to determine the optimal set of practices for water quality protection. The Lincoln Lake Watershed modeling study in Arkansas linked the SWAT model with a nondominated sorting genetic algorithm-II (NSGA-II) to determine low-cost spatially explicit options for litter application timing and amount, buffer strips, and grazing management (Rodriguez et al. 2011). The objective functions in the optimization formulation consisted of minimizing pollutant (total N and total P) loads and minimizing total watershed costs. The study revealed that implementation of all practices in the low-cost solution would reduce total P by 76% while increasing total watershed cost by less than 2%, when compared to the baseline scenario.

A similar simulation-optimization framework was developed by the Iowa NIFA–CEAP team to search for the most cost-effective set of conservation practices in the Squaw Creek Watershed for reduction of total N (Rabotyagov et al. 2010b). Two targets for N reduction were used in the objective function: reduction of mean annual N loadings and a safety-first type of constraint aiming to achieve the desired reduction levels while incorporating the stochastic nature of climate data in the decision-making process. The lowest marginal cost of reducing mean annual N loadings by 30% on a per-cropland acre basis was approximately US$1 per year. The mix and location of practices changed as the pollutant reduction target moved, which was consistent with findings from the Calapooia Watershed Project in Oregon.

Results in the Squaw Creek in Iowa indicated that the cost of implementing the climate-resilient optimal set of practices would dramatically increase, compared to optimizing for mean annual N loadings. An interesting finding of the study was that although the cost of implementation of practices would increase with more stringent N reduction targets, the type of practices

(c) SWCS. For Individual Use Only

in the climate resilient optimal solutions would not necessarily change. Terraces and grassed waterways, when used in combination with no-tillage, appeared consistently in the optimal solutions for reduction of N loadings in the watershed. These findings could help watershed managers focus their efforts on promotion of a few practices that consistently rank among the most cost-effective options.

The same framework developed by the Iowa NIFA–CEAP team was also used to develop a tradeoff frontier for the entire Upper Mississippi River Basin to determine least-cost solutions for achieving water quality targets (Rabotyagov et al. 2010a). The study concluded that reduction of N yield by 30% at the basin outlet could be achieved at a minimum cost of US$1.4 billion per year. Implementation of the optimal set of conservation practices would also result in reduction of total P yield by 36%. Similarly, the study concluded that reduction of total P yield by 30% would cost US$370 million per year in the optimal case, while a 9% N reduction would be achieved at the same time.

In general, the use of simulation-optimization frameworks was found to be an efficient approach for identification of critical sources areas and allocation of an optimal suite of practices. These tools can help decision makers evaluate feasibility of meeting water quality targets at different budget levels, prioritize subwatersheds for implementation of watershed plans, and identify the optimal location and type of conservation practices.

Lessons Learned

As described in previous sections, watershed modeling served as the primary tool in most of the NIFA–CEAP watershed studies for evaluation of existing and potential conservation practices; examination of the roles of location, timing, and interactions between practices on water quality; and identification of optimal sets of conservation practices to meet water quality targets. The modeling efforts collectively advanced modeling tools and their applications for watershed-scale assessment of critical source areas and conservation practices and also revealed critical knowledge gaps and needs for credible evaluation of land treatment options at the watershed scale.

Expertise Requirements for Development and Application of Watershed Models

Considerable expertise was required to properly apply watershed models in the NIFA–CEAP studies. This expertise can be grouped into three categories: (1) comprehensive knowledge of hydrologic and biogeochemical processes and essential characteristics of the watershed system under study, (2) computer programming and geographic information systems skills, and (3) adequate knowledge of statistical concepts for exploratory data analysis. Also, optimization studies, when conducted, required significant expertise in mathematical programming, particularly evolutionary algorithms. Therefore, NIFA–CEAP modeling activities required close collaboration between researchers and scientists from traditionally disparate disciplines.

Lack of essential expertise could severely jeopardize successful accomplishment of modeling goals and objectives. For example, the modeling component of the Ohio Project was provided by researchers at another institution due to the unavailability of modeling expertise at the lead institution. The selected model (AnnAGNPS) could not be properly implemented due to lack of adequate climate, elevation, and land-use data. More importantly, it became evident

(c) SWCS. For Individual Use Only

that the selected model also lacked the capacity to represent essential characteristics of the system, including subsurface drainage systems. The modeling effort was aborted near the completion of the project. Resources allocated to the modeling effort did not yield direct benefits for conservation assessment in the watershed. The Oregon study experienced similar issues due to disconnect between the natural resource scientists and the modeling group.

Application of comprehensive watershed models in other locations and by other organizations requires availability of similar expertise. Agencies that choose to use complex watershed models must ensure that their personnel are trained for the development, calibration, and credible application of watershed models. Otherwise, significant investments may be made in ill-advised adoption of modeling tools that either do not serve their intended goals or that cannot be used due to lack of adequate expertise. The consequences of making decisions based on uncorroborated models may also include investments wasted in management options that fail to perform to predicted outcomes.

Model Selection and Enhancement for Conservation Planning

Existing expertise within the project teams, availability of technical support from model development teams, proper documentation, and the ease of use of the model interface were identified as the primary reasons for the selection of particular models in the NIFA–CEAP watershed studies. Among all other modeling tools, the SWAT model was the favorite choice in most of the projects. The SWAT model has been applied broadly within the United States and internationally, and the soundness of its algorithms for simulating hydrologic and water quality processes has been extensively examined. Successful calibration of the model in the study watersheds also confirmed the performance validity of its streamflow and water quality predictions at the watershed outlet. Another attractive feature of SWAT and also of APEX, for conservation assessment, is the availability of explicit options for representation of a variety of tillage and nutrient management operations.

Some of the projects used multiple models for analysis. The Kansas Project compared the performances of the SWAT and AnnAGNPS models in the Cheney Lake Watershed. Results indicated that SWAT performed better for streamflow, sediment, and nutrient predictions at the watersheds outlet. Other studies in USDA Agricultural Research Service benchmark watersheds have arrived at the same conclusion. The Spring Creek Watershed Project in Pennsylvania compared the estimated sediment yields using SWAT and SedNet. The results showed considerable discrepancies between the outcomes of the two models. But the models, taken together, confirmed that sediment yield in the watershed was dominated by channel bank erosion. When pointing to the same conclusions, application of multiple models to address similar goals could establish confidence in their outcomes and model-based recommendations.

The Goodwater Creek Watershed Project in Missouri opted to use SWAT and APEX in conjunction in order to circumvent the limitations of SWAT in routing water, sediment, and chemicals overland from field to field. On the other hand, SWAT provides the capacity for simulating movement of water and chemicals in streams and is more suitable for studies in larger watersheds. Integrated APEX/SWAT modeling was also used in the national CEAP in the Upper Mississippi River Basin. This approach shows promise for conservation assessment at both field and watershed scales.

The ever-increasing temptation to link models or to incorporate new algorithms for decision-making purposes drives rising model complexity. A more complex model that includes more

(c) SWCS. For Individual Use Only

processes and parameters may perform better. However, in doing so, the propensity for parameter interactions and nonuniqueness tends to increase (figure 5.2). A serious issue then arises: are available flow and water quality data sufficient for parameter identifiability and reduction of uncertainties associated with the more complex model?

Reconceptualization of SWAT by the New York NIFA–CEAP Cannonsville Reservoir team for incorporation of variable source areas and also incorporation of new algorithms for representation of shallow restrictive soil layers (i.e., claypan) by the Missouri Goodwater Creek Watershed team enhanced the performance of the model for streamflow, dissolved P, and atrazine loads at the watershed outlets. Similarly, incorporation of variable source areas in the WEPP model and its linkage with CONCEPTS culminated in substantial improvements in the sediment load predictions. Additional studies should be carried out to examine whether improving the model performance has increased the uncertainties in model parameters and predictions.

The Eagle Creek Watershed NIFA–CEAP in Indiana demonstrated the importance of availability of strategic flow and water quality data at various locations within the watershed for parameterization of the SWAT model. The study revealed that the nonuniqueness problem can result in inaccurate identification of critical pollutant source areas and pathways. An important lesson learned from the study was how general knowledge of watershed response can improve the calibration procedure. For example, it became evident that imposing acceptable ranges for the contribution of subsurface drainage systems to nitrate yield was vital for credible estimation of model parameters. Model calibration without imposing these constraints also resulted in very good performance for all variables (streamflow, nitrate, and atrazine) at all sampling stations (one streamflow gage and ten water quality sampling locations) according to the performance criteria recommended by Moriasi et al. (2007). However, the model predicted that nearly 80% of nitrate was delivered to the stream via surface runoff pathways. This prediction contradicts the general understanding of the system according to past observations that suggest that nitrate from subsurface drainage systems contributes to nearly 75% to 95% of nitrate yield.

Measurements of important within-watershed fluxes, such as denitrification and nitrate from subsurface drainage systems, are generally not available. Therefore, these outputs cannot be included in the objective function(s) for calibration purposes. But they can be represented in constraint functions, which result in rejecting parameter sets that do not satisfy acceptable ranges for these outputs. The modeling community, as a whole, should contribute to the development of these measures that can be used across the world.

Scientific Basis of Existing Models for Conservation Planning

One of the most important lessons learned from NIFA–CEAP watershed studies is the identification of deficiencies in our knowledge and in existing modeling tools for representation of critical natural processes and key management actions at the watershed scale. In general, the complexity and nonlinear nature of watershed processes overwhelm the capacity of existing modeling tools to reveal the water quality impacts of conservation practices.

Overland routing, i.e., moving water and pollutants downgradient from field to field, is a key modeling issue. Overland processes play a significant role in the delivery of sediment and contaminants from upland areas to the stream network. Incorporation of overland processes or lack thereof could be particularly consequential for capturing interactions between conservation practices. Most exiting watershed models trade off representation of overland processes for

(c) SWCS. For Individual Use Only

improving computational efficiency. Accurate appraisal of the effects of buffer strips and riparian zones calls for inclusion of overland processes in watershed models.

Other major modeling needs for credible assessment of conservation practices at the watershed scale include the following:

- Gully erosion
- Legacy sediments in the channel network
- Overland routing
- In-stream biogeochemical processes
- Subsurface processes
- Sound numerical algorithms for representation of conservation practices

Consider the Cheney Lake Watershed in Kansas as an example. According to field observations and measurements, gully erosion from agricultural fields has been identified as a primary mechanism for transport of sediment and P to the lake. The SWAT model was used to simulate hydrologic and water quality fluxes in the system, but it does not have the capacity to simulate ephemeral gully erosion, one of the essential characteristics of the system. Similarly, nearly 15% of sediment loads originate from stream banks in the watershed. The SWAT model cannot adequately capture movement of legacy sediments in the channel network. Long-term evolution of the channel network due to sediment deposition and bed/bank erosion cannot be captured in SWAT. (A new version of SWAT called SWAT-DEG was developed that captures channel evolution processes [Allen et al. 2002]. These developments were grafted into the SWAT 2009 model). Therefore, calibration of the model for sediment and P yields at the watershed outlet resulted, most probably, in an overestimation of sediment and P loadings from upland areas (e.g., sheet and rill erosion). Overestimating sheet erosion, in turn, could affect the estimated benefits of conservation practices, such as conservation tillage and terraces.

The current suite of available watershed models cannot adequately capture the interactions between upland (hill slope and field scale) pollutant loadings and within-channel sediment/nutrient processes. Reduction of sheet erosion from upland areas may expedite channel degradation and erosion. Therefore, watershed-scale analysis of long-term benefits of conservation practices through modeling exercises, as solicited in the NIFA–CEAP request for proposals, necessitate improving the channel routing component of the available widely used models or development of new models capable of capturing such interactions.

The Cannonsville Reservoir Watershed in New York presents an example where significant advances in modifying SWAT and GWLF were needed to include saturation excess runoff processes, i.e., variable source area activity. The participation between local and state managers, farmers, extension personnel, and university scientists facilitated a reconceptualization of available models to more accurately represent hydrological and water quality processes in the system. Otherwise, the modeling results pertinent to the landscape position of areas with high runoff potential and/or that are susceptible to erosion could be misleading, which in turn results in implementation practices in "wrong" locations.

Simulation of groundwater processes and surface-subsurface interactions with existing watershed models is a challenge. Comprehensive watershed models, such as SWAT, include simplistic empirical, lumped algorithms for representing subsurface movement of water and chemicals. Therefore, prediction of impacts of conservation practices on constituents that are transported primarily via groundwater pathways remain elusive. For example, effects of the land-use change from agriculture to natural prairie in the Walnut Creek Watershed in Iowa pre-

(c) SWCS. For Individual Use Only

dicted by SWAT did not match observed trends. Groundwater models, such the US Geological Survey Modular Three-Dimensional Groundwater Flow Model (MODFLOW) (Harbaugh 2005), provide a more accurate representation of groundwater processes, but this type of model is complex and difficult to use. Moreover, most groundwater models do not have options for simulation of chemicals and cannot represent land management options. Improvements in the subsurface component of existing watershed models, such as APEX, SWAT, and AnnAGNPS, are essential for assessing conservation effects on groundwater quality.

Numerical Representation of Conservation Practices

Incorporation of unambiguous procedures for representation of land treatment options must be considered as a priority in the development of the next generation of watershed models. The inconsistency in practice representation in models can result in markedly different estimated environmental benefits. Availability of explicit functionalities for simulation of management options within watershed models could minimize the subjectivity of the procedure. Numerical procedures adopted for representation of conservation practices were not consistent among the NIFA–CEAP watershed studies.

For example, the Iowa Project used the Universal Soil Erosion Equation (USLE) practice factor for representation of grassed waterways. This factor was originally developed to account for the influence of terraces, strip-cropping, and residue management on sheet erosion; suggested values for grassed waterways or filter strips, among other practices, are not available. Therefore, the appropriateness of representing grassed waterways by changing the USLE practice factor is not widely accepted and could be a source of considerable error.

The method for representation of grassed waterways and residue management in the Little River Experimental Watershed CEAP in Georgia was not consistent with other NIFA–CEAP studies. In fact, such consistency was lacking among all projects, and therefore, comparison of estimated benefits of the same practice from different projects was challenging.

The latest version of SWAT (SWAT 2009) grafted the work of White and Arnold (2009) into SWAT for enhancing the representation of filter strips. Also, specific functionalities were embedded in SWAT 2009 to represent grassed waterways using a process-based approach that routes water and pollutant loading from fields before they enter the stream network. Explicit options for terracing operations, tile drainage, contouring, filter strips, and strip cropping were also made available (Arnold et al. 2011). The Indiana Eagle Creek Watershed and the Missouri Goodwater Creek Watershed NIFA–CEAP watershed studies took advantage of these new model developments and reported significant improvements in the representation of practices.

Finally, none of the existing modeling tools that are widely used for conservation assessment at the watershed scale can account for operation and maintenance of practices. As described in Chapter 3, implementation of conservation practices is not necessarily followed by proper operation and maintenance in many cases. Lack of proper maintenance can degrade the benefits of practices (Bracmort et al. 2006). Therefore, greater emphasis must be placed on incorporating the degradation of practices over time in the modeling process.

Availability of Watershed Data

Access to information about the exact location of conservation practices is essential for evaluation of their effects, particularly when the model application goal is to examine the role of location, timing, and suite of practices on water quality. In NIFA–CEAP watersheds, with

(c) SWCS. For Individual Use Only

the exception of three projects, difficulties in accessing such information essentially prevented modeling the effects of practices in a spatially explicit manner. Information about the amount and timing of nutrient and pesticide applications was also vital for evaluating the performance of land treatment options at the watershed scale.

Model Application in Mixed Land-Use Watersheds

Effects of changes in land use on fluxes of water and pollutants must be considered when evaluating the performance of land treatment options. The Lincoln Lake Watershed in Arkansas documented that increased sediment and nutrient loads in some tributaries could be mainly attributed to urban development. Rapid urbanization over the course of project assessment period was also documented in the Eagle Creek Watershed in Indiana, though the impacts on atrazine and nitrate loads were minimal. In the New York Project, modeling results indicated that land-use change from pasture to forest reduced total P yield to the Cannonsville Reservoir by 20% over the 1992 to 2005 period, while land-use change and conservation practices in combination reduced total P yield to the reservoir by 45%. Models that are used for conservation assessment and planning must include the capacity for representing temporal changes in land use, particularly in watersheds where land use evolves over the course of the assessment period. This consideration motivated incorporation of new algorithms for representation of land-use change in SWAT 2009.

Discrepancy Between Modeling Outcomes and Trends from Monitoring

In general, the environmental benefits of land treatment options from NIFA–CEAP modeling studies were grossly overestimated in comparison with monitored trends in reductions of sediments and nutrients. Several of these factors may be responsible, including the following:

- Modeling uncertainties
- Lack of sensitivity and/or statistical power in the monitoring program
- Lag time between implementation of conservation practices and reduction of pollutants at the watershed outlet
- Degradation of practices due to operation and maintenance issues
- Unaccounted disturbances in the systems

Enhancements in the scientific basis of modeling tools along with adoption of proper application protocols can address most of these issues. Several approaches have been suggested to reduce parametric and structural modeling uncertainties, while making best use of available data. Ongoing model development efforts have focused on incorporating more accurate representation of overland, subsurface, and channel process in commonly used watershed models. Ideally, these improvements will address the issues of lag time and legacy contaminants. Finally, our enhanced watershed models can be used in exploratory mode as tools for forensic investigation of unaccounted disturbances and sources of pollution and can also provide insight needed for improving monitoring programs.

(c) SWCS. For Individual Use Only

Recommendations: What would make Simulation Modeling Better for Conservation Assessment and Planning at the Watershed Scale?

Enhancing modeling capacity for conservation assessment and planning can only be achieved via a concerted effort among model development teams, users of models, and agencies that require models to support management actions or decisions. Enhancing the scientific basis of watershed models is necessary for conservation effect assessment but is not sufficient to ensure their proper applications. Users must adopt good modeling practices to successfully achieve study objectives. Funding organizations can also play a critical role by designing proper program synopses and questions that can be addressed using watershed models.

Model Development

The primary role of model development teams and the broader research community is establishing credibility for model algorithms. In this context, development, incorporation, and communication of sound algorithms are the key tasks. Pressing algorithm development needs include representation of gully erosion processes, fate and transport of legacy sediments in the stream network, overland routing processes, and land treatment options. Other important issues to address include enhancing representation of in-stream biogeochemical processes and subsurface processes.

Developers and researchers must insist upon and facilitate proper use of models. The model user community may not have the capacity or resources to link commonly used models with calibration (automatic or manual), sensitivity analysis, and uncertainty analysis procedures, among other relevant analytical tools. Such linkages will ensure users have access to tools needed for proper application of models. Model version control is another critical issue. A significant number of researchers continue to improve existing models. These improvements should be reconciled seamlessly with the original models after proper peer-review process and quality control and quality assurance protocols have been applied.

Finally, the NIFA–CEAP experience attests to the importance of modeling support for selection and application of watershed models. Model development teams should provide technical support via help-desk services, theoretical and user's guide documentations, and training for application of models. An inventory of successful case studies could also serve as a mechanism for learning good modeling practices.

Model Application

Users must establish credibility for model application in a watershed system of interest according to the purpose of the analysis. This entails selection of a proper model and demonstration of sufficient correspondence between model outputs and the observed behavior of the system under study. Synthesis of the NIFA–CEAP watershed studies revealed apparent inconsistencies among modeling efforts. Therefore, adoption of a standard procedure for model application is highly recommended.

Following a modeling protocol serves multiple purposes, including (1) reducing potential modeler bias, (2) providing a roadmap to be followed, (3) allowing others to repeat the study, and (4) improving acceptance of model results. The model application process is iterative and

(c) SWCS. For Individual Use Only

should be conducted according to the overall modeling purpose. The *Guidance for Quality Assurance Project Plans for Modeling* report (USEPA 2002) describes the level of quality assurance required for different modeling purposes. Similar to monitoring, all modeling activities must be coordinated in a comprehensive quality assurance and quality control program that assures the credibility of modeling steps. Even if an official Quality Assurance Project Plans (QAPP) is not required for a project, preparation of such a plan is extremely useful as a roadmap that specifies how a modeling effort is conducted to meet intended goals.

Model application for conservation assessment and planning at the watershed scale should include the following steps:

- Define the modeling problem and application mode (prediction, exploratory analysis, knowledge-building, or collaborative learning). The modeling purpose defines the level of details and quality assurance required in the subsequent steps.
- Specify, unambiguously, modeling goals, objectives, and hypotheses. In the context of watershed management and conservation assessment, model application goals and objectives typically encompass only a subset of the overall project goals. The linkage between modeling goals and the overall project goals must be clearly stated. Defining and testing hypotheses using rigorous statistical approaches could establish required levels of confidence in the model-generated information and inferences. For example, one could test the hypothesis that the influence of interactions between a given set of land treatment options on total P loading is negligible at a 95% confidence level (Chaubey et al. 2010). Similarly, it could be hypothesized that implementation of upland erosion control measures could result in increased channel erosion. In general, the use of statistical hypothesis testing is more practical when models are used for exploratory and knowledge-building purposes.
- Collect watershed data and identify needs for future data collection. Data requirements can change according to the modeling goals and objectives. For example, when the goal of model application is to identify priority subwatersheds, streamflow and water quality data at multiple stream locations must be available. Also, when the goal of model application is to examine the environmental benefits of current conservation practices, information on their exact locations and characteristics are required. Climate data have been shown to play a critical role in accuracy of model predictions. Therefore, diligent efforts for collection of climate data from various sources are critical to proper characterization of watersheds in the models.
- Select a model(s) suitable for the study. The selected model(s) must include capacities to represent essential characteristics of the system and land treatment options at desired spatial and temporal scales. Additionally, selection of an appropriate model(s) must consider the availability of hydrologic and water quality data, along with watershed data, such as chemical usage and conservation practices. Evaluating the tradeoffs between model complexity, identifiability, and model performance could help identify an appropriate level of model complexity (figure 5.2). The use of multiple models, when feasible, is highly recommended.
- Set up the model. Several representation issues must be considered in model configuration and creation of model inputs. First, the number and characteristics of subwatersheds, stream segments, and hydrologic response units or computational cells must be determined according to the heterogeneity of watershed characteristics, such as slope, soils,

(c) SWCS. For Individual Use Only

land use, and climate (Tarboton 1997; FitzHugh and MacKay 2000; Jha et al. 2004; Arabi et al. 2006b). Soil, land use, and elevation characteristics should be represented at the coarsest spatial resolutions that capture their heterogeneities within the watershed (Chaubey et al. 2005).

An important representation issue for conservation assessment is availability of information on field boundaries or common land units. Ideally, model configuration permits mapping outputs to field boundaries where management decisions are made. Moreover, representation of first order streams and riparian buffers in the model may be critical for accurate representation of in-stream processes on water quality fluxes (Alexander et al. 2000 and 2007; Cho et al. 2010).

- Conduct sensitivity analysis. Sensitivity analysis can serve several purposes: (1) selection of important model parameters for calibration purposes, (2) examination of the sensitivity of model outputs to numerical representation of conservation practices (Arabi et al. 2008), and (3) identification of critical natural processes and key management actions that control hydrologic and water quality response of a watershed system (Arabi et al. 2007a). Appropriate local, screening, and/or global sensitivity analysis methods for each of these purposes must be applied (Saltelli et al. 2000; Tarantola et al. 2002; Arabi et al. 2007a).
- Calibrate and test the model. Watershed management typically involves modeling multiple hydrologic and water quality variables at multiple sites within the watershed. Calibration of a model for a single variable (e.g., streamflow) at the watershed outlet is typically insufficient for estimation of important hydrologic and water quality model parameters. Application of multiobjective approaches for parameter estimation is recommended to make best use of available data from the watershed. Also, general understanding of watershed processes, such as denitrification rates or nitrate loads from subsurface drainage, must be used to increase the confidence in calibration results.

  A diverse set of performance measures and error statistics in the calibration process must be evaluated to examine the model performance under different hydrologic and climatic conditions (Nash and Sutcliffe 1970; Gupta et al. 1998; Legates and McCabe 1999; Moriasi et al 2007). Approximately eight years of daily data that encompass average, wet, and dry years must be included so that a sufficient range of hydrologic events are captured during calibration (Yapo et al. 1996; Gan et al. 1997).
- Analyze uncertainties. Credibility of model-derived information depends greatly upon analysis and communication of uncertainties. Modeling studies, particularly when stakeholders are affected by their outcomes, must evaluate propagation of uncertainties from various sources (input, parameter, structure, and measurement) forward into model outcomes (Beven and Binley 1992; NRC 2001). When models are used in the predictive mode, predictions errors should be explicitly estimated. For example, quantifying environmental benefits of conservation practices should be reported within a confidence limit computed at a desired significance level (Arabi et al. 2007b). Several analytical and sampling-based methods have been suggested for computation of prediction uncertainty. In particular, Bayesian methods (e.g., Markov Chain Monte Carlo methods) have been gaining increasing popularity (Freer et al. 1996; Liu and Gupta 2007; Vrugt et al. 2009).

  On the other hand, uncertainties associated with exploratory application of watershed models can serve as sources of creativity. In these cases, uncertainties could reveal changes in the system behavior in response to unanticipated changes in system inputs. Uncertainties

(c) SWCS. For Individual Use Only

in the exploratory model application serve to build scenarios for making decisions under uncertain conditions, and therefore, do not need to be reduced. In the knowledge-building mode, analysis of model structure uncertainty reveals knowledge deficiencies, soundness of model structure, and additional values gained by embedding new algorithms. In these cases, analysis of uncertainty exposes tradeoffs between model complexity, performance, and identifiability (Snowling and Kramer 2001; Wagener et al. 2001).

- Interpret modeling outcomes, test hypotheses, and summarize results. Synthesis of modeling studies is best supported by pair-wise comparison of scenarios, hypothesis testing, and multivariate statistical evaluation of important factors. Modeling outputs should be generated according to the target audience.
- Document the modeling process, including decisions made throughout the effort. The required level of documentation varies with the model application. Documentation plans should address the content, format, and frequency of reports. This documentation should include analysis of the ability of reports to meet management expectations and to provide ongoing feedback for the project.

Organizations Supporting Model Development and Application

Organizations that support model development and application must abide by the following:

- Require an official QAPP for model application as supplementary information when disseminating requests for proposals
- Require peer review for approval of the modeling study
- Encourage inclusion of stakeholders in the model building and application process
- Encourage the use of models for collaborative learning purposes

The formal QAPP must require proposals to justify that model selection is appropriate and feasible (just like a water quality monitoring database). Finally, integrated research, education, and extension projects are well suited for promotion of stakeholder engagement in the model-building process. However, questions related to conservation assessment must be posed appropriately to promote adaptive community learning through the continual mutual feedback between scenario-generation and systematic analysis.

References

Alexander, R.B., E.W. Boyer, R.A. Smith, G.E. Schwarz, and R.B. Moore. 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resources Association 43:41-59.

Alexander, R.B., R.A. Smith, and G.E. Schwarz. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403:758-761.

Allen, P.M., J.G. Arnold, and W. Skipworth. 2002. Erodibility of urban bedrock and alluvial channels, North Texas. Journal of the American Water Resources Association 38(5):1477-1492.

Arabi, M., J.R. Frankenberger, B. Engel, and J.G. Arnold. 2008. Representation of agricultural management practices with SWAT. Hydrological Processes 22:3042-3055.

Arabi, M., R.S. Govindaraju, and M.M. Hantush. 2006a. Cost-effective allocation of watershed management practices using a genetic algorithm. Water Resources Research 42:W10429.

Arabi, M., R.S. Govindaraju, M.M. Hantush, and B. Engel. 2006b. Role of watershed subdivision on evaluation of long-term impact of best management practices on water quality. Journal of the American Water Resources Association 42(2):513-528.

(c) SWCS. For Individual Use Only

Arabi, M., R.S. Govindaraju, B. Engel, and M. Hantush. 2007a. Multiobjective sensitivity analysis of sediment and nutrient processes with a watershed model. Water Resources Research 43:W06409.

Arabi, M., R.S. Govindaraju, and M.M. Hantush. 2007b. A probabilistic approach for analysis of uncertainty in evaluation of watershed management practices. Journal of Hydrology 333:459-471.

Arnold, J.G., J.R. Kiniry, R. Srinivasan, J.R. Williams, E.B. Haney, and S.L. Neitsch. 2011. Soil and Water Assessment Tool Input/Output File Documentation Version 2009. College Station, TX: Texas A&M University Sytem.

Arnold, J.G., R. Srinivasan, R.S. Muttiah, and J.R. Williams. 1998. Large area hydrologic modelling and assessment part I: Model development. Journal of the American Water Resources Association 34(1):73–89.

Beck, M.B., J.R. Ravetz, L.A. Mulkey, and T.O. Barnwell. 1997. On the problem of model validation for predictive exposure assessments. Stochastic Hydrology and Hydraulics 11(3):229–254.

Behera, S., and R.K. Panda. 2006. Evaluation of management alternatives for an agricultural watershed in a sub-humid subtropical region using a physical process model. Agriculture, Ecosystems & Environment 113(1–4):62–72.

Bekele, E.G., and J.W. Nicklow. 2005. Multiobjective management of ecosystem services by integrative watershed modeling and evolutionary algorithms. Water Resources Research 41:W10406.

Beven, K.J. 1993. Prophecy, reality and uncertainty in distributed hydrological modelling. Advances in Water Resources 16:41–51.

Beven, K.J., and A.M. Binley. 1992. The future of distributed models: Model calibration and uncertainty prediction. Hydrological Processes 6:279–298.

Bicknell, B.R., J.C. Imhoff, J.L. Kittle, Jr., A.S. Donigian, Jr., and R.C. Johanson. 1996. Hydrological Simulation Program–FORTRAN, User's Manual for Release 11. Athens, GA: United States Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory.

Bingner, R.L., and F.D. Theurer. 2003. AnnAGNPS technical processes documentation, Version 3Ð2. Oxford, MS: USDA ARS National Sedimentation Laboratory.

Bishop, P.L., W.D. Hively, J.R. Stedinger, M.R. Rafferty, J.L. Lojpersberger, and J.A. Bloomfield. 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. Journal of Environmental Quality 34:1087–1101.

Borah D.K., and M. Bera. 2003. Watershed-scale hydrologic and nonpoint source pollution models: Review of mathematical bases. Transactions of the American Society of Agricultural Engineers 46(6):1553–1566.

Bracmort, K.S., M. Arabi, J.R. Frankenberger, B.A. Engel, and J.G. Arnold. 2006. Modeling long-term water quality impact of structural BMPs. Transactions of the American Society of Agricultural and Biological Engineers 49(2):367-374.

Brugnach, M., and C. Pahl-Wostl. 2007. A broadened view on the role for models in natural resource management: Implications for model development. *In* Adaptive and Integrated Water Management: Coping with Complexity and Uncertainty, ed. C. Pahl-Wostl, P. Kabat, and J. Moltgen, 184-203. New York: Springer Berlin Heidelberg.

Chaubey, I., L. Chiang, M.W. Gitau, and S. Mohamed. 2010. Effectiveness of best management practices in improving water quality in a pasture-dominated watershed. Journal of Soil and Water Conservation 65(6):424-437, doi:10.2489/jswc.65.6.424.

Chaubey, I., A.S. Cotter, and T.A. Costello. 2005. Effect of DEM data resolution on SWAT output uncertainty. Hydrological Processes 19(3):621-628.

Cho, J., G. Vellidis, D.D. Bosch, R. Lowrance, and T. Strickland. 2010. Water quality effects of simulated conservation practice scenarios in the Little River Experimental Watershed. Journal of Soil and Water Conservation 65(6):463-473, doi:10.2489/jswc.65.6.463.

Cohn, T.A., D.L. Caulder, E.J. Gilroy, L.D. Zynjuk, and R.M. Summers. 1992. The validity of a simple statistical model for estimating fluvial constituent loads: An empirical study involving nutrient loads in Chesapeake Bay. Water Resources Research 28(9):2353-2363.

(c) SWCS. For Individual Use Only

Easton, Z.M., D.R. Fuka, M.T. Walter, D.M. Cowan, E.M. Schneiderman, and T. Steenhuis. 2008a. Re-conceptualizing the Soil and Water Assessment Tool (SWAT) model to predict runoff from variable source areas. Journal of Hydrology 348:279– 291.

Easton, Z.M., M.T. Walter, and T.S. Steenhuis. 2008b. Combined monitoring and modeling indicate the most effective agricultural best management practices. Journal of Environmental Quality 37:1798-1809.

Eshleman, K.N., K.M. Kline, R.P. Morgan II, N.M. Castro, and T.L. Negley. 2008. Contemporary trends in the acid–base status of two acid-sensitive streams in western Maryland. Environmental Science and Technology 2008(42):56–61.

FitzHugh, T.W., and D.S. MacKay. 2000. Impacts of input parameter spatial aggregation on an agricultural nonpoint source pollution model. Journal of Hydrology 236(1-2):35-53.

Flanagan, D.C., J.E. Gilley, and T.G. Franti. 2007. Water Erosion Prediction Project (WEPP): Development, history, model capabilities, and future enhancements. Transactions of the American Society of Agricultural and Biological Engineers 50(5):1603-1612.

Freer, J., K.J. Beven, and B. Ambroise. 1996. Bayesian estimation of uncertainty in runoff prediction and the value of data: An application of the GLUE approach. Water Resources Research 32:2161-2173.

Gan, T.Y, E.M. Dlamini, and G.F. Biftu. 1997. Effects of model complexity and structure, data quality, and objective functions on hydrologic modeling. Journal of Hydrology 192(1):81-103.

Gassman, P.W., M.R. Reyes, C.H. Green, and J.G. Arnold. 2007. The Soil and Water Assessment Tool: Historical development, applications, and future research directions. Transactions of the American Society of Agricultural and Biological Engineers 50(4):1211–1250.

Ghidey, F., C. Baffaut, R.L. Lerch, N.R. Kitchen, E.J. Sadler, and K.A. Sudduth. 2010. Herbicide transport to surface runoff from a claypan soil: Scaling from plots to fields. Journal of Soil and Water Conservation 65(3):168-179, doi:10.2489/jswc.65.3.168.

Gibbons, M. 1999. Science's new social contract with society. Nature 402(Supplement 2):C81–C84.

Gitau, M.W., L.C. Chiang, M. Sayeed, and I. Chaubey. 2011. Watershed modeling using large-scale distributed computing in Condor and the Soil and Water Assessment Tool model. Simulation: Transactions of the Society for Modeling and Simulation International, doi:10.1177/0037549711402524.

Goolsby, D.A., and W.A. Battaglin. 2001. Long-term changes in concentrations and flux of nitrogen in the Mississippi River Basin, USA. Hydrological Processes 15:1209–1226.

Goolsby, D.A., W.A. Battaglin, B.T. Aulenbach, and H.P. Hooper. 2000. Nitrogen flux and sources in the Mississippi River Basin. Science of the Total Environment 248:75–86.

Gupta, H., S. Sorooshian, and P. Yapo. 1998. Toward improved calibration of hydrologic models: Multiple and noncommensurable measures of information. Water Resources Research 34(4):751-763.

Haith, D.A., R. Mandel, and R.S. Wu. 1992. GWLF, Generalized Watershed Loading Functions, Version 2.0, User's Manual. Ithaca, NY: Cornell University, Department of Agricultural and Biological Engineering.

Hanna, S.R. 1988. Air quality model evaluation and uncertainty. Journal of the Air Pollution Control Association 38:406-442.

Harbaugh, A.W. 2005. MODFLOW-2005, the U.S. Geological Survey modular ground-water model —the Ground-Water Flow Process: US Geological Survey Techniques and Methods 6-A16.

Harmel, R.D., R.J. Cooper, R.M. Slade, R.L. Haney, and J.G. Arnold. 2006. Cumulative uncertainty in measured streamflow and water quality data for small watersheds. Transactions of the American Society of Agricultural and Biological Engineers 49(3):689-701.

Harmel, R.D., C.G. Rossi, T. Dybala, J.G. Arnold, K. Potter, J. Wolfe, and D. Hoffman. 2008. Conservation Effects Assessment Project research in the Leon River and Riesel watersheds. Journal of Soil and Water Conservation 63(6):453-460, doi:10.2489/jswc.63.6.453.

Harmel, R.D., and P.K. Smith. 2007. Consideration of measurement uncertainty in the evaluation of goodness-of-fit in hydrologic and water quality modeling. Journal of Hydrology 337:326-336.

(c) SWCS. For Individual Use Only

Heathman, G.C., D.C. Flanagan, M. Larose, and B.W. Zuercher. 2008. Application of SWAT and AnnAGNPS in the St. Joseph River Watershed. Journal of Soil and Water Conservation 63(6):552–568, doi:10.2489/jswc.63.6.552.

Hooper, R.P., B.T. Aulenbach, and V.J. Kelly. 2001. The national stream quality accounting network: A flux-based approach to monitoring the water quality of large rivers. Hydrological Processes 15:1089-1106.

Jha, M., P.W. Gassman, S. Secchi, R. Gu, and J. Arnold. 2004. Effect of watershed subdivision on SWAT flow, sediment, and nutrient predictions. Journal of the American Water Resources Association 40(3):811-825.

Johnes, P.J. 2007. Uncertainties in annual riverine phosphorus load estimation: Impact of load estimation methodology, sampling frequency, baseflow index and catchment population density. Journal of Hydrology 332:241-258.

Konikow, L.F., and J.D. Bredehoeft. 1992. Ground-water models cannot be validated. Advances in Water Resources 15:75–83.

Korfmacher, K.S. 2001. The politics of participation in watershed modeling. Environmental Management 27(2):161–176.

Kuhnle, R.A., R.L. Bingner, C.V. Alonso, C.G. Wilson, and A. Simon. 2008. Conservation practice effects on sediment load in the Goodwin Creek Experimental Watershed. Journal of Soil and Water Conservation 63(6):496-503, doi:10.2489/jswc.63.6.496.

Lam, Q.D., B. Schmatz, and N. Fohrer. 2011. The impact of agricultural best management practices on water quality in a north German lowland catchment. Environmental Monitoring Assessment 183:351-379.

Lee, K.Y., T.R. Fisher, T.E. Jordan, D.L. Correll, and D.E. Weller. 2000. Modeling the hydrochemistry of the Choptank River Basin using GWLF and Arc/Info: I. Model calibration and validation. Biogeochemistry 49(2):143–173.

Legates, D.R., and G.J. McCabe. 1999. Evaluating the use of "goodness-of-fit" measures in hydrologic and hydroclimatic model validation. Water Resources Research 35(1):233-241.

Licciardello, F., D.A. Zema, S.M. Zimbone, and R.L. Bingner. 2007. Runoff and soil erosion evaluation by the AnnAGNPS model in a small Mediterranean watershed. Transactions American Society Agricultural & Biological Engineers 50(5):1585–1593.

Liu, Y., and H. Gupta. 2007. Uncertainty in hydrologic modeling: Toward an integrated data assimilation framework. Water Resources Research 43:W07401.

Maret, T.R., D.E. MacCoy, and D.M. Carlisle. 2008. Long-term water quality and biological responses to multiple best management practices in Rock Creek, Idaho. Journal of the American Water Resources Association 44(5):1248–1269.

Moatar, F., and M. Meybeck. 2005. Compared performance of different algorithms for estimating annual nutrient loads discharged by the eutrophic River Loire. Hydrological Processes 19:429-444.

Moriasi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Bingner, R.D. Harmel, and T.L. Veith. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Transactions of the American Society of Agricultural and Biological Engineers 50(3):885–900.

Mostaghimi, S., S.W. Park, R.A. Cooke, and Y. Wang. 1997. Assessment of management alternatives on a small agricultural watershed. Water Research 31(8):1867-1878.

Nash, J.E., and J.V. Sutcliffe. 1970. River flow forecasting through conceptual models: Part 1. A discussion of principles. Journal of Hydrology 10(3):282-290.

Newson, J. 2007. Measurement and Modeling of Sediment Transport in a Northern Idaho Stream. Masters thesis, Biological and Agricultural Engineering, College of Graduate Studies, University of Idaho, Moscow, ID.

NRC (National Research Council). 2001. Assessing the TMDL approach to water quality management. Washington, DC: National Academy Press.

(c) SWCS. For Individual Use Only

Pandey, A., V.M. Chowdary, B.C. Mal, and M. Billib. 2009. Application of the WEPP model for prioritization and evaluation of best management practices in an Indian watershed. Hydrological Processes 23:2997-3005.

Parajuli, P.B., N.O. Nelson, L.D. Frees, and K.R. Mankin. 2009. Comparison of AnnAGNPS and SWAT model simulation results in USDA-CEAP agricultural watersheds in south-central Kansas. Hydrological Processes 23:748-763.

Polyakov, A., A. Fares, D. Kubo, J. Jacob, and C. Smith. 2007. Evaluation of a non-point source pollution model, AnnAGNPS, in a tropical watershed. Environmental Modeling and Software 22(11):1617-1627.

Rabotyagov, S.S., T. Campbell, M. Jha, P.W. Gassman, J.G. Arnold, L. Kurkalova, S. Secchi, H. Feng, and C.L. Kling. 2010a. Least-cost control of agricultural nutrient contributions to the Gulf of Mexico hypoxic zone. Ecological Applications 20:1542–1555.

Rabotyagov, S.S., M. Jha, and T.D. Campbell. 2010b. Nonpoint-source pollution reduction for an Iowa watershed: An application of evolutionary algorithms. Canadian Journal of Agricultural Economics 58(4):411–431.

Renscheler, C.S., and T. Lee. 2005. Spatially distributed assessmen tof short- and long-term impacts of multiple best management practices in agricultural watersheds. Journal of Soil and Water Conservation 60(6):446-456.

Richardson, C.W., D.A. Bucks, and E.J. Sadler. 2008. The Conservation Effects Assessment Project benchmark watersheds: Synthesis of preliminary findings. Journal of Soil and Water Conservation 63(6):590-604, doi:10.2489/jswc.63.6.590.

Robertson, D.M., and E.D. Roerish. 1999. Influence of various water quality sampling strategies on load estimates for small streams. Water Resources Research 35:3747-3759.

Rodriguez, H.G., J. Popp, C. Maringanti, and I. Chaubey. 2011. Selection and placement of best management practices used to reduce water quality degradation in Lincoln Lake watershed. Water Resources Research 47:W01507.

Rykiel, E.J. 1996. Testing ecological models: The meaning of validation. Ecological Modelling, 90:229–244.

Sadeghi, A., Y. Kwang, C. Graff, G. McCarty, L. McConnell, A. Shirmohammadi, D. Hively, and K.A. Sefton. 2007. Assessing the Performance of SWAT and AnnAGNPS Models in a Coastal Plain Watershed, Choptank River, Maryland, U.S.A. American Society of Agricultural and Biological Engineers Meeting, Paper No. 072 032. St. Joseph, MI: American Society of Agricultural and Biological Engineers.

Saleh, A., and B. Du. 2004. Evaluation of SWAT and HSPF within BASINS program for the upper North Bosque River Watershed in central Texas. Transactions of the American Society of Agricultural Engineers 47(4):1039-1049.

Saltelli, A., K. Chan, and E.M. Scott. 2000. Sensitivity Analysis. New York: John Wiley & Sons.

Santhi, C., J.G. Arnold, J.R. Williams, L.M. Hauck, and W.A. Dugas. 2001. Application of a watershed model to evaluate management effects on point and non-point source pollution. Transactions of the American Society of Agricultural Engineers 44(6):1559–1570.

Schneiderman, E.M., D.C. Pierson, D.G. Lounsbury, and M.S. Zion. 2002. Modeling the hydrochemistry of the Cannonsville Watershed with Generalized Watershed Loading Functions (GWLF). Journal of the American Water Resources Association 38(5):1323–1347.

Schneiderman, E.M., T.S. Steenhuis, D.J. Thongs, Z.M. Easton, M.S. Zion, G.F. Mendoza, M.T. Walter, and A.L. Neal. 2007. Incorporating variable source area hydrology into the curve number based Generalized Watershed Loading Function model. Hydrological Processes 21:3420–3430.

Shoemaker, L., T. Sai, and J. Koenig. 2005. TMDL Model Evaluation and Research Needs. Report EPA/600/R-05/149, National Risk Management Research Laboratory, Office of Research and Development, Cincinnati, OH: United States Environmental Protection Agency.

Snowling, S.D., and J.R. Kramer. 2001. Evaluating modelling uncertainty for model selection. Ecological Modelling 138:17-30.

(c) SWCS. For Individual Use Only

Spear, R.C., T.M. Grieb, and N. Shang. 1994. Parameter uncertainty and interaction in complex environmental models. Water Resources Research 30(11):3159-3169.

Srivastava, P., J.M. Hamlett, P.D. Robillard, and R.L. Day. 2002. Watershed optimization of best management practices using AnnAGNPS and a genetic algorithm. Water Resources Research 38(3):1–14.

Stenback, G.S., W.G. Crumpton, K.E. Schilling, and M.J. Helmers. 2011. Rating curve estimation of nutrient loads in Iowa rivers. Journal of Hydrology 396:158-169.

Tarantola, S., N. Giglioli, J. Jesinghaus, and A. Saltelli. 2002. Can global sensitivity analysis steer the implementation of models for environmental assessments and decision-making? Stochastic Environmental Research and Risk Assessment 16(1):63-76.

Tarboton, D.G. 1997. A new method for the determination of flow directions and contributing areas in grid digital elevation models. Water Resources Research 33(2):309-319.

Tolson, B., and C. Shoemaker. 2007. Dynamically dimensioned search algorithm for computationally efficient watershed model calibration. Water Resources Research 43:W01413.

Tomer, M.D., and M.A. Locke. 2011. The challenge of documenting water quality benefits of conservation practices: A review of USDA-ARS's conservation effects assessment project watershed studies. Water Science & Technology 64(1):300–310.

Tuppad, P.C. Santhi, X. Wang, J.R. Williams, R. Srinivasan, and P.H. Gowda. 2010. Simulation of conservation practices using the APEX model. Applied Engineering in Agriculture 26(5):779-794.

Ullrich, A., and M. Volk. 2010. Influence of different nitrate-N monitoring strategies on load estimation as a base for model calibration and evaluation. Environmental Monitoring and Assessment 171(14):512-527.

USDA CSREES (Cooperative State Research, Education, and Extension Service). 2004. Evaluating the Effects of Conservation Practices on Water Quality within the Biophysical Setting of a Watershed. Fiscal Year 2004 Supplemental Request for Applications, Integrated Research, Education, and Extension Competitive Grants Program–National Integrated Water Quality Program. Washington, DC: USDA Cooperative State Research, Education, and Extension Service.

USDA CSREES. 2005. Conservation Effects Assessment Project (CEAP): Evaluating the Effects of Conservation Practices on Water Quality within the Biophysical Setting of a Watershed. FY 2005 Supplemental Request for Applications, Integrated Research, Education, and Extension Competitive Grants Program–National Integrated Water Quality Program, Washington, DC: USDA Cooperative State Research, Education, and Extension Service.

USDA CSREES. 2006. Conservation Effects Assessment Project (CEAP): Evaluating the Effects of Conservation Practices on Water Quality within the Biophysical Setting of a Watershed. FY 2006 Supplemental Request for Applications, Integrated Research, Education, and Extension Competitive Grants Program–National Integrated Water Quality Program, Washington, DC: USDA Cooperative State Research, Education, and Extension Service.

USDA NRCS (Natural Resources Conservation Service). 2010. Assessment of the Effects of Conservation Practices on Cultivated Cropland in the Upper Mississippi River Basin. Final Report. http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/ceap/na/?&cid=nrcs143_014161.

USDA NRCS. 2011a. Assessment of the Effects of Conservation Practices on Cultivated Cropland in the Chesapeake Bay Region. Final Report. http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/ceap/na/?&cid=stelprdb1041684.

USDA NRCS. 2011b. Assessment of the Effects of Conservation Practices on Cultivated Cropland in the Great Lakes Region. Final Report. http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/ceap/na/?&cid=stelprdb1045403.

USEPA (United States Environmental Protection Agency). 2002. Guidance for Quality Assurance Project Plans for Modeling. EPA QA/G-5M. Washington, DC: United States Environmental Protection Agency, Office of Environmental Information.

(c) SWCS. For Individual Use Only

USGS (United States Geological Survey). 2004. Load Estimator (LOADEST): A Fortran Program for Estimating Constituent Loads in Streams and Rivers. Techniques and Models Book 4, Chapter 5. Reston, VA: United States Geological Survey.

Veith, T.L., M.W. Van Liew, D.D. Bosch, and J.G. Arnold. 2010. Parameter sensitivity and uncertainty in SWAT: A comparison across five USDA-ARS watersheds. Transactions of the American Society of Agricultural and Biological Engineers 53(5):1477–1486.

Van Liew, M.W., T.L. Veith, D.D. Bosch, and J.G. Arnold. 2007. Suitability of SWAT for the Conservation Effects Assessment Project: Comparison on USDA Agricultural Research Service watersheds. Journal of Hydrologic Engineering 12(2):173-189.

Veith, T.L., M. L. Wolfe, and C.D. Heatwole. 2003. Optimization procedure for cost effective BMP placement at a watershed scale. Journal of the American Water Resources Association 39:1331–1343.

Vrugt, J.A., C.J.F. ter Braak, H.V. Gupta, and B.A. Robinson. 2009. Equifinality of formal (DREAM) and informal (GLUE) Bayesian approaches in hydrologic modeling. Stochastic Environmental Research and Risk Assessment 23(7):1011-1026.

Yapo, P., H.V. Gupta, and S. Sorooshian. 1996. Calibration of conceptual rainfall-runoff models: Sensitivity to calibration data. Journal of Hydrology 181:23–48.

Yuan, Y., S.M., Dabney, and R.L. Bingner. 2002. Cost effectiveness of agricultural BMPs for sediment reduction in the Mississippi Delta. Journal of Soil and Water Conservation 57(5):259 -267.

Yuan, Y., M.A. Locke, and R.L. Bingner. 2008. Annualized agricultural non-point source pollution model application for Mississippi Delta Beasley Lake Watershed conservation practices assessment. Journal of Soil and Water Conservation 63(6):542–551, doi:10.2489/jswc.63.6.542.

Wagener, T., D.P. Boyle, M.J. Lees, H.S. Wheater, H.V. Gupta, and S. Sorooshian. 2001. A framework for development and application of hydrological models. Hydrology and Earth System Sciences 5:13-26.

White, M.J., and J.G. Arnold. 2009. Development of a simplistic vegetative filter strip model for sediment and nutrient retention at the field scale. Hydrological Processes 23(11):1602-1616.

Whittaker, G., R. Färe, R. Srinivasan, and D.W. Scott. 2003. Spatial evaluation of alternative nonpoint nutrient regulatory instruments. Water Resources Research 39(4):1079.

Williams, J.R., and R.C. Izaurralde. 2006. The APEX model. *In* Watershed Models, ed. V.P. Singh and D.K. Frevert, 437-482. Boca Raton, FL: CRC Press, Taylor and Francis Group.

Zhou, X., M.J. Helmers, M. Al-Kaisi, and H.M. Hanna. 2009. Cost-effectiveness and cost-benefit analysis of conservation management practices for sediment reduction in an Iowa agricultural watershed. Journal of Soil and Water Conservation 64(5):314-323, doi:10.2489/jswc.64.5.314.

(c) SWCS. For Individual Use Only