

Refining Models for Quantifying the Water Quality Benefits of Improved Animal Management for Use in Water Quality Trading

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SUPPLEMENTAL PAPERS

Management Practices to Improve Water Quality on Central and Western Rangelands

Assessing Potential Impacts of Livestock Management on Groundwater

Management Options for Animal Operations to Reduce Nutrient Loads

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EXECUTIVE SUMMARY

Water quality trading (WQT) allows point-source permittees to meet their water quality obligations by purchasing credits from other point or nonpoint sources that have reduced their discharges. Non-permitted animal operations and permitted animal operations could be attractive partners in such a WQT program. To incorporate animal operations into water quality trading, methods for quantifying pollutant reductions resulting from changes in management practices must be sufficiently accurate or conservative so that regulators and point-source purchasers can be confident in the results. Already under way in many places, water quality trading typically uses computer models or empirical data synthesis to estimate the water quality benefits of management practices. Any uncertainties are usually addressed by adjusting the trading ratio to result in a more conservative estimate of the amount of water quality benefit required to offset water quality impacts. This report provides an overview of existing measurement and modeling methods and tools to inform efforts to build updated and more integrated methods for quantifying water quality benefits of animal operation management for use in WQT programs. This is an academic review of models and methods, not a guide to how these tools can be adapted for use in water quality trading programs. For a more practical exploration of model application see Electric Power Research Institute (2011) evaluation of the Nutrient Trading Tool and Watershed Analysis and Risk Management Framework.

The first section of this report provides an overview of how the Clean Water Act underlies water quality trading programs, how animal operations fit in, and how water quality trading works. It also reviews what needs to be quantified for different types of management practices in order to determine changes in pollutant loading and how improved quantification can help support water quality trading programs.

The second section provides an overview of models used to estimate various aspects of animal management-water quality relationships. The discussion starts with models of animal production of nutrient waste, reviews models for surface water transport, and then covers hydrological, empirical, and mechanistic process-based models used to assess transport and transformation of pollutants in watersheds. The section then describes the potential for quantifying unexpected effects of nutrient transformation and transport by considering losses to groundwater and the atmosphere. This section also summarizes how quantification may be different for rangeland and grazing land systems as compared to confined feeding operations.

The third section of the report describes how direct measurement and monitoring of nutrient losses is evolving with new technologies and how it can be used to improve the quantification and modeling of nutrients over time, to measure cumulative change in a water body, and perhaps eventually how it could be used to quantify edge-of-field losses.

The fourth section of the report recommends how to update and refine existing models and tools to reduce uncertainties in water quality quantification, how to make sure they are viable for use by practitioners, and how they should be built to improve over time.

Ultimately, water quality trading requires quantification of the nutrient loading reductions associated with management practices and some estimate of the uncertainty associated with this quantification. This process can be straightforward and robust for some practices, such as reducing nutrients imported to an animal operation or wastes exported from a watershed, but there is much less confidence in quantification of other practices that rely on chemical or microbial processes to reduce pollutant transport, for example, microbial denitrification to reduce nitrate loading to streams. Because water quality trading carries the logistical and economic implications of changing animal production practices as well as trading pollution credits, reliable estimates of the load reduction from new practices on water quality are important. When they are lacking, animal system managers bear the burden of higher trading ratios, which lowers the number of credits they receive for their estimated load reduction. Ultimately, targeted and coordinated

investment in developing, improving, and integrating selected animal production and hydrologic models at both the farm and watershed scales will be needed to improve quantification.

This report makes several key points and recommendations:

- The computational foundation of many water quality models has not been updated for 20 or more years.
- The federal government could provide support to state WQT programs by using a rigorous external and academic review to select the best empirical and process-based models and to focus resources on improving and adapting these models to address the needs of WQT programs.
- Models should be developed in linkable modules and updated as technologies become available. They should integrate spatial methods and creating opportunities for user-friendly interfaces.
- Given the uncertainties in groundwater and emissions modeling, modeling outcomes may be best used to indicate potential risks and areas for additional assessment. Effort should be directed toward incorporating methods or linking models to estimate the likelihood that certain management practices would increase groundwater contamination or atmospheric emissions.
- WQT programs need tools to create simple and defensible crediting calculations.

ANIMAL AGRICULTURE: WATER QUALITY IMPACTS, WATER QUALITY TRADING, AND QUANTIFICATION METHODS

How Do Animal Operations Affect Water Quality?

Animal feeding operations (AFOs) with high animal densities are often concentrated sources of animal waste and associated nutrients (Negahban, Fonyo, Boggess, and Jones 1993; Boggess, Johns, and Meline 1997). These nutrients can migrate offsite through hydrologic pathways and have negative environmental effects on local surface waters. However, removing animal agriculture from a watershed is not a viable option for reducing pollutant loading. Animals provide a major share of food for human consumption, and animal production is socially and politically important throughout the United States (Steinfeld et al. 2006). Animal agriculture can also play an instrumental role in supporting sustainable grazing land, preserving wildlife and other forms of biodiversity, and enhancing soil fertility and nutrient cycling (Mearns 1996).

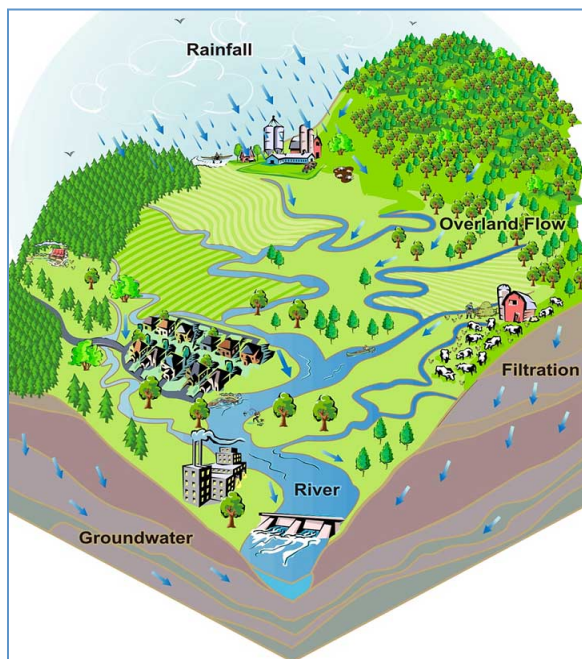
Water pollutants that may originate from animal production systems include nutrients such as nitrogen (N) and phosphorus (P), sediment, and organic matter; emerging contaminants of concern include pathogens, antibiotics, and hormones (US GAO 2008).¹

In 1996, states reported to the Environmental Protection Agency (EPA) that animal feeding operations were a contributing source in 10% of impaired rivers and streams. A U.S. Geological Survey (USGS) study of 16 watersheds found that manure was the largest source of N loadings at six of these sites, primarily in the southeast and mid-Atlantic states (Puckett 1994). USGS modeling of total N and P export from watersheds in the country's major water resource regions found that, nationally, the median contribution of N from animal agricultural sources was 14%, compared with 22% for commercial fertilizer and 0.8% for point sources (sewage treatment plants, factories). For P, the median contribution

¹ Animal operations can influence water quality parameters beyond nutrients (Burkholder et al. 2007; Ellison et al. 2009), including salinity, oxygen concentrations, and turbidity from soil erosion (Agouridis et al. 2005). Other pollutants from animal operations include pharmaceutical compounds such as hormones and antibiotics, heavy metals, pesticides, and endocrine disruptors (USEPA-OW 2013). These additional water quality effects are often influenced by the same characteristics of the operations that influence nutrients, but can also depend on other independent and pollutant-specific factors.

from animal agricultural sources was 26%, compared to 17% for commercial fertilizer and 3% for point sources (Smith and Alexander 2000). The USGS National Water Quality Assessment Program found that the highest concentrations of N in streams occurred in agricultural basins, and were correlated with N inputs from fertilizers and manure (USGS NWQAP 1999). An analysis of fecal coliform bacteria in streams found that concentrations were partly a function of the number of both confined and unconfined animals in a watershed. In the Chesapeake Bay watershed, which is impaired by nutrients, animal operations are estimated to contribute 17% of the N entering the bay and 26% of the P (NAS 2011).

Figure 1. How Animal Operations Affect Water.



Source: Gary Austin, Department of Landscape Architecture, University of Idaho.

Nutrients may come from a variety of activities, including leaching from manure storage and processing facilities, application of waste to fields, and deposition of manure in pastures, barnyards, and feedlots (Figure 1). Nitrogen is most often transported as dissolved nitrate (NO_3), which is highly mobile, but also as ammonium (NH_4) and dissolved organic N (DON) (Di and Cameron 2002; Driscoll et al. 2003). NO_3 can affect aquatic ecosystems (Galloway et al. 2003), but also has direct effects on wildlife and human health (Burkholder et al. 2007). Phosphorus can be transported in both organic and inorganic forms, and in dissolved forms or bound to sediments. Because P binds tightly to soils and sediments, erosion is a major transport pathway. However, dissolved P is a concern because it is more bioavailable in aquatic systems (Hansen, Daniel, Sharpley, and Lemunyon 2002; Sharpley, Kleinman, and Weld 2004).

Nitrogen and P can also become elevated in groundwater due to leaching through soil profiles and into aquifers (Heathwaite and Dils 2000). Groundwater N and P contamination can have negative consequences for water supplies and their suitability for human consumption and use (Townsend, Howarth, and Bazzaz 2003); it also contributes to surface water contamination over longer time frames as groundwater re-enters surface drainage networks (Scanlon et al. 2005; Burow, Nolan, Rupert, and Dubrovsky 2010; Meals, Dressing, and Davenport 2010; Wick, Heumesser, and Schmid 2012). Nitrogen can also be transformed into gaseous forms such as harmless N_2 gas or nitrogen oxides that contribute to air pollution, acid rain, climate change, and ozone destruction (Ravishankara 2009). Manure management

in the United States contributed 5% of the country's nitrous oxide emissions, and crop systems that produce animal feed as well as human food and bio-based products account for almost 70% (USEPA Nitrous Oxide Emissions).

Clean Water Act, Animal Operations, and Water Quality Trading

The Clean Water Act (CWA) is the primary federal statute addressing surface water quality. It is broad in scope and includes industrial, municipal, and agricultural point sources. Point sources that discharge through "discrete conveyances" such as pipes or ditches, are regulated through National Pollutant Discharge Elimination System (NPDES) permits (73 *Federal Register* 225; 2008). The CWA has a limited ability to address nonpoint source pollution and exempts nonpoint sources from the permit program, limiting the CWA's ability to address nonpoint sources (Clean Water Act 1977). An important provision of the CWA is the Total Daily Maximum Load (TMDL). The TMDL provision program is designed to restore impaired waters through NPDES permits and other limitations on pollutants. Section 303(d) of the CWA requires states to report impaired waters to USEPA and to develop TMDLs. A TMDL is the maximum amount of a pollutant that can be discharged to waters without violating water quality standards. A TMDL must identify all point and nonpoint pollutant sources, and allocate them a pollutant load. States then use the allocations to develop an implementation plan to reduce the pollutant load and prevent water quality violations. In many watersheds, point source controls are insufficient to meet the TMDL goals because nonpoint sources are the more significant pollutant load. Nonpoint source pollutant loads can be included in a TMDL, but their enforcement is left to the states. WQT is one tool that can be used to help states and point source dischargers meet their water quality objectives under a TMDL.

State of WQT Programs

A 2008 survey identified 57 WQT experiments, pilot programs, and demonstration projects in various stages of implementation around the world (Greenhalgh and Selman 2012; Selman et al. 2009). Of these, 26 had established trading rules and the opportunity for trading to occur, 21 were under development, and 10 were complete or inactive. All but six were in the United States. Despite considerable interest, point-nonpoint trading has not been very successful (Breetz et al. 2004, 2005; Hoag and Hughes-Popp 1997; Newburn and Woodward 2012; Ribaldo and Gottlieb 2011; Shabman and Stephenson 2002, 2011; King and Kuch 2005).

Some of the largest confined animal feeding operations (CAFOs) are considered point sources under the CWA and must have a National Pollutant Discharge Elimination System (NPDES) permit. In particular cases, determined at the state level, smaller facilities can also be identified as point sources. The USEPA estimates that there are approximately 15,300 animal operations that may be subject to permit requirements, out of a total of more than 900,000 (USDA-NASS 2009).

Permits for CAFOs are specifically aimed at the production area, where animal are housed or contained, and require zero direct discharge. However, this is not true for areas outside the production area, particularly cropland receiving manure. Such cropland can be an important source of non-point nutrient pollution (Ribaldo and Gottlieb 2011; Ribaldo et al. 2003). Because nonpoint sources do not flow through a discrete conveyance, the pollution is relatively difficult to identify and control and typically requires multiple management efforts across the landscape.

The management of manure and storm water on non-permitted (nonpoint source) AFOs and NPDES permitted (point source) CAFOs may be quite similar. Federal rules require both to identify site-specific conservation practices to control storm water, to use site-specific nutrient management practices, and to

maintain consistent testing protocols and record keeping.² Both types of facilities will have some continued discharge from cropland they manage despite proper nutrient management plans.

Water Quality Trading

Water quality trading (WQT) is a market-based approach to achieve water quality objectives at a reduced cost. It allows firms with different pollution abatement costs to allocate pollution abatement among themselves in a more efficient way.

Water quality trading is a type of cap and trade program. In a cap and trade program, trading is organized around the creation of good called a discharge allowance, which is a time-limited permission to discharge a fixed unit of pollutant into the environment. The regulatory agency first determines the maximum amount of discharge of a particular pollutant that a water body (stream, lake, or estuary) can absorb and

still meet environmental quality goals (the cap). Discharge allowances equal to the cap are then allocated to all regulated dischargers through an auction or some other means. By allowing discharge allowances to be traded, a market is created that can allocate abatement efficiently among regulated sources. WQT programs require: (1) heterogeneity in marginal abatement costs across pollution sources, (2) sufficient opportunity for operators to make improvements beyond regulatory requirements, and (3) quantification methods (measurement or models) to estimate changes in pollutant loads resulting from changes in management practices.

USEPA policy on water quality trading (USEPA-OW 2003) allows regulated point sources to meet their discharge requirements by purchasing reductions from unregulated point or nonpoint sources (e.g., sources that do not need a discharge permit can voluntarily participate in trading if they meet the program requirements).³ Some 23 active WQT projects allow trading between point and nonpoint sources (Greenhalgh and Selman 2012). Benefits of allowing nonpoint sources to be a source of credits include: 1) Abatement of nonpoint-source

Baselines

All point/nonpoint trading programs must establish an eligibility baseline for nonpoint sources. Fields not meeting the baseline criteria cannot produce credits for sale in a market until the baseline criteria are achieved. EPA's trading policy states that where a TMDL is in place, the load allocation should serve as the baseline for nonpoint sources to generate credits (U.S. EPA 2007). Nonpoint sources would be expected to meet their TMDL load allocation before being able to sell credits. Such a baseline implies a higher level of management than current practices. For example, Virginia's trading program requires that a field be treated with five management practices before being able to sell credits (nutrient management, soil erosion controls, cover crops, fencing (for livestock), and vegetative buffers). Very few fields meet this level of management. This baseline only applies to nonpoint sources wishing to enter a trading program. The ability to sell credits is implicitly expected to be a strong enough incentive for farmers to be willing to meet the baseline in order to trade. Where a TMDL is not established, the state can select a baseline that represents current practices.

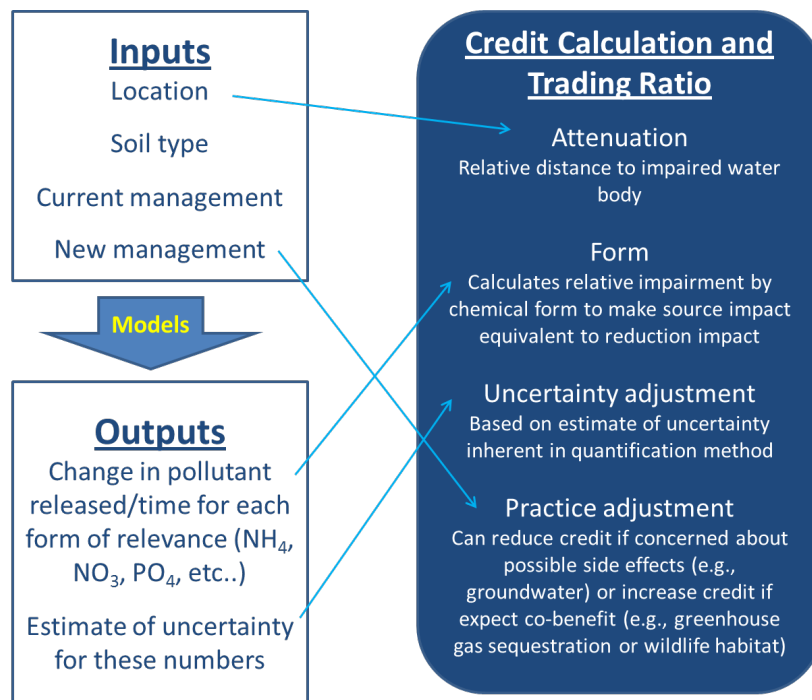
² NPDES-permitted facilities also must ensure adequate waste storage, properly manage mortalities, make sure that clean water is diverted from the production area, prevent direct contact between animals and waters, and guarantee that chemicals and pollutants are not disposed of in waste treatment systems unless those facilities are designed to treat them.

³ When animal operations are legally defined as CAFOs and subject to NPDES permit requirements, the production area of the CAFO is treated as a point source, which requires zero direct discharge for all but the largest storm events, assuming base facility design criteria are met. However, this is not true for areas outside the production area, particularly cropland receiving manure. A CAFO might find additional pollutant reductions from the land application area above and beyond that achieved through implementation of the facility nutrient management plan required for CAFOs. Additional reductions could, in theory, be used to generate credits for point-to-point source trades in a WQT program.

pollution is often much less expensive than abating point source pollution; 2) Greater communication and cooperation between point and non-point sources may help achieve a common goal; 3) Unregulated nonpoint sources play a larger role in water pollution control (Faeth 2000; Fang, Easter, and Brezonik 2005; Selman et al. 2009; Van Houtven et al. 2012; Shortle 2013); and allow new or expanded point sources into a capped watershed.

Because nonpoint source discharges are more difficult to observe than point discharges, credits are usually calculated at the field level using a modeling tool such as the Maryland Nutrient Trading Tool (Maryland Department of Agriculture 2013). The farmer gives the location, soil and field characteristics, current or baseline management practices (including information such as the size of the area or number of animals affected), and which practices he intends to implement. The models estimate changes in nutrient loads expected and this information is then used to calculate the number of credits that would be produced (Figure 2).

Figure 2. What Needs to Be Quantified to Calculate Water Quality Trading Credits?



Uncertainty

A major issue in the development and implementation of a point/nonpoint trading program is how uncertainty is addressed. Equivalency of credits is an important concept in trading programs; ideally, the discharge reductions a point source purchases in a market have the same impact on water quality as if the NPDES permittee reduced its own discharge. This assures that water quality goals are actually met. Establishing equivalency between point and nonpoint source discharges must account for two factors; agricultural practice effectiveness and location relative to the point source and/or the surface water being protected.

Translating BMP practice implementation into load estimates is subject to four types of uncertainty:

- **Management uncertainty**—Models and estimates may not be able to account for variability in how well a BMP performs over time due to differences in management skill and effort on the part of farmers. The performance of some practices, such as nutrient management, cannot be assessed even with on-site inspections.
- **Scientific uncertainty (on BMP removal effectiveness)**—There is incomplete information and knowledge about the pollutant losses at the field and farm level for management practices. Quantifying and measuring pollutant losses are complicated by site-specific circumstances: complex physical, biological, and chemical causal relationships that govern pollutant movement and transformations in soil, water, and the atmosphere. Models are developed based on laboratory and field level studies, but such studies often use different and incomplete field measurements to quantify and track nutrient fluxes. These uncertainties are readily observed and acknowledged in the literature.
- **Modeling uncertainty**—Models synthesize known scientific relationships into equations that estimate loads from different, assumed individual behavioral and practice changes. Every model is a simplification of underlying processes that give rise to agricultural nutrient losses, which generates error. Models can quantify reductions at the edge of a field relatively well. The problem comes in scaling this up and translating the implications of edge-of-field reductions to a watershed scale or the scale of discharge. In a review of intensive watershed studies of agricultural BMP implementation, Osmond et al. (2012) found that, in general, the “complexity and nonlinear nature of watershed processes overwhelm the capacity of existing modeling tools to reveal water quality impacts of conservation practices.” Osmond et al. (2012) also report that “models grossly overestimated the effectiveness of conservation practices.”
- **Weather stochasticity**—Nonpoint source pollution is driven by weather. Variations in weather, particularly rainfall, influence the amount of pollutants actually leaving a field, as well as how well a BMP performs. Some BMPs perform better than others when runoff is high. Weather variability is a major source of uncertainty in annual predictions of how many abatement credits a BMP produces.

Under the Clean Water Act a regulated point source cannot transfer its legal liability under the permit (Selman et al. 2009). Thus, a credit generator or a third-party aggregator cannot and does not take on the NPDES permit liability. If a regulated point source is legally responsible for achieving a particular discharge goal, the uncertainty about credits generated by nonpoint sources and the risk of an enforcement action may make them a less attractive option. A point source’s control strategy is generally a long-term decision, and it may be unwilling to rely on an uncertain source of credits because of the decision’s inherent irreversibility (McCann 1996). These factors may push point sources toward providing their own internal discharge controls or trading with other point sources, rather than relying on nonpoint credits. Measurement problems have been cited as obstacles in several trading programs (Breetz et al. 2004).

One way that trading programs address uncertainty is through a trading ratio. There are ratios to adjust for each of these four types of uncertainty, and together these make up a trading ratio (Figure 2). Trading ratios in water quality programs generally range from 2:1 to 5:1 (CTIC 2006). This means that a point source would have to purchase up to 5 units of pollutant reduction from a nonpoint source to ensure that a single unit of its discharge is “covered.” While providing assurance that the nonpoint source reduction provides the expected gain in water quality, a trading ratio increases the effective price of nonpoint credits, thereby reducing demand from point sources if they can find cheaper reductions elsewhere.

Improving quantification and reducing uncertainty about the performance of management practices would reduce some of the liability concerns of the point source purchasers and reduce the trading ratio, thereby reducing the effective price of the nonpoint reductions.

How Changes in Pollutant Loads Can Be Quantified

Direct measurement of changes in pollutants coming from animal facilities and their contribution to impairment of downstream waterways is not currently possible for WQT programs. Sensors, sampling, and so on at the scales needed are prohibitively expensive and may not sufficiently quantify everything needed. Thus models (or calculations) are used to determine WQT credits now and for the foreseeable future. Direct measurement is used to improve models and understanding of BMPs and to assess cumulative outcomes of management at a watershed scale.

Models vary from simple empirical relationships or calculations based on some field measurements that are applied in similar places, to more complex empirical models that integrate more activities, transformations, or movements of pollutants, to process-based models based on the physics and chemistry of transport and reactions. To ultimately estimate the change in pollutant loading to a specific water body, a number of different steps need to be quantified along the way (Table 1).

Table 1. Types of Models Needed to Quantify Each Step in Nutrient Release, Transformation, and Transport.

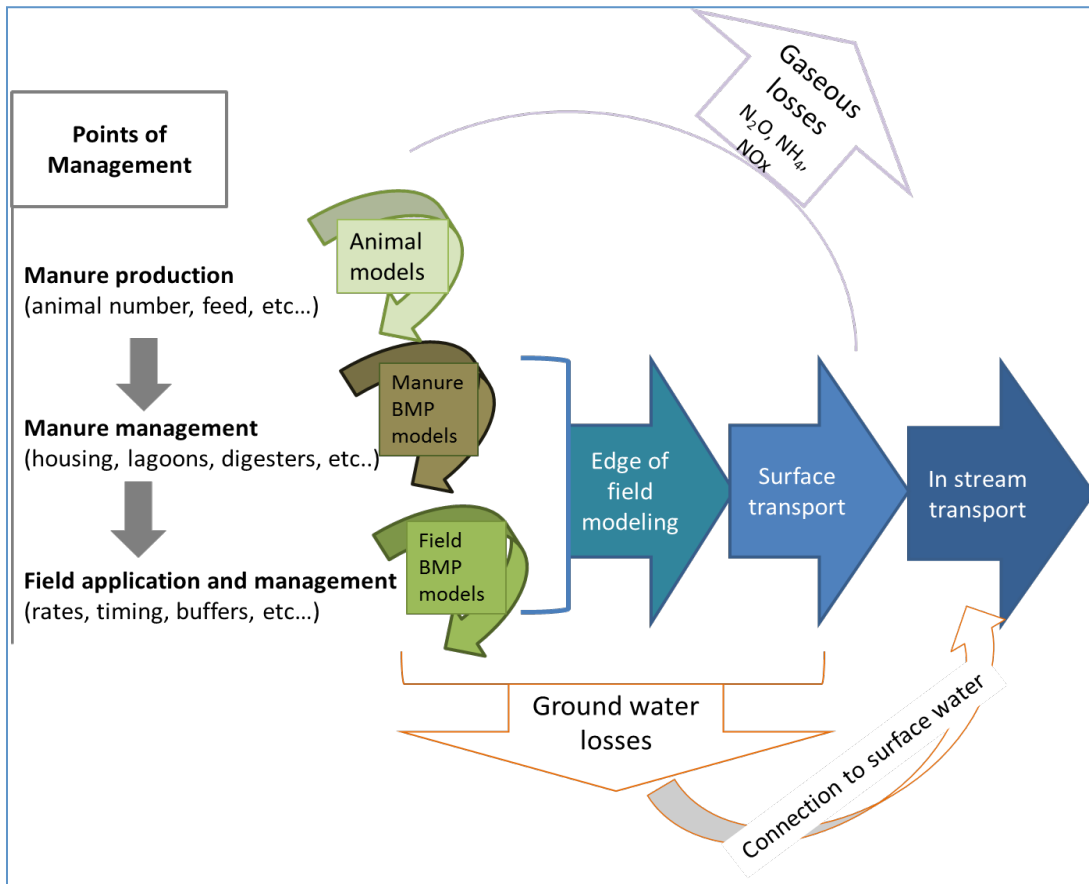
Pieces that need to be quantified	Model or calculation type	Examples*
Amount and nutrient content of animal manure	Animal excretion	Livestock nutrient excretion models
Amount and form of nutrients in manure management structures (lagoons, pits, digesters)	Manure treatment	DNDC, IFSM
Amount and form of nutrients in excess and mobile in soils	Fertilizer and manure application	P-Indices, ADAPT-N, IFSM, and most all models with a simple mass balance (based on yields and inputs)
Amount and form of nutrients transported to edge of field	Surface transport and transformation	APPLE, CREAMS/GLEAMS, EPIC
Amount and form of nutrients transported to waterway	Surface transport and transformation	APEX, WEPP, SWAT, GWLF
Amount and form of nutrients transported to impaired water body	In-stream transport	SWAT, GWLF, HSPF
Other environmental implications		
Groundwater loading and contribution to surface waters	Groundwater	ISSm
Atmospheric emissions	Biogeochemical	DAYCENT, DNDC, APEX, IFSM

Note: All example models listed here are described in detail in Section 2 of this report. This is not an exhaustive list and does not include many of the empirical models currently used in trading programs.

An animal operation can implement a variety of practices to reduce its pollutant contribution. AFOs can reduce nutrients available for loss by holding fewer animals, exporting manure out of the watershed, or shifting animal diets. Other practices attempt to reduce nutrient releases without reducing total nutrient quantities in the system. These include better manure collection and containment from barnyard and feedlots; preventing nutrient loss from storage and processing facilities; proper use of manure on cropland; and land and crop management to reduce, capture, or treat runoff and erosion (for details, see Morris 2014). Practices can be implemented in the animal housing, within the field, and at the edge of the field or farm. The type of practice will determine what types of measurement or modeling will be needed

to estimate pollutant reductions resulting from its use (Figure 3). No one model exists to estimate all the different transformations and movements of pollutants from animal operations to impaired waters, thus a series of empirical estimates and models must be used in combination. Figure 3 suggests including models of groundwater and atmospheric transformations. Although these flows are quite difficult to measure, some approach to estimate the risk or potential scale of losses through these pathways could be useful in avoiding unintended consequences. Any practice expected to have a significant impact on groundwater would likely not be allowed in a trading program. So, the modeling may be most useful where we have less clarity on the risks.

Figure 3. Models Needed for Quantifying Changes in Nutrients Resulting from Management.



Animal feeding strategies that match production requirements with feed content can reduce total nutrients in manure, which in turn will reduce nutrients in manure storage, processing, and land application. Livestock nutrient excretion models (animal models) are necessary to track the impact of these reductions.

Management strategies that affect manure storage and processing (storage structure; and physical, chemical, or biological treatment) can directly change nutrient released at the site, but can also change manure composition that is applied in the field. Both of these changes need to be quantified with animal system models that incorporate the nutrient transformations resulting from these management activities.

Quantifying the impacts of changes in field application and manure distribution (grazing density, location, filter strips, cropping practices, and constructed wetlands) will require input from the animal excretion

and animal system models. Once the type and amount of nutrients applied in the manure are determined, empirical or biogeochemical process models can be used to quantify transformations that occur in the field (soils processes, crop uptake, wetland dynamics, etc.) and to estimate excess nutrients.

After direct release from storage and processing or excess nutrients from field application are estimated, surface and lateral flows to the streams must be quantified, and from there the transport within the streams to the point of impairment, to determine the delivered load.

How Can Quantification Methods Be Improved to Better Support WQT Programs?

Improved and more standardized quantification methods may help support water quality trading programs by

- Lowering the costs to state regulators to develop program-specific calculations and tools;
- Improving risk-averse point source buyers' confidence in nonpoint source credits;
- Possibly lowering uncertainty and trading ratios, which could encourage non-point source trading;
- Integrating methods for estimating the potential risk of unintended consequences to groundwater, local air quality, climate change, and ozone depletion; and
- Improving watershed-scale modeling of BMPs to help programs target region-specific, least-cost best management practices to address pollutant loads and to provide additional benefits or avoid unintended consequences.

Standardized methods or protocols for calculating credits are needed for robust non-point trading, yet there are no standard models, tools, or calculators designed specifically for WQT applications. While a number of existing models and tools are available to spatially assess pollutant contribution, they have not been designed to calculate the pollutant load delivered from localized (farm-scale) management actions. The assessment of land practices on water quality is currently achieved through a combination of calculations based on empirical data adjusted for their application sometimes in conjunction with analysis tools such as SPARROW (USGS 2011), or specific hydrological models such as the Chesapeake Bay Model (USEPA 2010). Also, the USDA has been supporting efforts to adapt the APEX (Williams, Izaurrealde, and Steglich 2008; Gassman et al. 2010) model into a modeling tool called the Nutrient Tracking Tool (NTT), which can incorporate transformations and transport of nutrients all the way from management practice applied (BMP) to edge of field (see EPRI 2011 for a review of NTT).

METHODS AND MODELS FOR ESTIMATING NUTRIENT LOADING FROM ANIMAL FACILITIES

WQT programs depend on models that can reliably predict changes in pollutant loads in response to changes in management. Generally, current models are fairly reliable at the animal to field scales, but get progressively more difficult to use and are increasingly uncertain at watershed scales. Following this structure, WQT opportunities occur along a continuum from animal husbandry to manure containment and field application, to field and pasture management and edge-of-field containment, and ultimately to transport and transformations between field edges and watershed outlets. The primary focus is surface water quality, but because it may be important to consider groundwater, air quality, and greenhouse gas emissions, we also review approaches to addressing these system-wide challenges.

Numerous models could be used in a WQT program. This report neither details how these models were developed and how well they function for water quality trading nor recommends specific improvements to specific models. However, it explains how models are generally constructed, what kinds of errors they contain and why, and how these errors can be rectified (Krueger, Freer, Quinton, and Macleod 2007; Vadas et al. 2012). Four main pieces of model structure could be improved: (1) the *perceptual model*,

which is our basic understanding of how a system behaves; (2), the *conceptual model*, which is the mathematical equations used to describe the system; (3) the *procedural model*, which is how model equations are written and combined in computer code, and (4) *model calibration and validation*, which are the methods used to evaluate how well a model works.

Shortcomings in model development can exist in all four of these areas and can contribute to poor model predictions. For example, a poor perceptual model may occur when we simply do not understand the mechanisms that control fate and transport of a pollutant, or what we think controls them is in fact not correct. Historically, whether the earth is round or flat is an example. Even when a model is perceptually correct, the way it is translated into mathematical equations, or the conceptual model, can be incorrect. This can be due to a lack of data or inexperience on the part of model developers in translating data into robust and reliable model equations, and then integrating those equations properly. After model equations are developed and tested, they may actually be used differently in different models, giving rise to errors in the procedural model. For example, two model developers may incorporate the same empirical equation differently into their models on the basis of their understanding of the equation, especially over different spatial and time scales, or how it fits into the existing model structure. This can ultimately make two models give different predictions for the same set of parameters even though they are ostensibly supposed to operate the same. Finally, how we decide to run and test a model during calibration and validation can determine how well we think it performs, and this process can either expose or hide model weaknesses. Testing a model with limited data over short time periods or few scenarios may not reveal weaknesses that will be important when applying the model to other scenarios. This should not prevent an otherwise well-developed and diligently constructed model from being used for water quality trading. On the other hand, some models suffer from too many errors and should be avoided, even though they are advertised as robust and dependable.

Finally, model uncertainty is different from model error. All models have uncertainty that is introduced as scientific data are generated and models are built and tested (Bolster and Vadas 2013). Sources of uncertainty in all nutrient transport models that would be used for water quality trading can be grouped into three categories:

1. *Model structure uncertainty* is associated with approximating complex physical phenomena with simplified mathematical equations and the numerical methods used to solve those equations.
2. *Model input uncertainty* includes measurement uncertainty and the use of unrepresentative values for input variables. Model input variables are those physical quantities required to run the model but are measured independently, such as rainfall or manure application rates.
3. *Model parameter uncertainty* is from the constants that describe relationships between variables, the values of which are generally obtained through model calibration. Parameter uncertainty can come from using incorrect calibration performance measures (i.e., optimization targets); using inaccurate, incomplete, or unrepresentative datasets during calibration; or ignoring uncertainty in calibration data.

Although the importance of explicitly accounting for model uncertainty is widely acknowledged, it is not standard practice. This can be because some models cannot generate prediction uncertainties, some model users may not have experience in calculating uncertainty, or there is a fear that uncertainty may not be properly understood or relatively large uncertainty will undermine model credibility and the science behind it. However, ignoring uncertainty could lead to greater skepticism if modelers cannot quantify how confident they are in predictions. Thus, estimating uncertainty with model predictions yields more realistic model output and can lead to better use of modeling results and help alleviate skepticism.

Quantifying Effects of Animal Management

One opportunity to reduce N and P loss from a livestock operation is to reduce N and P available at the source, which is animal excretion. Non-spatial models that predict N and P excretion from livestock can also be included in groundwater and surface water models. Empirical models of animal nutrient excretion are useful because they require few and readily available inputs, but they may be limited in the number and types of management alternatives they consider. More mechanistic animal models such as IFSM and DNDC may simulate more management alternatives and may link them to other farm-specific components such as milk and meat production and nutrient export pathways such as gas emissions, but they are more complicated and may require more of less available data.

Review of N Excretion Models from Livestock Systems

Prediction of N excretion from livestock has been reported in the literature for several animal species. Estimates are inconsistent mainly because of assumptions used for predictions. The N excretion models can be categorized as (1) mass balance, (2) empirical, and (3) mechanistic.

Mass Balance

This is an accounting of N input (consumption by the animal) and output (in animal product) (Table 2). For example, N excreted by a dairy cow is the net difference of feed N consumed and N in milk.

Table 2. Dry Matter and Nitrogen Excretion Calculated by Water Quality Task Force of the UC Committee of Consultants.

		Dry Matter kg/head/day (lbs/head/day)			Nitrogen kg/head/day (lbs/head/day)			Nitrogen (% of intake)
Region	Ration	Intake	Output (milk)	Intake – output	Intake	Output (milk)	Excretion	Excretion
Santa Ana	Alfalfa	19.5(43.0)	3.07(6.77)	16.4(36.1)	0.518(1.14)	0.132(0.291)	0.386(0.851)	74.5
Central Valley	Multiple forage	19.3(42.5)	3.07(6.77)	16.2(35.7)	0.409(0.902)	0.132(0.291)	0.277(0.611)	67.7
North Coast	Cereal forage	20.0(44.1)	2.66(5.86)	17.3(38.1)	0.445(0.981)	0.114(0.251)	0.331(0.730)	74.4

Source: recalculated from UC ANR 2006

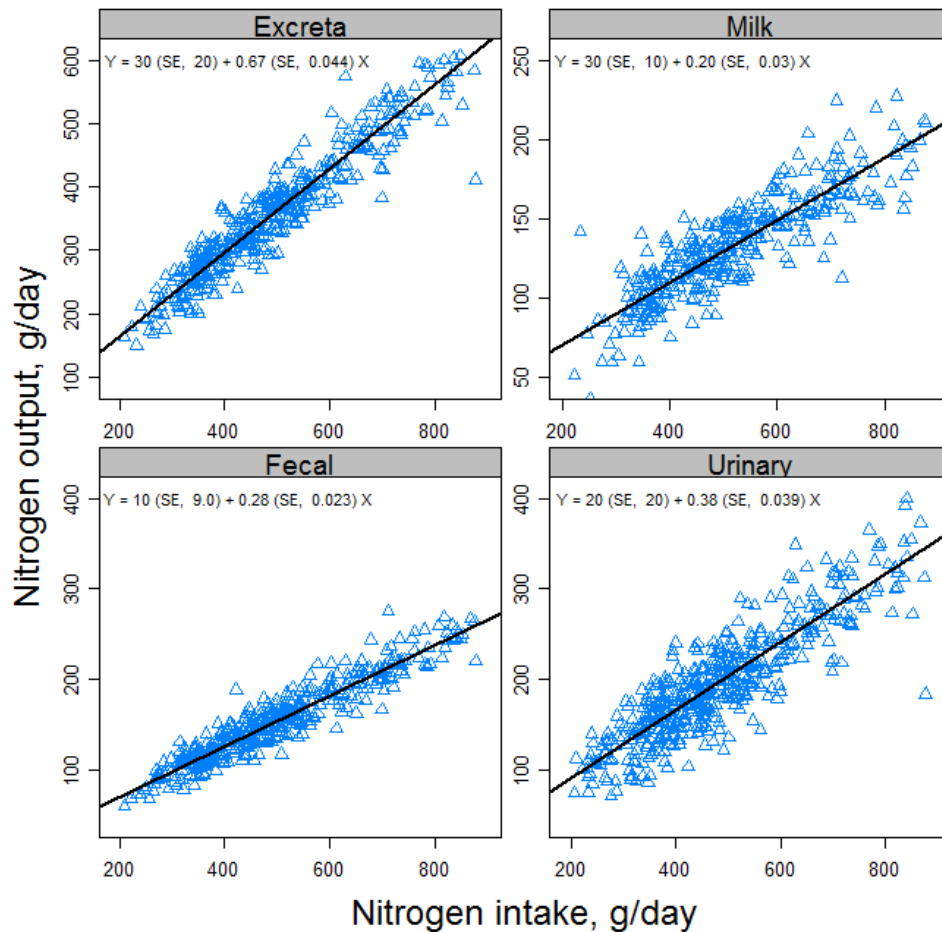
Note: Similar calculations for swine and poultry operations can also be made.

Empirical Models

Empirical models rely on data that correlate with measurements of N excreted, such as feed type, N ingested, and type of animal. Model accuracy relies heavily on data quality. Published empirical prediction equations for livestock N excretion are in Table A1, Appendix 1.

Dairy/beef cattle: Based on National Research Council (NRC) recommendations, the American Society of Agricultural and Biological Engineers (ASABE 2005) reported several equations and tables for estimating N excretion from dairy and beef cattle (equations 1 and 2, Appendix 1). Other empirical equations (equations 3–8, Appendix 1) have been developed by Kebreab et al. (2001, 2010). In general, N excretion is highly correlated with N intake (Castillo et al. 2000; Castillo et al. 2001a, b; Kebreab et al. 2001) (Figure 4). Other variables, such as energy content of the diet, can influence N excretion (Reynolds and Firkins, 2005; Kebreab et al. 2010; Reed 2012). In general about 67% of N consumed is excreted (Kebreab et al. 2001). For purposes of WQT, empirical models that calculate N content in manure will be adequate (e.g., Equation 3, Appendix 1).

Figure 4. Univariate Relationships between N Intake and N Excretion in Feces, Urine, and Milk.



Note: Excreta are the combination of fecal and urinary N. The responses have been adjusted for study effect. Standard errors, which were highly significant and a magnitude less than the parameter estimates, are given in parentheses.

Swine/poultry: ASABE (2005) gives equations for poultry (including laying hens), horses, and swine. Equations for broilers are included in Appendix 1 as examples for monogastric animals such as swine and poultry (Equations 9 and 10, Appendix 1).

The recently released nutrient requirement for swine (NRC 2012) provides several equations for different classes of animals (i.e., growing, lactating, and gestating). However, N requirement is the sum of essential and non-essential amino acids, which also influences N excretion. For purposes of WQT, a simple model for manure N content per finished animal (Equation 10, Appendix 1) is more appropriate because the added detail needed for phase prediction does not improve overall prediction uncertainty.

Mechanistic Models

A mechanistic model is based on knowledge of the animal biological system. A mechanistic model of N metabolism traces N transformations through pathways from feed consumption to excretion, including endogenous N losses. At each junction, the rate at which a particular protein or N compound is converted or transported depends on many factors, including the chemistry of the compound (e.g., protein vs. non-protein N, lysine vs. methionine), feed properties (e.g., energy content, fiber content, digestibility of

protein and energy sources), and animal physical properties. This kind of model is useful for understanding how N is processed, transported, and transformed and requires substantially more information than empirical models. While this level of detail may be more appropriate for modelling the fate of excreted N (e.g., Figure 2), it may be too complex for WQT purposes.

Dairy/beef: Kebreab et al. (2002) developed a mechanistic model of N metabolism in the ruminant. The model requires inputs of N fractions consumed and feed digestibility and includes microbial, amino acid, and urea pools to predict N partitioning. Model output includes N excretion in feces, urine and milk. Other mechanistic models of N excretion, such as MOLLY (Baldwin, 1995) and COWPOLL (Dijkstra et al. 1992), require detailed dietary input and software knowledge.

Swine/poultry: Strathe et al. (2012) developed a dynamic pig growth model (Davis Swine Model) for predicting manure volume and N content. Main model inputs are diet nutrient composition, feed intake, water to feed ratio, and initial body weight. The model partitions dietary digestible nutrients through intermediary metabolism to body protein and fat.

Although there are some mechanistic models of N excretion in poultry, most of them are industry-based and not publicly available. Gous (2007) suggested that feeding programs can be optimized using advanced simulation models. Most current models are based on least-cost, linear programming models for ration formulation.

Review of P Excretion Models from Livestock Systems

Empirical Models

Dairy/beef: Based on the NRC recommendations, ASABE (2005) reported equations for P excretion (e.g., for dairy cattle, Equations 11 and 12, Appendix 1).

As with N, P excretion is strongly related to P intake (Dou et al. 2002) (Table 3). In ruminants, most P is excreted in feces unless high concentrations of P are consumed.

Table 3. Phosphorus Intake, Excretion, and Water-Soluble P in Fecal Samples Affected by Dietary P Concentration.

Diet P <i>g kg⁻¹ (lb ton⁻¹)</i>		P Intake		Fecal P excretion			
				Total P <i>g d⁻¹ cow⁻¹ (lb d⁻¹ cow⁻¹)</i>		Water-soluble P	
3.4	(1.7)	97.3	(0.214)	41.8	(0.0921)	24.2	(0.0533)
5.1	(2.55)	142.1	(0.313)	96.8	(0.213)	76.4	(0.168)
6.7	(3.35)	171.0	(0.377)	113.0	(0.249)	94.4	(0.208)

Swine/poultry: The NRC (2012) gives empirical requirement estimates for growing-finishing pigs for different body weights and assumes that available P above requirement and undigested P is excreted. Schulin-Zeuthen et al. (2007) developed a regression equation for swine P retention and excretion (Equation 14, Appendix 1). For poultry, the ASABE (2005) gives an empirical equation for broiler P excretion (Equation 15, Appendix 1).

Mechanistic Models

Dairy/beef: Hill et al. (2008) developed a mechanistic model of P digestion and metabolism. Model inputs were total P intake in three forms (inorganic, phytic acid, and organic) and milk yield. Outputs were P in feces, urine, and milk. The P in feces was partitioned into phytate-bound P, organic P, and inorganic P. The model predicted excretion with reasonable accuracy and precision but with some systematic bias. The model predicted that total P in the diet had a greater effect on P excretion than any one diet fraction (Hill et al. 2008).

Swine/poultry: There are few mechanistic P excretion models for swine and poultry (Letourneau-Montminy et al. 2011; Dias et al. 2010). Kebreab et al. (2009) developed a dynamic and mechanistic model of P and calcium metabolism in layers that can evaluate feeding strategies to reduce P excretion to the environment in poultry manure. Such mechanistic models are data-intensive and may be better suited for research questions than WQT.

Applicability of N and P Excretion Models for Watershed Applications

Most of the mechanistic models of N and P excretion require detailed input information on chemical composition of animal diets, and to a lesser extent animal physiology. Although these models may provide more detailed information about the forms of N and P excreted than empirical models, which may be useful input to field-scale and watershed models, these models are currently not user friendly and require expert software knowledge. Thus, their applicability to WQT, especially at the watershed level, is limited. There is ongoing development of mechanistic manure biogeochemistry models (e.g., DNDC; Li et al. 2000; Figure 2) that interface with mechanistic excretion models (Salas, personal communication). These models may eventually be useful for determining how much N is lost to the atmosphere from manure storage and the specific forms of N and P applied to soils, the latter of which is useful as input to field-, farm- and watershed-scale models.

Whole-farm models may be useful for WQT because of the prebuilt farm-specific management options and interconnections. For example the Integrated Farm Systems Model (IFSM) (Rotz et al. 2012) integrates cattle nutrient excretion with manure storage and application to fields, and nutrient loss by multiple water and gas pathways. The model runs on default values that can easily be changed.

Several empirical nutrient excretion models are well-suited for WQT (Appendix 1). These models generally require very little input data and provide good estimates of total N and P excretion by different animals and the general excretion form, e.g., urine vs. feces. For WQT, the important variables are animal numbers and average crude protein and P content of the diet. With those readily available variables, we can calculate manure N and P contents for use with most current farm and watershed models. However, models that spatially integrate farms within a watershed are needed to assess contribution of livestock to water quality issues at a watershed scale.

Quantifying Effects of Best Management Practices for Manure Storage and Land Application

Among animal operations, animal-housing facilities can vary widely. Consequently, where and how manure is deposited by animals and how much manure is collected, stored, and processed before land application or transport off the farm vary. Manure processing may be intended to remove pollutants using chemical or biological means. Storage and treatment structures such as lagoons, anaerobic digesters, and compost piles biologically can alter the nutrient content and fate of manure, such as promoting ammonia

volatilization, nitrous oxide production, and denitrification. Total P is typically conserved in these containment structures. For WQT, empirical relationships or table values could be used to quantify the effects of manure storage and/or treatment on the nutrient content and forms in manure. However, more complex, biogeochemical-process models may be more appropriate when there is a need to address broader ranges of nutrient forms or changes to the manure storage system not considered by empirical research.

The Integrated Farm System Model (IFSM) includes empirical equations for N loss during manure storage and anaerobic digestion (Rotz et al. 2012). The storage component of IFSM is basic and is most useful for tracking the timing of manure application. The anaerobic digestion component primarily accounts for the carbon content changes to the manure, but also reflects the decomposition of organic N to ammonia N. IFSM is designed to be used in developing manure management strategies and, as such, is relatively parsimonious and reasonably user-friendly. There is also the Manure-DeNitrification-DeComposition (Manure-DNDC) model (Li et al. 2012). The core DNDC model is based on biogeochemistry principles, and the manure component tracks biogeochemical transformations through manure generation, storage, treatment, and field application. Manure-DNDC can be used to quantify changes in manure N and C for composting, lagoon storage, and anaerobic digestion. The model was developed to quantify GHG and ammonia emissions, but also to track the major forms of N and C applied to crops or pasture in its field component. This output from the Manure-DNDC model is designed to be added to the soil pools defined in the original DNDC model. Although the development of a web-based DNDC is underway, it is currently a relatively difficult model to use and requires substantial knowledge of the processes that are simulated.

Models for estimating P removal by chemical treatment of manure are not common, probably because chemical treatment of manure is a less common practice. However, there are examples of chemical equilibrium models developed for wastewater treatment systems being adapted for manure systems (e.g., Çelen et al. 2007). Similarly, agricultural models for predicting nutrient content following solids separation were not found in the literature.

Quantifying Surface Transport to Edge of Field and to Waterways

There are a variety of models to quantify the nutrients transported to the farm field edge and surface water. Some consider small, homogenous agricultural fields independently, others lump all of a watershed's heterogeneity into a few average characteristics, and more recent models adopt distributed or quasi-distributed representations of the landscape in which different parcels of land can react uniquely to rain and management practices (Heathwaite, Quinn, and Hewitt 2005). Models also range from event-based, to continuous-time, to quasi-steady-state or long-term average scales (Singh 1995). Most models are in a continuous state of evolution, making it impractical to provide a comprehensive overview of all their variations. Rather, we highlight their primary approaches and potential for use in a WQT program.

Two popular sub models deserve separate attention because they underpin many of the surface water quality models. The Natural Resources Conservation Service (NRCS)⁴ Curve Number (CN) model (USDA-SCS, 1972) and the Universal Soil Loss Equation (USLE) are empirical models used to characterize nutrient and sediment transport. Originally developed for other purposes, their use in predicting nutrient transport to edge of field and waterways is subject to criticism. Many models correct for this misapplication error with significant calibration routines. The misapplication of these fundamental equations becomes increasingly problematic when models are used in ungauged watersheds where calibration is not possible.

⁴ The NRCS was formerly the Soil Conservation Service (SCS) and the NRCS-CN model is sometimes referred to as the Soil-Cover Complex model.

Many models use empirical equations to describe the rainfall-runoff relationship, namely the NRCS-CN model. This approach uses empirically-derived, tabulated CN-parameters that relate storm runoff to soil characteristics and land use in ways that are generally consistent with infiltration excess generation of storm runoff (e.g., Walter and Shaw 2005). The NRCS-CN model is also embedded in the SCS-TR 55 model (USDA-SCS 1986), which allows modelers to estimate peak runoff rate. The CN model was developed for engineering design purposes and is probably inappropriate for continuous water quality models (Garen and Moore 2005).

Another nearly ubiquitous component of water quality models is the Universal Soil Loss Equation (USLE) and its revised (RUSLE) and modified (MUSLE) formulations. This is a largely empirical model based on estimating soil loss from fields, which is not strongly correlated with sediment loads in streams (e.g., Trimble 1999; Boomer et al. 2008). USLE-type models continue to be used because they are simple to implement and few alternatives have been proposed.

For U.S. rangelands, a large database of rainfall-runoff relationships and sediment transport was accumulated in the early 1990s. It comprised data from 444 rainfall event simulations at 26 rangeland locations in the Western U.S. About 60% of the variability in runoff could be explained using publicly available sources. Adding information about on-site measured ground cover and surface roughness helped to explain about 70% of the variability. Predictions of the sediment yield were slightly worse as only 60% of variation could be explained (Pierson, Pachepsky, and Weltz 2004). Importantly, a substantial effect of the plot size on sediment yield and runoff-to-rainfall ratio was found, making it imperative to apply scaling corrections to parameters derived from field experiments (Pachepsky, Pierson, Spaeth, and Weltz 2004). Further analysis of this database should be beneficial, as it allows one to quantify essential drivers of nutrient transport from rangelands.

Adapt-N (Melkonian et al. 2008) is a web-accessible tool coupling the Precision Nitrogen Management (PNM) model with near real-time weather data to make predictions for N application. PNM combined a maize N-uptake model with the Leaching Estimation And Chemistry Model (LEACHM) (Wagenet and Hutson 1987). The tool is specifically designed for 16 climate regions in New York State (Melkonian et al. 2007). LEACHM is unique in that it is based on fundamental physics of water and solute flow and does not rely on the empirical NRCS-CN and USLE models (Coats and Smith 1964; Van Genuchten and Wierenga 1976). Biogeochemical N processes include nitrification, denitrification, manure mineralization, and plant residue mineralization. The model is sensitive to a number of parameters that are difficult to determine (Hutson and Wagenet 1992; Lotse et al. 1992; Jemison et al. 1994). Adapt-N is not designed as a water quality risk tool but for improving the efficiency of N-fertilizer use. Additional modeling is necessary to simulate the effects of a change in fertilizer management on surface water quality.

Phosphorus indices are state-developed tools that use tables of rules or guidelines to classify field-level risks of P loss. They are intended to be risk management tools for general manure and fertilizer management and use information about P sources (soil P levels, fertilizer, and manure), and transport processes (erosion and runoff) to generate a field-scale assessment (Sharpley et al. 1994). Although the original P Index proposed by Lemunyon and Gilbert (1993) used the NRCS-CN equation and USLE to generate transport factors, many states have lessened their dependence on these models. For example, the New York State P Index uses USLE factors but employs a series of “rules” to identify areas likely to generate saturation excess runoff. Many similar approaches have been proposed in the context of Critical Source Areas (CSAs), which are based on the combination of source and transport factors similar to the P Index (e.g., Meals et al. 2008). About half of the 48 states that use the P Index do so with other measures of the environmental P threshold (Sharpley et al. 2003). Baker et al. (2001) is one of the few examples of trying to eliminate the use of NRCS-CN and USLE altogether.

Many different P-Indices exist across the United States, and Osmond et al. (2012) has shown that different indices give widely different levels of risk for the same field management conditions. Because few studies have been conducted to review the accuracy of P indices (Sharpley et al. 2012), it is difficult to know if some P indices accurately assess P loss from fields. Several short-comings in P indices and related improvements have been identified (Bolster, Vadas, Sharpley and Lory 2012; Gburek et al. 2000). Several researchers (Marjerson et al. 2011; Bolster, Vadas, Sharpley, and Lory 2012; Buchanan et al. 2013) have proposed methods to address some of these issues, albeit at a considerable sacrifice to the simplicity of current approaches.

The P index and Adapt-N can help producers guide their nutrient use, but do not always quantify pollutant loss. While some of these models do strive to quantify nutrient loss from fields, most of them lack the ability to quantify the impact of various management decisions and practices on downstream water quality. To the degree that they quantify nutrient loss from a field, they could be used to provide a worst-case estimate of nutrient loads to streams; however there may be additional nutrient retention or transformations between the field edge and the streams that would be missed, especially with respect to N. For example, most forms of the P index deliver a relative risk to surface water quality but stop short of predicting actual P loss from fields. Such tools are better in voluntary management situations, where farm practice recommendations are desired over pollutant load quantification. However, there are a few examples of these types of tools that provide quantifiable estimates of edge-of-field losses.

The **Annual Phosphorus Loss Estimator (APLE)**, a similarly simple management model, was developed in response to criticisms of the P Indices approach (Vadas et al. 2012). The model quantifies P loss from sediment-bound and dissolved P in runoff at the edge of a homogenous field. The quantification aspect of APLE is more useful to WQT than the original P Index risk results for certain management practices. APLE is recommended for WQT programs with fertilizer reductions as the primary best management practice for credit generation. As a field-scale model, APLE does not predict the effect of best management practices implemented outside of the field, e.g., buffer strips and constructed wetlands. Some APLE P loss equations have been integrated into the Wisconsin P Index (Good et al. 2012), another user-friendly tool that uses RUSLE and the CN to quantify P loss from fields.

Two early field-scale models that continue to be incorporated into newer models are **Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS)** (Knisel 1980; Singh 1995) and **Groundwater Loading Effects of Agricultural Management Systems (GLEAMS)** (Leonard et al. 1987). Several simplifying assumptions of the CREAMS/GLEAMS models limit applications to the field-scale, i.e., a homogeneous area, with one crop growing at a time and with the uniform fertilizer application, tillage, and planting dates. Although the original CREAMS allowed users to model storm runoff with a physically-based model (Green and Ampt 1911), later revisions used the NRCS-CN method. These models also use USLE to model soil erosion. Soil type enrichment coefficients determine organic N and particulate P losses. Nitrate in runoff is a function of soil porosity and nitrate concentration in soil surface layer.

The field-scale **Environmental Policy Integrated Climate (EPIC)** model (Williams et al. 1984) was developed to assess the effects of erosion on agricultural productivity but has since expanded to include a wide range of management practices and environmental impacts.⁵ **The Agricultural Policy/Environmental eXtender (APEX)** model (Williams, Izaurralde, and Steglich 2008; Gassman et al. 2010) was created to extend EPIC's capabilities to the farm-scale. Storm runoff is calculated with the NRCS-CN equation, erosion is estimated with one of several variants of the USLE, and wash-off coefficients based on soluble nutrients near the soil surface are used to estimate N and P in storm runoff. Phosphorus loss in EPIC/APEX is assumed to occur mostly in the sediment phase, therefore the soluble-P

⁵ EPIC is also sometimes referred to as the Erosion-Productivity Impact Calculator.

equation is simplified, especially for manure applications to fields. The P loading function for sediment transport is almost identical to the organic N transport. GLEAMS was added to EPIC/APEX to simulate pesticide transport by runoff, percolation, evaporation, and sediment. Crop growth and yields, tillage operations, irrigation, drainage, liming, furrow diking, pruning, thinning, harvesting, pesticide, herbicide, and fertilizer application can all be simulated. Additional code is available with APEX for structural management practices that effect nutrient and sediment transport (Waidler et al. 2011). Several interfaces are available for download (<http://epicapex.tamu.edu>). These assist the user in handling the input files and storing the scenario predictions. Some are linked with GIS applications for enhanced visualization and ease of subarea delineation. The Nutrient Tracking Tool (NTT)⁶ is a recent web-based application of the APEX model designed to provide a quick estimate of N and P credits for WQT (Saleh et al. 2011). The EPIC, APEX, and NTT models offer many user-friendly features, including a web-based, rapid assessment routine. The models can be used to model several common management practices including contour buffer strips, fertilizer management, and riparian buffers. The greatest caution in using EPIC/APEX is the oversimplification of hydrologic processes, as well as some issues with soil and manure P simulation.

Water Erosion Prediction Project (WEPP) is a field to small watershed-scale model for simulating soil erosion, sediment transport, and associated contaminants (Flanagan and Nearing 1995), and was developed specifically as a physically-based alternative to the USLE. Storm runoff is based on solutions of the Green and Ampt (1911) equation (Mein and Larson 1973; Chu 1978) and the kinematic wave equations. Options for simulating single storms, continuous simulations, single crop or crop rotation, irrigation, contour farming, and strip cropping are available. WEPP borrows a water balance using a module from SWRRB (Williams et al. 1985) and percolation from EPIC (Williams et al. 1984). The program design is modular, intentionally allowing for modules to be updated. Because WEPP was originally developed for erosion, NPS nutrient applications have been mostly limited to sediment-bound P (Perez-Bidegain et al. 2010). However, WEPP is likely to include a broader range of NPS pollutants in the near future, and a new user-friendly web interface has recently been beta-tested (E.S. Brooks, pers. comm.). Relationships of the WEPP model parameters with publicly available soil properties have been developed (Flanagan 2004), and this expands the applicability of this model to areas where no measurements of those parameters have been done.

The **Integrated Farm System Model (IFSM)** is a whole-farm, strategic planning tool that simulates a suite of major farm activities and calculates economic budgets (Rotz et al. 2012). Major model components include crop growth, grazing, machinery inputs, tillage and planting, harvest routines, crop storage capacity, herd and feeding, manure and nutrient handling, nutrient impacts as P loss, ammonia emissions, hydrogen sulfide emissions, grazing animals emissions, and greenhouse gas (GHG) emissions. This model also includes water quality predictions, mostly based on SWAT or SWAT-like approaches (e.g., Sedorovich et al. 2007), but without any spatial consideration. Phosphorus loss in runoff is modeled using functions from EPIC and SWAT with modifications from Vadas et al. (2005). Nitrogen cycling within each field is modeled with functions from the Nitrate Leaching and Economic Analysis Package (NLEAP) model (Shaffer et al. 1991). IFSM balances N losses from the field due to volatilization, leaching, and denitrification daily, but does not estimate lateral movement between fields or to waterways. Nutrient content of manure is estimated for the herd based on the nutrient content of the feed.

APEX and IFSM are promising tools for use in WQT trading because of their economic and management practice submodels. No other model reviewed in this report provides an integrated framework for modeling best management practices and their associated costs. Developing IFSM to spatially route nutrients between fields could expand use of the model to management practices that capture and remove nutrients between fields.

⁶ Formerly abbreviated as NTrT.

Quantifying Watershed Transport and Transformations

To quantify the effects of animal- to field-scale management on water quality, a watershed scale model is necessary. This larger scale modeling is also necessary for assessing best management practices that are implemented outside of farm fields, such as constructed wetlands and riparian buffers. Modeling nonpoint source pollution at the watershed scale is substantially more challenging than modeling at the animal and field scales. The first challenge is that significant modeling experience and input data are needed to run these models, which increases the transaction costs in a WQT program. Secondly, watershed scale models need to be calibrated, which presumes some monitoring data exist to compare with model output. Because model users can often calibrate any of these models to achieve good agreement between modeled and measured stream discharge and nutrient loads, especially monthly averages, it may seem like the model is correctly simulating the suite of transport and transformation processes. In reality, because these models generally have many parameters that can be adjusted, one can usually get the same answer using different sets of parameter values (e.g., Beven 1993). In other words, it is easy to get the “right” answer for the wrong reasons. A good modeler may be able to use these models in ways that estimate the nonpoint-source nutrient sources reasonably well (e.g., Easton et al. 2008), but there is considerable uncertainty of the effects of many management practices, especially those that require specific landscape positioning to be effective (e.g., Endreny 2002). The scale and structure of current watershed models are best suited to assessing the potential impact of large scale changes, e.g., the conversion of corn-cropping to alfalfa or the changes in fertilizer application rates. They are less than ideal for evaluating processes at sub-field scale or processes that require explicit routing of water and nutrients through specific landscape units, e.g., denitrification of nitrate-rich groundwater passing through a riparian buffer. Until resources are devoted to updating the science of these watershed models, we recommend using field and farm-scale models to generate WQT credits. Here we briefly describe the current status of several common models. All of these continue to evolve, and many researchers have used them effectively, so their applicability to WQT will likely improve over time.

The **Generalized Watershed Loading Function (GWLF)** was one of the early attempts at water quality modeling at the watershed scale (Haith et al. 1992). It uses the NRCS-CN model and USLE and characterizes the landscape as a set of hydrologic response units (HRUs) based on unique combinations of land use and soil groups. The model predicts stream flow, sediment, and nutrient loads at the watershed-outlet. Thus, it is a “lumped model” because there is no internal routing and the water budget is averaged over the whole watershed. Water quality is estimated using “wash-off coefficients,” which essentially assign a constant pollutant concentration to storm runoff. GWLF can be used to correlate land use with water quality but the effect of many BMPs is captured by adjusting the wash-off coefficients with little empirical justification.

The **Soil and Water Assessment Tool (SWAT)** is one of the most commonly used watershed models. Many components of the field-scale EPIC, CREAMS, and GLEAMS models were combined into a watershed framework to create a water quality model called the Simulator for Water Resources in Rural Basins (SWRRB) (Arnold et al. 1990, 1991), which subsequently evolved into the SWAT, which is a semi-lumped model (Arnold et al. 1998; Neitsch, Arnold, Kiriny, and Williams 2011). The addition of models for in-stream kinetics such as QUAL2K (Chapra et al. 2008), sediment routing, carbon-cycling, and various management practices (Gassman et al. 2007; Waidler et al. 2011) as well as improvements to the nutrient dynamics models (Vadas et al. 2010) are more physically based than earlier models. However, runoff volume is almost always estimated with the NRCS-CN method, although there is an option to use the Green and Ampt (1911) infiltration equation, which simulates infiltration from excess storm runoff. Peak runoff rate is predicted using a modified rational method proposed by Williams et al. (1995). SWAT delineates the watershed into HRUs based on land cover/use and soil type, but integrates storm runoff and pollutant loads at the sub-watershed level. Erosion is calculated with MUSLE. Because no water is exchanged between HRUs, many BMPs are difficult to meaningfully model in SWAT.

Usually BMPs are modeled by simply changing parameters within the model that the modeler believes represents the effect of a BMP (e.g., Bracmort et al. 2006).

SWAT has a reasonably well-developed user interface that works within ESRI's ArcGIS. Technical experts will likely be comfortable using these models and will find support from the user community. SWAT's dominance has meant that alternative models lag behind in terms of ease of use and familiarity. However, we re-emphasize that wholehearted adoption of the NRCS-CN method for runoff and the USLE model for erosion can be inappropriate for continuous, water quality applications.

The **Hydrologic Simulation Program-FORTRAN (HSPF)** (Bicknell et al. 2001) is another commonly used model and likely the most flexible. In addition to agricultural NPS pollution, HSPF has a wide range of applications, including flood control planning and operations, hydropower studies, river basin and watershed planning, storm drainage analyses, point pollution analyses, soil erosion and sediment transport studies, evaluations of urban and agricultural best management practices (BMPs), and pesticide and nutrient assessments (Donigan et al. 1995). The hydrologic component of the pervious land module is based on the water budget concept used in the Stanford Watershed Model (Crawford and Linsley 1966) and uses the Philip (1957) infiltration model to determine storm runoff, in which infiltration excess storm runoff is implicit (Johansen et al. 1980). The model structure is probably flexible enough to develop representations of other storm runoff processes, although this will require a modeler with a strong hydrology background. Sediment removal is modeled as wash off of detached sediment. Soil detachment is a function of rainfall, land cover, land management, and soil detachment properties. Many of the erosion process parameters are based on the USLE (USEPA 2010). HSPF is currently the core watershed model of the Better Assessment Science Integrating Point and Non-point Sources (BASINS) utilized and distributed by USEPA for Total Maximum Daily Load modeling. One of its most prominent applications is as the basis of the Chesapeake Bay Watershed Model (USEPA 2010).

Another example of a comprehensive model is the **Integrated Surface and Subsurface model (ISSm)**, which is essentially composed of three models: SWAT (hydrological model), and MODFLOW and MT3DMS (groundwater models). This model can predict surface and subsurface water quality and quantity as affected by anthropogenic activities in watersheds with regulated channel networks (Galbiati et al. 2006). Given the considerable expertise required to run any of the component models that make up ISSm, this model faces many of the same challenges as the others.

The **Watershed Analysis Risk Management Framework (WARMF)**; <http://www.epa.gov/athens/wwqtsc/html/warmf.html>) was developed by Systech Water Resources as a tool for users to develop and evaluate water quality management alternatives for a river basin. WARMF is organized into five linked modules. The engineering module is the dynamic simulation model that calculates daily runoff, groundwater flow, hydrology, and water quality of river segments and stratified reservoirs. Algorithms are derived from other models, such as the (1) Integrated Lake-Watershed Acidification Study model (ILWAS) for snow, river, and groundwater hydrology, lake hydrodynamics, and mass balance for acid base chemistry; (2) Areal Nonpoint Source Watershed Environment Response Simulation model (ANSWERS) for sediment erosion, deposition, resuspension, and transport; (3) Water Quality Analysis Simulation Program 5 (WASP5) for water body sediment sorption-desorption of pesticides and phosphorus and kinetics of nutrients and algal dynamics; and (4) Storm Water Management Model (SWMM) for urban pollutant accumulation on land surfaces and pollutant mixing and washoff. ILWAS routes precipitation through the vegetation canopies, soil horizons, streams, and lakes using mass balance concepts and equations that relate flow to hydraulic gradients. For erosion, ANSWERS simulates raindrop soil detachment using rainfall intensity and USLE factors, flow erosion using unit-width flow and USLE factors, and transport and deposition of sediment sizes using modified Yalin's equation.

WARMF is a relatively new model that has conceptually addressed many of the shortcomings in the hydrology that limit the meaningful functionality of the models discussed previously. It is supported by a company committed to keeping the code updated and providing useful interfaces that make the model widely accessible. The model accounts for many potential contaminants. Its complexity and comprehensiveness create many parameters and associated calibration challenges. The model developers are leveraging readily available data to reduce the number of calibration parameters. For application to water quality trading, the model needs to be tested at scales relevant to land management, i.e., field and sub-field scales (the model output has only been tested at the watershed scale). Moreover, descriptions of many processes, especially nonpoint source nutrient transport, need to be better described and traced back to more fundamental research so that the model's scientific basis can be examined. Although a complex, comprehensive model can be valuable, a model focused specifically on nutrients may be preferable for use in the WQT context.

Avoiding Unintended Consequences

Nitrogen transformations are frequent and common in the environment and can result in different pathways and forms of N loss (Townsend and Howarth 2010). Nitrogen can leach into groundwater or be emitted in a number of gaseous forms and create other environmental problems. Groundwater provides drinking water for more than half of U.S. residents. Where groundwater is shallow, nitrate contamination is common and a public health concern (Nolan, Hitt, and Ruddy 2002). Nitrogen can also be released into the atmosphere as ammonia (NH_3), which often results in regional N deposition and acid rain. Nitrogen can also be emitted as NO_x (nitric oxides), which contributes to air pollution. It is often released as N_2O (nitrous oxide), a potent greenhouse gas (almost 300 times the 100-year global warming potential of carbon dioxide) (Myhre et al. 2013) and now the largest contributor to stratospheric ozone destruction (Ravishankara 2009). The other form in which N is commonly emitted is N_2 , which is non-reactive and the most abundant gas in our atmosphere. Estimating these potential loss pathways can help to avoid WQT programs that incentivize management practices that result in unintended impacts on groundwater and the atmosphere. Surface and ground water quality models do not capture these pathways of N transport well resulting in large uncertainties, thus conservative program design to reduce risks of these losses is prudent. Some models and tools are available to estimate groundwater and atmospheric losses to help trading programs assess the risks of these unintended consequences.

If WQT and watershed managers are concerned about groundwater or atmospheric emissions, they can make policy choices and restrict questionable BMPs. Also, EPA provides guidance for pollutant feasibility evaluations to determine which pollutants and settings are appropriate or inappropriate for trading. However, they will need ways to assess these risks.

Modeling Groundwater Effects

Management measures in AFOs that address surface water quality may inadvertently lead to a deterioration of groundwater quality.⁷ Groundwater pollutants may also eventually compromise surface water quality when a portion of the groundwater returns to the stream as baseflow. Uncertainty in quantifying groundwater losses stems from an insufficient understanding of the groundwater-surface water interface. The unsaturated subsurface zone in soils is important to model because there is great potential for denitrification to occur. Denitrification is promoted in several management practices (e.g., wetlands and denitrifying bioreactors) as a way to remove excess N from the aquatic landscape permanently as, ideally, nonthreatening N_2 gas. These are often very effective practices, but current models do not simulate denitrification very well. Watershed models in this review strive to couple surface water and groundwater models, but do so in an overly simple manner that under accounts for the possibility of biological processes such as denitrification. Empirical relationships may be used to remedy

⁷ This section draws heavily on Harter, Kourakos, and Lockhart (2014).

discrepancies between shallow subsurface and deep ground water. However, a lack of field data leads to significant uncertainty in unsaturated zone and groundwater model predictions, especially when characterizing the impact of leaching on groundwater quality.

An assessment of nonpoint groundwater quality effects resulting from management practices involves three integrated systems: nitrate/salt source system (corral construction, lagoon design, crop management system), the root zone and underlying unsaturated zone, and the groundwater system. Groundwater nonpoint source assessment tools are grouped into three categories (NRC 1993):

1. Overlay and index methods for maps of qualitative indicators of groundwater vulnerability to pollution (Aller et al. 1987; National Research Council 1993; Civita and De Maio 2004; Pavlis, Cummins, and Donnell 2010).
2. Statistical approaches for the likelihood of pollution—some using existing water quality datasets and associated explanatory variables, some using regression (Nolan, Hitt, and Ruddy 2002), some using fuzzy logic (Uricchio, Giordano, and Lopez 2004), and some using artificial neural networks (Khalil, Almasri, McKee, and Kaluarachchi 2005).
3. Process-based methods that explicitly simulate the physics of soil and groundwater flow and transport. These approaches include zero-order mixing models (Mercado 1976; Lee 2007) or one-dimensional, plug-flow models that assume vertical advective flux of contaminants into the aquifer (Refsgaard et al. 1999; Cho and Mostaghimi 2009). More complex approaches include coupled one-, two-, or three-dimensional numerical flow and transport models. Three-dimensional models of soil and groundwater transport are computationally demanding (Harter and Morel-Seytoux 2013). At sufficiently high resolution (centimeter to meter scale), their application is limited to small sites. Alternatively, when simulating entire groundwater basins, these models are operated under relatively coarse resolution (hundreds of meters to kilometers) making significant assumptions about the physics of effective flow and transport processes at the scale of resolution.

An alternative method to a fully three-dimensional simulation approach is the streamline transport method (Ginn 2001; Weissmann, Zhang, LaBolle, and Fogg 2002; McMahon et al. 2008; Herrera 2010; Kourakos, Klein, Cortis, and Harter 2012). The method focuses on “sinks” or outlets of groundwater flow systems, for example, a gaining river section with groundwater discharge to the river or pumping wells with groundwater discharge into the well casing. However, this modeling approach is still largely in development and probably not appropriate for WQT at this time. See the supplemental paper by Harter et al. (2014) for more details and Appendix 2 for groundwater modeling resources and Appendix 3 for Vadose zone modeling resources.

Evaluating subsurface flows of water and contaminants has its challenges, especially limited data for important parameters. For example, hydraulic conductivity of an aquifer is highly variable but only measured in a few locations. In addition, preferential flow occurs in fractured rock and Karst aquifers, which is difficult to predict. On the other hand, the rapid transport of surface contaminants to the groundwater in these systems may offer limited attenuative capacity. Uncertainty also arises from the complexity of land use decisions that cannot be accurately captured on a regional scale. These land use decisions translate into boundary conditions and stresses that critically drive assessment models.

Our review suggests that adding groundwater monitoring to research and coupling the evaluation of monitoring data with integrated assessment and models may improve our understanding of the effects of livestock management practices on groundwater quality (Harter, Kourakos, and Lockhart 2014). Research on the integration of the models—coupling source systems with root zone/unsaturated zone pollutant fate and transport models, groundwater models, and surface water models is currently an emerging field in hydrologic simulation. There is an overwhelming lack of groundwater related data on the effects of

changing management practices in animal operations. Additional regulatory, funding, programmatic, and research resources are needed to address groundwater quality-related concerns in the context of WQT. Attention should be given to developing tools that help predict the risk of significant groundwater contamination across regions and common management activities.

Modeling Atmospheric Emissions

Empirical estimates (Millar et al. 2010) and biogeochemical models such as DAYCENT, DNDC, and APEX can be used to estimate nitrous oxide emissions (Olander and Hagen-Kozyra 2011). However, these estimates are based on very limited field data at this time, resulting in high uncertainty. While all these models include manure application on land, they do not include all the BMPs relevant to livestock management. Additional field research and data collection will be necessary to improve our estimates of atmospheric emissions of nitrogen. Strategic investment and coordination in data collection could reduce costs and enhance the efficiency by which such data is collected.

How Might Animal Grazing Systems Differ in Their Impact and Quantification

In rangelands in the central and western regions of the United States, the major activity is extensive cattle and sheep grazing at low stocking rates (>10 acres per animal unit per year) (Valentine 2000).⁸ There is little use of fertilizers, irrigation, or imported feedstuffs. Concerns have been raised that sediment, pathogen and nutrient pollution by livestock grazing on rangelands can degrade water quality (Belsky, Matzke, and Uselman 1999; Derlet and Carlson 2006; Myers and Whited 2012). Pollutants of concern include sediment, fecal coliform and E. coli, and N and P (Field and Samadpour 2007; Hubbard, Newton, and Hill 2004).

Management practices designed to reduce the impact of grazing on water quality are different than those for croplands. Practices focus more on how the number of animals and their distribution over time affect sediment, microbial contaminants, and N and P loss. For grazing and rangeland, animal manure (dung) is spread in patches across the landscape. This is not often well captured by surface flow models, although improvements have been made (Vadas et al. 2011). While a couple of models incorporate both cropland and grazing animal systems and management, these are often separate. While modeling of surface and watershed transport will be the same for N and P, it is a bit different for sediment and microbial contaminants, which can lead to use of different models.

From a water quality perspective, rangelands of the central and western United States are most susceptible to high intensity storms that generate infiltration excess storm runoff, which can have considerable power on steep slopes.⁹ There are numerous mechanistic models that simulate this type of storm runoff; KINEROS2 (Goodrich et al. 2012) is one that has been explicitly incorporated into a rangeland decision support tool, Rangeland Hydrology and Erosion Model (RHEM) (Goodrich et al. 2011). Unlike models that rely largely on derivations of the USLE to predict soil erosion, RHEM uses the Water Erosion Prediction Project (WEPP) model to simulate erosion more mechanistically. Other modules have been coupled with KINEROS2 to predict nutrient and bacteria transport. This type of model typically requires lots of input data, much of which is difficult to obtain. However, a graphical user interface¹⁰ facilitates use. Because of the mechanistic nature of RHEM, it can reliably simulate changes in water quality from common rangeland management practices, such as timing and intensity of grazing to allow vegetation to

⁸ Content in this section was based on Tate et al. (2014).

⁹ While this process occasionally occurs in other parts of the country, its rarity relative to saturation excess processes and the relatively more complete vegetative cover make these high-intensity events less important.

¹⁰ The Automated Geospatial Watershed Assessment tool (AGWA) (www.tucson.ars.ag.gov/agwa) is an ARCGIS interface to support data organization, model parameterization, integration, and visualization for KINEROS2, RHEM, and the Soil Water Assessment Tool (SWAT9).

recover, using drinking water and feed placement to distribute the herd and avoid riparian areas and vegetated filters around waterways (Briske 2011b; Briske et al. 2011a; George, Jackson, Boyd, and Tate 2011).

Opportunities for grazing land systems to participate in regulatory driven water quality trading programs may be low given that impaired waters and pollutants that are under regulatory limits (TMDLs) are not in the same places as grazing lands. Voluntary mechanisms such as source water protection and state or federal support for best management practices (BMPs) are more promising tools for engaging rangeland managers in projects to improve water quality (Musengezi et al. 2012).

State of Modeling Tools

The state of water quality modeling is continually evolving. Current models generally have good predictive power at animal-to-field scales, which allow us to estimate the fraction of nutrients remaining on the farm. These models are already being incorporated into relatively user-friendly support tools with graphical user interfaces. With the exception of rangelands, are considerably more uncertain than animal, field, and farm models, and a number of challenges need to be addressed so they can be more widely adopted in water quality trading.

Modelers must have a good understanding of the physical processes underpinning nonpoint source pollution, because many of these processes—in particular, generation of storm runoff—are not meaningfully represented by water quality models' legacy submodels. These submodels implicitly assume that the infiltration capacity of the landscape is the limiting variable, whether they use the NRCS-CN, Green and Ampt, Philips Equation or other similar equation. Schneiderman et al. (2007) and Easton, Walter, and Steenhuis (2008) demonstrated the potential of using current models in ways that could overcome this problem, but these techniques require considerable modeling and hydrologic expertise.

Another challenge is an incomplete understanding of important processes. For example, riparian denitrification is recognized as an important means of removing nitrate from groundwater flowing into streams (Groffman 1992). However, our ability to model denitrification is relatively poor, and our understanding of how groundwater and surface waters interact is evolving.

There are a number of very good hydrological and improving biogeochemical models in the research arena that could substantially improve our ability to model nonpoint-source pollution at the watershed scale, e.g., TOPMODEL (Beven and Kirkby 1979), SMR (Frankenberger 1999), RHEESYS (Tague and Band 2004). In order to move these into an operational arena, financial resources are needed as well as institutional commitment to maintain and distribute the models.

MONITORING AND MEASUREMENT

Direct measurement and monitoring of nutrient losses is evolving with new technologies and can be used to adaptively improve the quantification and modeling of nutrients over time, to measure cumulative change of a water body, and perhaps eventually to quantify edge-of-field losses.

The Role of Measurement and Monitoring in Supporting WQT Programs

As technology and methods for directly measuring and monitoring N and P become increasingly accurate and more readily available or less expensive, more opportunities arise to use measurement as a complement to models. Although monitoring may not be applicable to WQT in the short term, it will likely benefit future WQT programs. Three ways measurement and monitoring can be used to support quantification in WQT programs are to

1. **Improve models**—Additional data at both the field and watershed scale, particularly data for relevant practices and affected watersheds, can assess uncertainty and refine or validate empirical and biogeochemical models.
2. **Measure and improve cumulative outcomes**—Monitoring of target water bodies can help assess the effectiveness of WQT programs and consider the role of other drivers of water quality (atmospheric deposition, groundwater recharge, and other non-point sources) to allow for adaptive management and improvement to programs over time. Direct measurement can assess if practices are leading to the desired water quality improvements, but measurement must be sustained over long periods (Easton, Walter, and Steenhuis 2008; Gregory et al. 2007)). Such efforts need to account for the complexity of time, climate, and ecological variability, which can mask the link between practices and water quality.
3. **Directly measure and monitor performance of animal management practices**—While impractical to implement in current programs due to cost and complexity of existing methods, developing technologies may make this possible in the future. Recent advances, such as remote sensing and in situ water quality sensors, are reducing costs and making basin-wide or site-level monitoring feasible. This is already occurring in the Great Miami River trading program.

Quantifying water quality benefits of management practices in animal operations will require a suite of measurements from the process-based practice scale to the whole-basin scale. Within and across these scales, measures must be integrated through empirical synthesis and linked to models. While a range of approaches and technologies are needed to link models and measurements for WQT markets, emerging sensor technologies for high-frequency water quality measurement show particular promise.

Measurement at Different Scales

Practice-specific Measurements

Practice-specific measurements play a number of roles. They (1) provide a rigorous and process-level understanding of the effects of specific practices on nutrient export, which is needed to build a foundation for WQT markets; (2) help identify co-benefits or tradeoffs of practices (e.g., effects on greenhouse gas emissions); (3) provide data for cross-site synthesis and model development and validation; and (4) assist in the development of more accurate or less costly approaches to quantifying nutrient reductions.

Measurements of a specific practice can quantify reductions in nutrient export even if field and watershed scale reductions vary based on time lags and complexity associated with larger systems. This is particularly valuable when changes can be compared to direct measurements of a process thought to control nutrient loss. Mechanistic understanding is valuable in developing models that can assess tradeoffs and co-benefits of a practice (Kulkarni, Groffman, and Yavitt 2008). Integrated, quantitative measurements of multiple benefits remain rare.

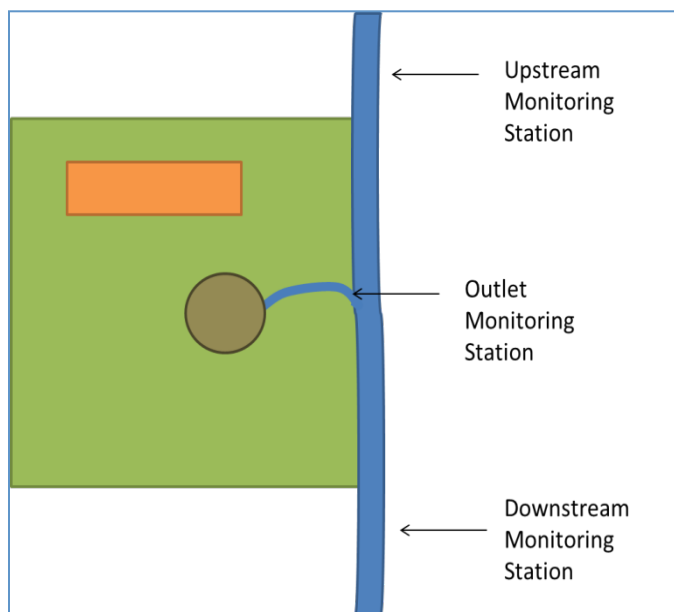
Operation-Scale Measurements

Measurements of nutrient export at the whole-operation scale can be valuable because the operation is an important scale for farmers making decisions about practices, and it is the scale to which credits will be applied. An operation can aggregate the outcome of multiple management adjustments applied together. As with practices, intensive operation measurements across a whole basin are too expensive, so the primary goal at the operation scale is to use targeted measurement to improve models.

Compliance data reported by permitted CAFOs—animal numbers, waste quantity and nutrient content, types of treatment, and rates of discharge or land application of wastes—can help estimate nutrients susceptible to loss or serve as inputs to models (Koelsch 2005).

Measuring nutrient export at the scale of operations is extremely challenging because of the number of nutrient loss pathways and because the boundaries of operations often do not coincide with landscape features that control hydrologic flow paths. However, because operations are such an important scale governing the implementation of practices, we highlight two approaches for direct measurements. Where operations are adjacent to waterways, changes in concentration between stations located upstream and downstream could be used to estimate export from the operation (Figure 5). This requires no other inputs from other land, managed differently or by others, between the monitors. In cases where operations occupy whole watersheds, nutrient export can be estimated from a single measurement over a period of time at the outlet of the watershed. These approaches can assess the effects of practices either by comparing multiple operations that differ in their practices or by comparing pollutant export over time as practices are put in place (e.g., Rao et al. 2009).

Figure 5. Schematic for Whole-Ecosystem Measurements of Nutrient Export from Animal Operations.



Note: By monitoring nutrient concentrations upstream and downstream of a facility, measurements can account for episodic surface-water runoff and longer-term groundwater contamination that will not be included in estimates of direct waste discharge. A comparison of measurements between conventional and alternative practices will provide the most direct and complete measure of nutrient-load reductions. This approach will work best when the hydrologic footprint of an animal operation can be definitively determined (i.e., through groundwater tracers).

The value of operation-scale measurements is that they more directly quantify nutrient export coefficients (Johnes 1996) compared to statistical analyses of large watersheds with mixed land uses and practices. As at the practice scale, there is considerable variability in nutrient export over time, spatial scales, and in the implementation and effectiveness of practices. As a result, effective field or operation-scale monitoring systems will often be difficult to establish with consistent QA/QC protocols due to inherent variation in field scale siting issues and difficulty in consistently capturing ephemeral flows at field scale.

Watershed-Scale Measurements

Ultimately it will be important to assess the effectiveness of water quality trading in the context of other changes taking place in the watershed to assess the overall impact on impaired waters. This requires long-term records because at large scale it often takes years for practices to have a measureable effect (Meals,

Dressing, and Davenport 2010), especially because weather creates variability in pollutant measurement from year to year (e.g., Ryan et al. 2006). Nonetheless, measuring changes in water quality is more a function of time and resources than technology or approach. Resources for long-term monitoring are scarce and measurements are often infrequent and spatially sparse.

Watershed-scale (ranging from a few hectares to 100s and 1000s of km²) measurements are the most common approach to quantify the export of nutrients from animal operations and other land uses, and have been used to infer the effects of management practices (e.g., Woli, Nagumo, Kuramochi, and Hatano 2004). One benefit is that watershed measurement integrates spatial heterogeneity and temporal variability (Ellison, Skinner, and Hicks 2009; Fan et al. 2012), but this can also complicate efforts to link nonpoint-nutrient export to specific areas or operations.

As noted above, models exist to predict nutrient export from animal operations. One type of model uses coefficients to estimate nutrient export as a function of farm nutrient balance or rates of land application, and these coefficients are often based on watershed-scale analysis of the abundance of a particular land use and measured water quality (Johnes 1996; Bishop et al. 2005; Rao et al. 2009; Li et al. 2011; Nestler et al. 2011), although relatively few studies have focused on animal operations. Uncertainty with export coefficient models can be reduced through more intensive sampling.

Other models incorporate more hydrologic and biogeochemical realism into their predictions (Band et al. 2001; Vadas et al. 2007; Vadas et al. 2008; Arnold et al. 2012). These process-based models are often calibrated using measured watershed data (Wade, Jackson, and Butterfield 2008). Such data can help constrain model parameters to realistic ranges (Arnold et al. 2012).

More computational power is enabling process models to simulate nutrient export at finer resolutions in space and time. Where networks of measurements (e.g., of streamflow) are available, models can use this spatial information to better assess their predictions, but such efforts are often limited by insufficient data (Dupas et al. 2013).

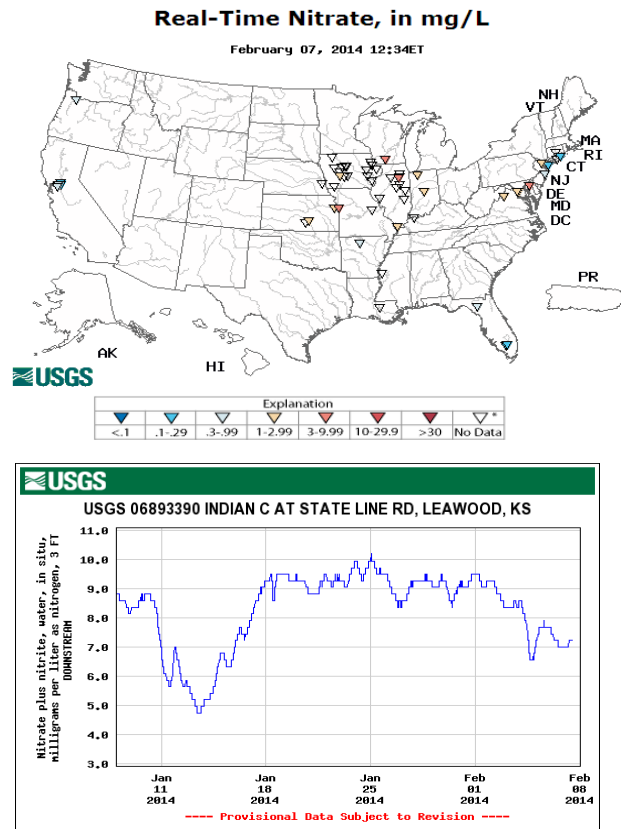
Currently, there are precision conservation technologies being developed that can identify where in a watershed a certain practice should be placed to have the greatest chance of having the benefit detected at the watershed outlet. These spatial technologies can improve identification and prioritization of WQT opportunities in a watershed compared to stand-alone simulation modeling (Tomer et al. 2013a,b).

Advances in Nutrient Sensors and Measurement of Water Quality

In situ sensors are instruments that directly measure and record the physical, chemical, and biological characteristics of water or air without the complication, expense, and error of lab analysis. In situ sensors that measure a limited number of water-quality parameters have been commercially available for several decades, including parameters such as temperature, barometric pressure, pH, specific conductivity, and dissolved oxygen. New technologies can dramatically increase the number, frequency, diversity, and sometimes accuracy and precision, of measured parameters. For example, new optical technologies can measure NO₃, colored dissolved organic matter, and proxies of biological activity such as chlorophyll A (Saraceno et al. 2009). The accuracy and stability of dissolved oxygen has also improved with new optical sensors. New ion-specific electrodes can measure a wide range of charged solutes. Finally, the miniaturization of analytical chemical techniques allows in situ sensors to measure chemicals like NH₄ and PO₄ (Cohen et al. 2013b) and meteorological sensors to continuously measure biogeochemically-significant gases and physical parameters (Harper et al. 2004; Gao et al. 2011). All these technologies benefit from, and depend on, improved data storage for longer and denser records, and data transmission for real-time observation of water quality parameters.

Many in situ nutrient sensors require substantial up-front investment as well as continued costs for maintenance. However, these costs can be comparable to, or even less than, laboratory analysis when the lifespan of the instrument and the density of data are considered. One limitation of some sensors, especially optical ones, is that their precision depends on environmental conditions (Downing et al. 2012; Cohen et al. 2013a). Despite these limitations, new technologies could dramatically improve assessment of animal operation practices and water quality.

Figure 6. High-Frequency Nitrate Monitoring by the U.S. Geological Survey.



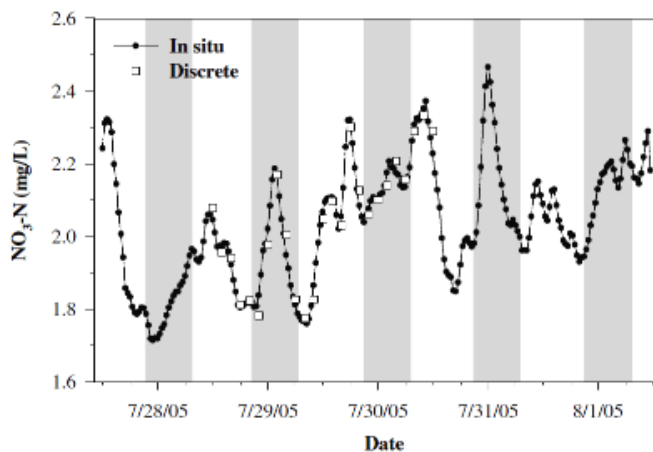
Note: (Top) The USGS currently maintains NO_3 sensors at >50 stations, primarily in the Northeast and Midwest. (Bottom) One month of hourly nitrate data from Indian Creek, Kan., showing large swings (between 5 and 10 mg $\text{NO}_3\text{-N L}^{-1}$) associated with high-flow events as well as finer-scale diel variation.

As in situ nutrient sensors have become commercially available, they are becoming tools of monitoring as well as research. Recently, the U.S. Geological Survey began to deploy real-time, high-frequency nitrate sensors as part of its ongoing water quality monitoring program (<http://waterwatch.usgs.gov/wqwatch/>; Figure 6), and have documented best practices for sensor operation and maintenance (Pellerin et al. 2013). A select number of local jurisdictions have also experimented with these technologies (e.g., Deschutes, Wash.) (Sackmann 2011). At present, nutrient sensors operated by monitoring programs are primarily deployed in larger rivers, where their value for WQT models may be limited. Current monitoring networks are also geographically restricted. However, continued advancements in technology and expansions of nutrient sensor networks are likely, both geographically and within river networks. Existing observation stations provide an opportunity to improve model-data links.

A major benefit of high-frequency measurements is improved precision and confidence in export estimates. The concentration of many solutes can vary widely over seasons, days, and even events

(Kirchner, Feng, Neal, and Robson 2004; Nimick, Gammons, and Parker 2011; Saraceno et al. 2009; Pellerin et al. 2012) (Figure 7). Infrequent samples miss these daily and event-based dynamics, and as a result seasonal and annual estimates can err significantly (Stelzer and Likens 2006). High-frequency measurements from in-situ sensors can reduce these errors (Worrall, Howden, Moody, and Burt 2013) as well as provide a better understanding of the origins and processes of nutrients export. As a result, uncertainty around both empirical and modeling estimates of the benefits of a nutrient management practice will be reduced, with benefits for both policy makers and WQT participants. With improved confidence in the nutrient reductions associated with particular practices, policymakers will be better able to predict reductions on a broad scale and to distinguish the relative benefits of various practices.

Figure 7. Hourly Measurements of Nitrate Concentration Over a Five-Day Period in the San Joaquin River, California.



Source: Pellerin et al. (2009).

Note: The measurements illustrate a large variation within and across days. The irregular pattern in NO_3 concentration is attributed to discharge patterns from surrounding farmlands.

By better describing the relationships between flow and concentration, high-frequency measurements enable nutrient fluxes to be better allocated among baseflow and rain events. In watersheds of mixed land use, temporal variation in the concentration of nutrients can help identify sources of nutrients (Pellerin et al. 2009) and infer the magnitude of transformation processes (Heffernan and Cohen 2010; Cohen et al. 2013b). Models may need refinement before they can be used to understand and predict high-frequency nutrient dynamics, but such model-data integration should be a high priority for research development (Radcliffe, Freer, and Schoumans 2009). These watershed-scale applications are valuable where nutrient export is likely to be highly variable in space and time.

High-frequency measurement may also improve estimates of episodic and daily variation in nutrient export from wastewater lagoons (a process that is poorly understood) and improve waste management (Wang, Zhang, Peng, and Takigawa 2008; Claros et al. 2012; Ruano, Ribes, Seco, and Ferrer 2012). In municipal wastewater treatment plants, continuous monitoring is used to optimize treatment, detect malfunction, and control the timing and volume of discharge (Vanrolleghem and Lee 2003). Such applications could clearly be transferred to wastewater lagoons associated with AFOs.

BUILDING BETTER QUANTIFICATION TOOLS

Models already and will continue to play a significant role in WQT programs (Krueger, Freer, Quinton, and Macleod 2007; Vadas, Bolster, and Good 2012; Bolster and Vadas 2013). For animal agriculture and nutrients, models in some form already exist to meet the requirements for WQT, which is to help develop credits and trade ratios by 1) quantifying nutrient loads and load reductions at a practice scale, and 2) quantifying the degree of nutrient-load delivery from the point of practice to the point of water-quality interest. Thus, developing a WQT program may be helped by the improved functionality and quality of models.

The process by which a WQT program may select which models to use helps reveal ways they could potentially be improved. Important questions about model characteristics for WQT include:

- Does the model adequately represent the pollution load reductions?
- Does the model operate at an appropriate scale and resolution?
- Can the model be adapted to local conditions and realistically represent local agricultural systems and practices?
- Do model data requirements match program data availability?
- Are model sensitivity and uncertainty appropriate relative to the magnitude of desired pollutant load reductions?
- Has the model reasonably kept pace with advances in scientific understanding, and has it been well developed and tested by rigorous scientific principles?
- Does the model deliver information in the same units and on the same time scale as the regulatory water quality standard?
- Is the model user-friendly for WQT personnel, and will the model give consistent results across multiple users for the same scenarios? Is the model practical and economical to set up and apply?
- Is the model transparent enough so model algorithms are understandable and simulated flows and pathways easy to follow? Does the model have adequate support to be applied and updated as needed?
- Is the model compatible with other program models so the program performance and success can be evaluated?

No single model may meet all of these criteria, and a WQT program will have to decide which criteria are absolutely critical and which are not. These criteria should also not be considered a checklist. While a model may ostensibly meet certain criteria, it may not do it very well. Failure of a single model to meet many of these criteria is evidence that a particular model should not be used or needs to be improved if it is the only available option. Failure of most available models to consistently meet select criteria may indicate the need for new modeling approaches in general.

Models can always be improved, and model weaknesses should not prevent developing a WQT program, especially because one of the best ways to improve a model is to have people use it in a targeted program. If model weaknesses are well understood, then a WQT program can be designed to account for those weaknesses. Model weaknesses are understood by knowing how models work, their equations and algorithms, and not just how to make them run and give output. While this is obvious, in practice some models can be so complex that it is very difficult for anyone except the model developers to know how the algorithms in the models work. This is typically truer of models like SWAT, which have many parts and operations because they are designed to simulate the fate and transport of multiple pollutants over large geographical areas, than models like APLE, which are designed to simulate a single pollutant at the field or practice scale. Thus, complexity typically arises from the scale of the model and not the type of model mathematics. Complexity also implies that model users may not be able to identify model

weaknesses when a model runs and generates reasonable output. Thus, it may fall to model developers to effectively communicate their understanding of the models and related weaknesses, but without creating unreasonable doubt about real model strengths and abilities.

Principles and Priorities for Using Models in WQT Programs and Model Improvement

Models need to provide reasonably accurate predictions of water quality impacts to allow generation of water quality credits and trade ratios. Models already exist to do this and are already being used in WQT programs, typically with conservative modeling assumptions and trade ratios. Model improvements can help reduce trade ratios and allow for more reliable trading credits.

Some specific model improvements include:

- Alternatives to the CN approach should be developed and implemented (e.g., TOPMODEL) for measures of runoff.
- Improvements to the USLE family of erosion models are needed.
- Better simulation of the link between field practices and water quality at the watershed outlet. This could include better nutrient cycling and sediment transport predictions in streams.
- Nutrient cycling in soil is generally well simulated, but incremental improvements can be made, especially to reflect technology changes in farm management and scientific understanding.
- Estimates of uncertainty are needed for full confidence in model predictions.

In general, current, empirical livestock nutrient excretion models are reasonably accurate, reliable, and easy to use, and are sufficient at this time because uncertainty due to nutrient excretion is much less than uncertainty due to other model operations such as hydrology and erosion prediction. Nutrient excretion models could be directly linked with farm management and/or water quality models, provided these links do not mean unnecessary or excessive model complexity or data requirements. However, direct measurement of nutrient content of animal excreta is still a viable alternative to models at the operation or field scale for quantifying trade credits and ratios.

Water quality in both surface and groundwater systems is important, and both should be considered when evaluating WQT credits. The best way to link groundwater and surface water considerations, and whether or not this needs to involve groundwater models for WQT programs, is debatable. However, coupled models will help identify potentially confounding practices and inform conservation practice implementation. The same principle applies to atmospheric greenhouse gas and ammonia emissions.

Management practices need to be vetted for the consistency with which they contribute to WQT goals. There should be an effort to focus on those practices that comprise the “low hanging fruit,” to guide trial implementation of WQT schemes that include practices that are relatively easy to implement and monitor, underpinned with fundamental work to include those same practices in models. Furthermore, simulating such practices in models through mechanistic representations often generates a more robust, dynamic model compared to simply assigning a coefficient or including a regression result. Field and farm practices must be linked to watershed-scale improvement when generating WQT credits. Models must work across scales to accurately track the impact of credit-generating management practices. Field and farm-scale models generate predictions of the effectiveness of a practice, but do not necessarily accurately predict the impact at the watershed outlet. Nested modeling systems are needed to generate the value of the credit by combining the potential practice effectiveness and its proportional contribution to watershed improvement. However, current models are much better at field-scale than watershed-scale water quality predictions, especially for fast, affordable estimates that minimize the expertise and resources needed to run the models. Linking field practices to the impact at watershed outlets is at the edge of scientific understanding, but remains a priority for WQT modeling.

The computing foundation of many water quality models has not been updated for 20 or more years. Model code and formats need to be kept up to date with computer technology and model languages, including spatial and online or mobile possibilities and user-friendly interfaces. Models should be constructed to be more modular and move towards smaller, linked models, instead of single, massive models that are difficult to operate, decipher, and modify. Modularity makes models more manageable and transparent, allowing them to be more easily updated based on scientific understanding. However, fancy technology does not improve a model's accuracy. The translation of water quality science into models often lags years and even decades behind current scientific understanding. Some model weaknesses can be readily and rapidly improved by addressing this situation. Furthermore, model development efforts need to be more collaborative instead of competitive. Similarly, researchers may have a sense of how to set up projects that leverage a combination of monitoring and modeling, but related guidelines are needed for projects in the context of conducting trials for WQT schemes.

APPENDICES

Appendix 1: Empirical Animal Nutrient Excretion Models

Table A1. Empirical Prediction Equations for Nitrogen and Phosphorus Excretion from Livestock.

Eq.	Prediction equations for nitrogen excretion ¹	Reference
1	$N_{E-T} = \beta \sum_{x=1}^n (DMI_x * C_{CP-x} * DOF_x * /6.25) - [41.2 * (BW_F - BW_I)] + [0.243 * DOF_n * [(BW_F + BW_I / 2)^{0.75} * (SRW / (BW_F * 0.96))^{0.75} * [BW_F - BW_I] / DOF_T]^{1.097}]$	ASABE (2005)
2	$N_E = (Milk \times 2.303) + (DIM \times 0.159) + (DMI \times C_{CP} \times 70.138) + (BW \times 0.193) - 56.632$	ASABE (2005)
3	$N \text{ excretion} = 30 \text{ (SE=20)} + 0.67 \text{ (SE=0.044)} NI$	Kebreab et al. (2010)
4	$N \text{ feces} = 10 \text{ (SE=9.0)} + 0.28 \text{ (SE=0.023)} NI$	Kebreab et al. (2001)
5	$N \text{ urine} = 20 \text{ (SE=20)} + 0.38 \text{ (SE=0.039)} NI$	Kebreab et al. (2001)
6	$N \text{ milk} = 30 \text{ (SE=10)} + 0.20 \text{ (SE=0.03)} NI$	Kebreab et al. (2001)
7	$N \text{ urine} = 47.8 \text{ (SE=20.1)} + 0.56 \text{ (SE=0.03)} NI - 71.4 \text{ (SE=12.2)} ME$	Kebreab et al. (2010)
8	$N \text{ milk} = 2.04 \text{ (SE=11.7)} + 0.10 \text{ (SE=0.023)} NI + 45.9 \text{ (SE=5.43)} ME$	Kebreab et al. (2010)
9	$N_{E-PH} = [FI_{PH} * (C_{CP}/6.25)] * (1 - N_{RF})$	ASABE (2005)
10	$N_{E-T} = \sum [FI_x * (C_{CP-x}/6.25)] * (1 - N_{RF})$	ASABE (2005)
Eq.	Prediction equations for phosphorus excretion ²	Reference
11	$P_{E-T} = \beta \sum_{x=1}^n (DMI_x * C_{P-x} * DOF_x) - [10.0 * (BW_F - BW_I)] + [5.92 * 10^{-2} * DOF_T * [(BW_F + BW_I / 2)^{0.75} * (SRW / (BW_F * 0.96))^{0.75} * [BW_F - BW_I] / DOF_T]^{1.097}]$	ASABE (2005)
12	$P_E = ((DMI \times 1000) \times C_p) - (Milk \times 0.9)$	ASABE (2005)
13	$P_E = P_i - [0.63 - (0.63 + 0.014)e^{-1.44(P_{ai})}]$	Schulin-Zeuthen et al. (2007)
14	$P_{E-PH} = [FI_{PH} * C_p] * (1 - P_{RF})$	ASABE (2005)
15	$P_{E-T} = \sum [FI_x * C_p] * (1 - P_{RF})$	ASABE (2005)

¹Abbreviations are as follows: N_{E-T} = total N excretion per finished animal (g N/finished animal); DMI is dry matter intake (kg dry feed/animal/day); C_{cp} = concentration of crude protein of total ration (g crude protein/g dry feed); DOF = days on feed for individual ration (days); SRW = standard reference weight for expected final body fat (478 kg is recommended as the SRW value); BW_F = live body weight at finish of feeding period (market weight, kg); BW_I = live body weight at start of feeding period (purchase weight, kg) and subscript x and n are ration number and total number of rations, respectively; N_E = total N excretion per animal per day (g/animal/d); Milk = milk production (kg milk/animal/day); DIM = days in milk (days); BW = average live body weight (kg); NI = N intake (g/d); N_{E-PH} = N excretion per phase (g N/phase); FI_{PH} = feed intake per phase (g feed/phase on wet basis); N_{RF} = retention factor for N (dimensionless, 0.602 for broilers); N_{E-T} = total N excretion per finished animal (g N/finished animal), and SE = standard error.

²Abbreviations are as follows: P_{E-T} = total P excretion per finished animal (g P/finished animal); C_p = concentration of P in total ration (g P/g feed on wet basis); P_E = total P excretion per animal per day (g/animal/d); P_E = P excretion (g P/day); P_i = total P intake; P_{ai} = available P intake; P_{E-PH} = P excretion per phase (g P/phase), P_{RF} = retention factor for P (dimensionless, 0.493 for broilers aged < 32 days and 0.4102 for ages >= 32 days) and P_{E-T} = total P excretion per finished animal (g P/finished animal).

Appendix 2: Groundwater Modeling Links on the Internet

(http://groundwater.ucdavis.edu/Materials/Groundwater_Modeling_Web-Links/)

Groundwater Modeling Software Distribution Sites:

[USGS, Groundwater Modeling Software](#)

[USEPA, Center for Subsurface Modeling Support](#)

[USEPA, Global Earth Observation System of Systems \(GEOSS\)](#)

[Integrated Groundwater Modeling Center, School of Mines, Colorado](#)

[Scientific Software Group \(commercial\)](#)

[Rockware, Inc. \(commercial\)](#)

Interactive Groundwater Modeling on the Web:

[Interactive Groundwater Flow and Transport Models \(University of Illinois\)](#)

Groundwater Modeling Software (public domain/free of charge):

[MODFLOW \(USGS\)](#)—3D finite difference, saturated flow

[MODFLOW Online Guide \(USGS\)](#)—online instructions to MODFLOW modules

[SUTRA \(USGS\)](#)—2D/3D finite element, sat/unsat, variable density flow, transport, heat

[VS2DI \(USGS\)](#)—2D finite difference, unsat flow & transport, heat flow

[MT3DMS \(University of Alabama\)](#)—3D transport model to MODFLOW

[PHAST \(USGS\)](#)—3D multicomponent, reactive transport (constant density flow and temperature)

[RT3D \(Pacific Northwest Laboratory\)](#)—MT3DMS + multispecies reactions

[TOUGH2 \(Lawrence Berkeley National Laboratory\)](#)—3D multiphase/unsat flow & transport

[“Interactive Ground Water \(IGW\)” \(Michigan State University\)](#)—integrated flow & transport

[ParFLOW \(Lawrence Livermore National Laboratory\)](#)—3D large grid flow

[ASMWin \(ETH Zuerich\)](#)—2D finite difference, flow and transport

[STOMP \(Pacific Northwest National Laboratory\)](#)—3D multiphase flow & transport

[IWFEM \(Ca. DWR\)](#)—3D finite element groundwater & surface water flow model

[HYDROGEOSPHERE](#) (integrated surface water/watershed/vadose zone/groundwater flow and process model)

GUIs for MODFLOW and Other Model Codes (commercial):

[ModelMuse \(USGS, public domain\)](#)—GUI for MODFLOW, PHAST

[ModelMate \(USGS, public domain\)](#)—GUI for parameter estimation (UCODE with MODFLOW)

[PMWIN \(public domain\)](#)

[Groundwater Vistas](#)

[Visual Modflow](#)

[Groundwater Modeling System \(GMS\)](#)

[Petrasim](#) (GUI for the TOUGH family of software, 3D modeling)

[EQuIS \(Earthsoft\)](#)—environmental data management software

[Argus ONE](#) (generic GUI—legacy/outdated software)

Groundwater Modeling Software (commercial):

[MODHMS \(Hydrogeologic Inc\)](#)—MODFLOW based watershed model

[MODFLOW-SURFACT](#) (Hydrogeologic Inc.)—MODFLOW w/unsat/multiphase flow

[SWIFT \(GeoTrans\)](#)—3D finite difference, flow, heat, brine, & transport

[FEFlow \(Wasy Ltd.\)](#)—3D finite element flow & transport & watershed & heat

[IGSM \(WRIME, Ca. Dept. Water Resources, U.S. Bureau of Reclamation\)](#)

[MikeSHE \(DHI Water and Environment\)](#)—watershed model w/ MODFLOW

Groundwater Modeling Demonstrations and Reviews:

[Flow and Transport Modeling, Homogeneous and Heterogeneous Aquifers \(IGW\)](#)

[Digital Library, Flow and Transport Modeling \(IGW\)](#)

[Reviews of Groundwater Modeling Software \(IGWMC\)](#)

Other Webpages with Modeling Links:

[Online Site Assessment Tool \(USEPA\)](#)—web calculators, conversions & resources

[Geochemical Modeling and Environmental Modeling Software Links](#)

[California Water and Environmental Modeling Forum](#)

[Vadose Zone Modeling Software Links](#) (UC Davis Groundwater Cooperative Extension Program)

Appendix 3: Vadose Zone Modeling Links on the Internet

(http://groundwater.ucdavis.edu/Materials/Vadose_Zone_Modeling_Web-Links/).

Modeling Software Distribution Sites:

[U.S. Salinity Laboratory: Vadose Zone Modeling Software](#)
[USGS, Groundwater/ Vadose Zone Modeling Software](#)
[USEPA, Center for Subsurface Modeling Support](#)
[USEPA, Software for Environmental Awareness](#)
[International Groundwater Modeling Center, School of Mines, Colorado](#)
[Scientific Software Group \(commercial\)](#)
[Rockware, Inc. \(commercial\)](#)
[Richard Winston's Groundwater Modeling Page](#)
[Hydrology, Hydraulics, Water Quality \(HHWQ\) modeling resources blog](#)

Vadose Zone Modeling Software (public domain/free of charge):

[STANMOD \(USDA\)](#)—1D analytical transport model, inverse modeling, fitting, w/ GUI
[HYDRUS-1D](#)—1D finite element, sat/unsat, flow, transport, heat, w/ GUI
[VLEACH \(USEPA\)](#)—1D finite difference, unsat flow & transport, screening model
[VS2DI \(USGS\)](#)—2D finite difference, unsat flow & transport, heat flow, w/ GUI
[VSAFT2 \(University of Arizona\)](#)—2D finite element, sat/unsat flow, transport w/ GUI
[R-UNSAT \(USGS\)](#)—2D analytical, unsat flow & transport, vapor/VOC, screening model
[Johnson & Ettinger Models \(CA DTSC\)](#)—vapor/VOC movement, screening model
[Johnson & Ettinger Models \(USEPA\)](#)—vapor/VOC movement, web-based screening tool
[SUTRA \(USGS\)](#)—2D/3D finite element, sat/unsat, variable density flow, transport, heat
[TOUGH2 \(Lawrence Berkeley National Laboratory\)](#)—3D multiphase/unsat flow & trans.
[ParFLOW \(Lawrence Livermore National Laboratory\)](#)—3D large grid sat/unsat flow
[STOMP \(Pacific Northwest National Laboratory\)](#)—3D multiphase flow & trans.

Vadose Zone Modeling Software (commercial):

[HYDRUS-2D](#)—2D finite element, sat/unsat, flow, transport, heat, w/ GUI
[HYDRUS-2D/3D](#)—fully 3D finite element, sat/unsat, flow, transport, heat, w/ GUI
[SEVIEW SESOIL](#)—1D, unsat flow & transport, runoff, w/GUI, screening model
[MODHMS \(Hydrogeologic Inc.\)](#)—MODFLOW based watershed model w/ unsat flow
[MODFLOW-SURFACT \(Hydrogeologic Inc.\)](#)—MODFLOW w/unsat/multiphase flow
[FEFlow \(Wasy Ltd.\)](#)—3D finite elements flow & transport & watershed, w/ GUI

Estimating Vadose Zone Properties—Soil Surveys, Databases, Software (public domain):

[California Soil Surveys, Webtool \(UC Davis\)](#)—CA Soil Surveys on the Web
[WebSoilSurvey \(USDA/NRCS\)](#)—Soil Surveys on the Web
[Soil Datamart \(USDA/NRCS\)](#)—Soil Survey documents, NRCS Soil Datamart
[ROSETTA \(USDA\)](#)—Estimates water retention, unsat K functions from basic soil data
[NeuroMultiStep \(UC Davis\)](#)—similar to Rosetta, Central Valley, California, soils, w/ GUI
[UNSODA \(USDA\)](#)—Database and program to estimate unsat properties
[RETC \(USDA\)](#)—A program to estimate unsaturated K from water retention data
[Online Site Assessment Tool \(USEPA\)](#)—web calculators, conversions & resources

Chemicals Databases (public domain):

[Online Site Assessment Tool \(USEPA\)](#)—web calculators, conversions & resources
[Estimating Henry's Law constant \(USEPA\)](#)—estimate Henry's law constant on the web
[Retardation Factor Calculator \(USEPA\)](#)—estimate retardation factor in soils

[NIST Chemistry Webbook](#)—lookup properties by chemical name or CAS number
[PhysPro Database \(Syracuse Research Corp.\)](#)—properties by CAS number
[Chemfinder.com](#)—lookup properties by chemical name or CAS number
[PubChem](#)—lookup properties by chemical name or CAS number
[Chemindustry](#)—lookup properties by chemical name or CAS number

Interesting Websites Related to Vadose Zone Modeling:

[Vadose Zone Modeling](#)—Vadose Zone Journal, Special Section, May 2008
[Unsaturated Zone Hydrology for Scientists and Engineers](#), USGS Online Book
[USEPA Technical Guide to Soil Contamination Screening Tools](#)
[USEPA Support for soil NAPL transport modeling and understanding](#)
[Modeling Software Reviews \(Integrated Groundwater Modeling Center\)](#)

Other Webpages with Modeling Links:

[Geochemical Modeling and Environmental Modeling Software Links](#)
[Groundwater Modeling Software Links](#) (UC Davis Groundwater Cooperative Extension Program)

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