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Benefits of Adopting Environmentally Superior Swine Waste Management Technologies in North Carolina: An Environmental and Economic Assessment

Final Report

Prepared for

C.M. (Mike) Williams, Ph.D.
Animal and Poultry Waste Management Center
North Carolina State University
Designee, Smithfield Foods—Premium Standards Farm Agreement
with the North Carolina Attorney General's Office

Prepared by

RTI International
Research Triangle Park, NC 27709

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Contents

Appendix

Figures

Tables

1 Background and Overview of the Environmental-Economic Benefits Assessment

Brian C. Murray, Ph.D.

The July 2000 agreement between the North Carolina Attorney General's office and Smithfield Foods and its North Carolina subsidiaries (the Agreement) allocates funds to be used for the development of environmentally superior alternatives to the anaerobic lagoon and sprayfield system for treating swine waste.[1] The Agreement calls for the Designee[2] to evaluate the economic feasibility of candidate technologies in determining whether the technologies are acceptable for implementation pursuant to the Agreement. The Designee has deemed that the economic feasibility assessment must evaluate both the costs to the industry and consumers of its products and the benefits to society of the environmental improvements associated with adoption. The cost and industry analyses are being led by a group of researchers from North Carolina State University's Agricultural and Resource Economics (NCSU-ARE) department and will be reported in a separate study to be released in mid-2004.[3] This report focuses on

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[1]The Agreement has since been joined by Premium Standard Farms and subsidiaries, but it is often referred to as "The Smithfield Agreement."

[2]The designee refers to Dr. C.M. (Mike) Williams of the North Carolina State University Animal and Poultry Waste Management Center (APWMC), who has been designated by parties to the Agreement to make the decision on which technologies are considered "environmentally superior."

[3]The co-Principal Investigators of the NCSU-ARE team examining technology costs and industry impacts are Dr. Michael Wohlgenant and Dr. Kelly Zering.

the benefits of adopting alternative technologies, as assessed by an interdisciplinary research team from RTI International.[4]

1.1 THE ROLE OF ENVIRONMENTAL BENEFITS ASSESSMENT

The environmental benefits analysis described in this report informs the technology determination decisions facing the Designee by producing the following types of information:

- identification of pathways by which environmental releases from swine operations affect the environment,
- quantification of environmental improvements in different media (air and water) from changes in swine waste management practices across operations within North Carolina,
- estimation of the monetary value of the quantified improvements in environmental quality, and
- description and qualification of environmental effects that are not easily quantified or monetized.

There are many reasons to identify, quantify, and monetize the benefits of the technology adoption decisions. First among these reasons is that the request for proposals issued for this study required an estimation of the monetized economic benefits of changes in environmental quality resulting from the adoption of alternative technologies.[5] Such an integrated assessment of the environmental and economic benefits provides rich insight into the complex interaction of technological, economic, and natural processes that determine the consequences of the technology change. Additionally, different technologies will likely produce different characteristics of environmental change. For instance, one technology may be very effective at reducing the odor from swine waste management but is less effective at reducing the level of ammonia emissions or the nutrient runoff from the sprayfield operations. Another technology may be very effective at reducing

[4] RTI International is the trade name for Research Triangle Institute, an independent university-affiliated research institute headquartered in Research Triangle Park, NC. For more information on RTI, see the web site (www.rti.org).

[5] RFP item C(5) called for the respondent to "estimate the economic (monetized) benefits to North Carolina households arising from the change in emissions to all environmental media due to the changes in all hog farm operations from current waste management practices to each of the alternative 'environmentally superior' technologies considered for the private cost estimates" (Williams, 2000).

ammonia emissions but is less effective at reducing odor and nutrient runoff. The process of determining the aggregate effects of these changes on environmental indicators in North Carolina and placing these changes in environmental quality in a common metric (dollar benefits) enables direct comparisons between the effectiveness of alternative technologies. For example, the pollutant reduction scenarios presented in Chapter 7 of this report suggest that ammonia emission reductions may have the highest marginal monetized benefit of any potential technology characteristic. In terms of the economic benefits generated, technology alternatives that reduce ammonia the most compare favorably to other technologies, all else equal. As described in Chapters 3 and 6, the benefits of ammonia reduction, however, rely critically on its modeled relationship to the generation of fine particulate matter (PM_{Fine}) and the corresponding health effects.

In terms of the economic benefits generated, technology alternatives that reduce ammonia the most compare favorably to other technologies, all else equal.

It is unlikely that a simple measure of net benefits (monetized benefits—cost) will provide the perfect metric for choosing among alternatives, because many of the benefits (and costs) may be difficult to quantify or monetize. Moreover, those factors that can be quantified and monetized are generally estimated with some degree of uncertainty. As a result, our assessment presents, where possible, ranges of estimates based on both variations in the underlying assumptions used to generate the estimates and the statistical properties of the estimates themselves. Additionally, qualitative assessment of effects is provided when quantification is not possible or when the quantified estimates are so small or uncertain that clear, unambiguous empirical judgments are not possible.

To summarize, estimates of the monetary value of environmental benefits can provide useful information for comparing environmentally superior technology alternatives. They allow one to compare options that produce different arrays of environmental effects to be placed on a comparable basis with each other and with the cost of adoption. Yet the omission of some categories of benefits and the uncertainty with which some of the included elements are estimated suggest that monetized environmental benefits should be employed with the normal cautions associated with any quantitative assessment used for decision-making purposes. These estimates should be combined with other scientific, engineering, and economic considerations in making a fully informed determination of what constitutes an environmentally superior technology alternative under the provisions of the Agreement.

1.2 SCOPE OF THE STUDY

The scope of the project is defined by the function it serves relative to other research components funded by the Agreement, the factors that are included and excluded from the analysis, and the population of swine operations and households to which it applies. Each of these scope dimensions is discussed in turn below.

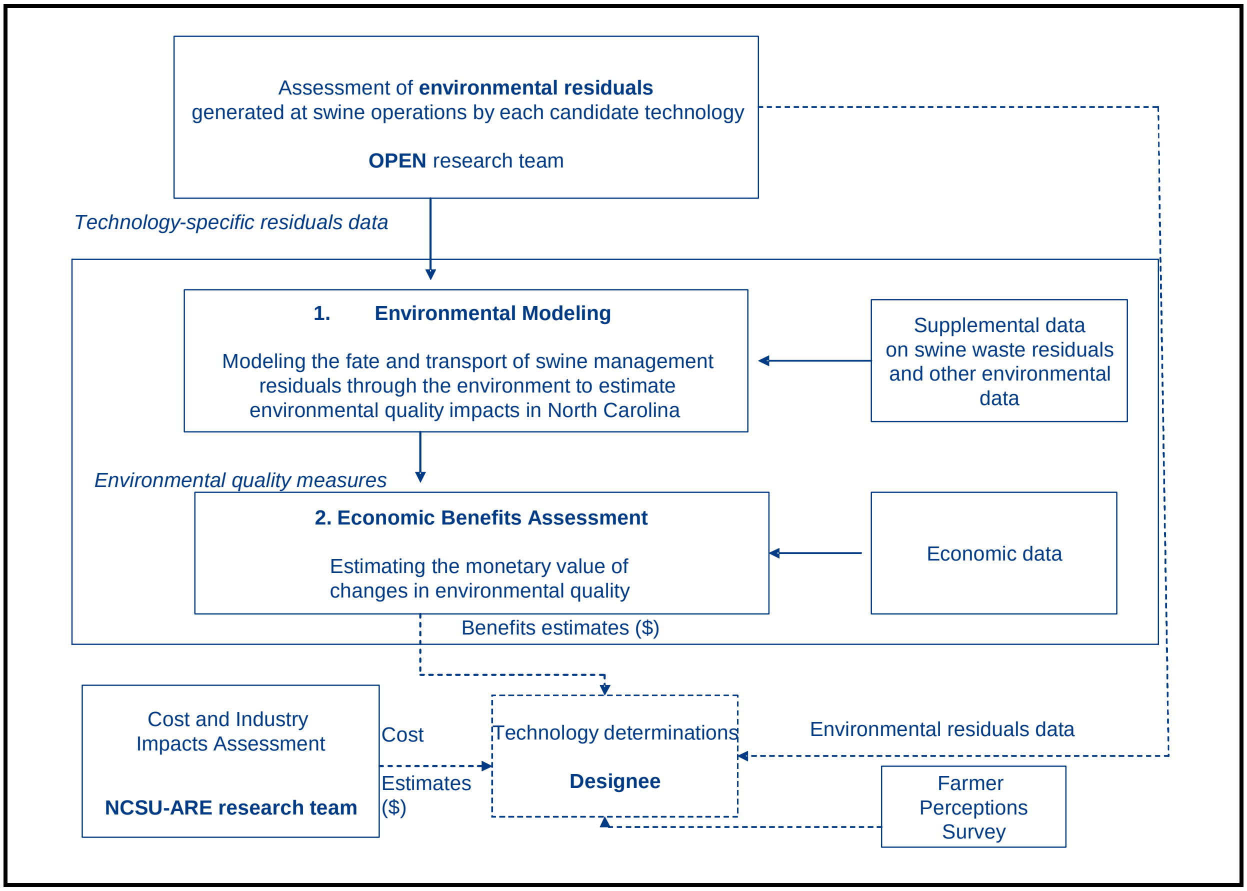
1.2.1 Distinction from Other Assessments being Performed under the Agreement

Figure 1-1 places the environmental benefits assessment in the context of the overall assessment of the Agreement's candidate technologies. The process is driven initially by environmental sampling of the different technologies' pilot site operations being conducted by a team of researchers from NCSU, University of North Carolina-Chapel Hill (UNC-CH), and Duke University. That effort is referred to as the OPEN (**O**dor, **P**athogens, **E**missions of **N**itrogen) team study.[6] The OPEN team is conducting repeated sampling of emissions and other environmental releases at the locations where each technology is being pilot tested. The OPEN team data are supplemented by data on nitrogen and phosphorous runoff from sprayfield operations derived from data from the technology pilot sites. When completed, these studies will collectively provide the first direct estimates of the effectiveness of each technology in reducing key residual loadings to the environment.

The research study described herein focuses on how the technology-specific changes in swine-related environmental residuals, as estimated by the OPEN team and supplemented by the RTI research team, translate into economic benefits of the corresponding environmental improvements. The two main components of the study are environmental modeling and economic benefits assessment. The results of the economic benefits assessment can be used along with the cost study being conducted

[6]The OPEN team co-Principal Investigators are Dr. Viney Aneja, NCSU (emissions of nitrogen); Dr. Mark Sobsey, UNC-CH (pathogens); and Dr. Susan Schiffman, Duke (odor).

Figure 1-1. Context of the Benefits Assessment Study



The environmental modeling stage of the analysis links the environmental residual changes that are projected to occur at swine operations adopting the technologies within North Carolina to measurable changes in environmental quality (air and water).

by the NCSU-ARE research team, the environmental residuals data, and other relevant information to help the Designee determine environmentally superior technologies.

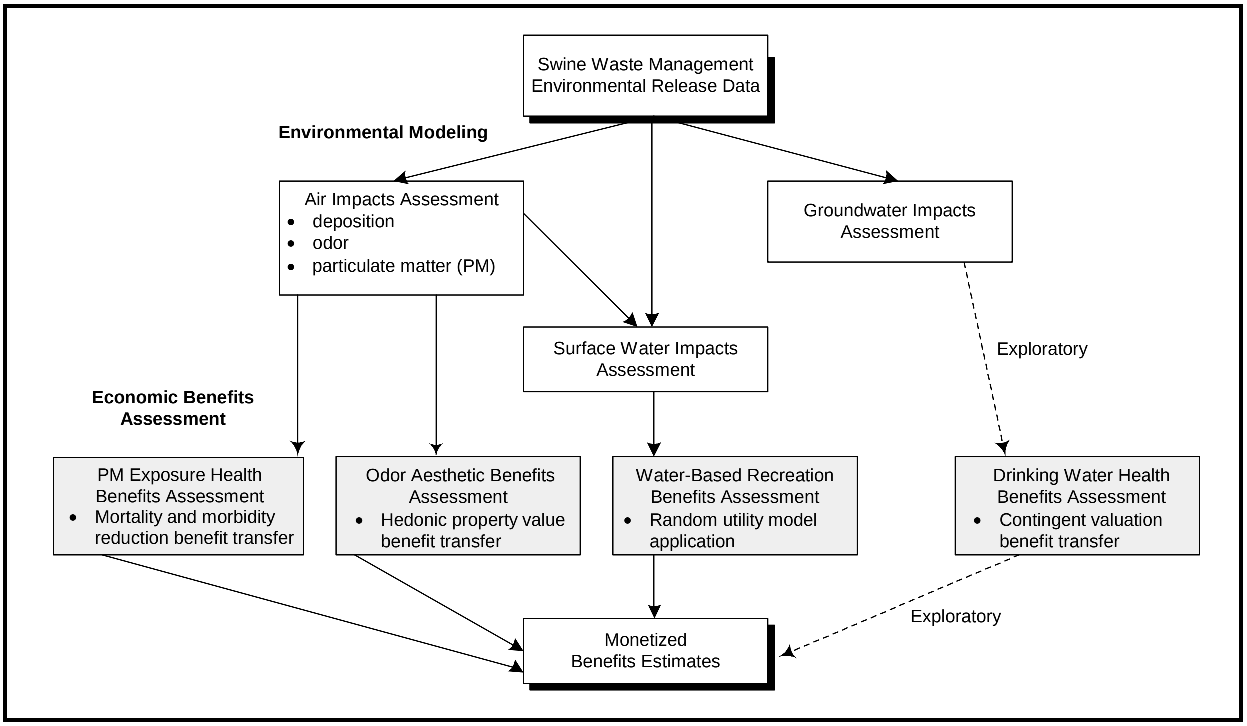
1.2.2 Environmental and Economic Dimensions of Analysis

Figure 1-2 provides more detail on the specific activities covered under the two main components of the benefits study: environmental modeling and economic benefits assessment.

Environmental Modeling

The *environmental modeling stage* of the analysis links the environmental residual changes that are projected to occur at swine operations adopting the technologies within North Carolina to measurable changes in environmental quality (air and water). In essence, this involves the following:

Figure 1-2. Overview of Benefits Assessment Approach and Linkages to Environmental Modeling



1. locating each potential source of environmental release within the state using a database locating each swine operation in the state layered upon a geographic information system (GIS);
2. modifying the environmental releases, as indicated by the technology adopted and the results of the OPEN team evaluations; and
3. simulating the effect of the identified changes in swine-related residuals on air and water quality measures within North Carolina, using a combination of monitoring data, process models, and statistical estimation methods customized to the conditions of eastern North Carolina.

Several studies (e.g., North Carolina Cooperative Extension Service, 1995; Aneja, Chauhan, and Walker, 2000; Cochran, Rudek, and Whittle, 2000; Palmquist, Roka, and Vukina, 1997) have identified the following potential environmental consequences attributed to the anaerobic lagoon and sprayfield system for swine waste management:

- air and water quality degradation caused by the emission, dispersion, and deposition of volatized ammonia (NH_3) and other gases into the atmosphere;

- unpleasant odors emanating from the lagoons and fields;
- degraded surface water quality from runoff of unabsorbed nitrogen from the fields;
- groundwater contamination through seepage of the waste from the lagoon and fields; and
- introduction of health-harming pathogens.

Swine residuals, particularly ammonia (NH_3) emissions, can affect air quality in ways that lead to nitrogen being deposited on land and in water bodies in areas surrounding the swine operations. This land deposition can ultimately reach water bodies through runoff and thereby combine with direct nitrogen deposition to water bodies and degrade water quality. Sprayfield operations under the current lagoon and sprayfield system can be the source of excess nutrients (nitrogen and phosphorous), which can run off and degrade surface water quality. And the combination of nitrogen deposition and runoff could leach into groundwater and affect the quality of drinking water in surrounding areas. Note that the environmental quality issues referenced above are potential effects. The objective of the environmental modeling component of the study is to estimate the empirical magnitude and distribution of these effects within the region of North Carolina directly affected by swine residuals. The results of that analysis are presented in the chapters that follow.

Another air quality factor to consider is the effect of odor on surrounding human populations. The odor effects for each of the candidate technologies are being directly estimated by the OPEN team. No further environmental modeling was employed to aggregate these effects across North Carolina (in contrast, say, to the fate and transport modeling of ammonia emissions, nitrogen, and phosphorous runoff just described). However, the economic analysis described below does develop aggregate measures of the benefits of odor improvements based on estimated point-specific reductions in odor straight from the OPEN team estimates.

The modeling of pathogen effects remains largely outside this benefits assessment study. At the time this study was proposed, request for proposal (RFP) respondents were instructed to base their methods on the current state of the science rather than to engage in research to develop new science. At that time, little was known about the fate and transport and health consequences of specific pathogens in the environment. Therefore, readers of this study should consider any pathogen reduction results found in the OPEN team research as (nonmonetized) supplemental benefits to those monetized in this study.

Economic Benefits Assessment

The economic benefits assessment stage links the change in environmental quality estimated in the environmental modeling phase of the study with the economic value to those affected by the environmental quality improvements.

The *economic benefits assessment* stage links the change in environmental quality estimated in the environmental modeling phase of the study with the economic value to those affected by the environmental quality improvements. The basic steps of analysis involve the following:

1. identifying the economic services affected by the changes in air and water quality for the region of interest,
2. determining the appropriate measures of economic value for these changes in environmental quality,
3. applying the economic value to the quantified projected changes in environmental quality due to changes in swine waste management (from the environmental modeling stage), and
4. aggregating measures across all projected effects to estimate total economic values of a given swine waste reduction scenario.

Environmental quality affects human welfare in many ways, thereby imparting economic value. It is beyond the current state of the science or the resources available for this study to estimate economic values for all potential sources of benefits. However, based to a large degree on the results of the environmental modeling outlined above, this study measures benefits in the following areas:

- ➤ recreational benefits from improvements in water quality,
- ➤ aesthetic benefits from reduction in odor,
- ➤ health benefits from reduced exposure to certain air pollutants, and
- ➤ health benefits from drinking water improvement.

Recreational benefits derive from the fact that people's water-based recreation choices are based in part on the water quality at the site of interest. All else equal, people prefer to recreate in areas with higher water quality as this may improve both the aesthetics and other attributes such as the catch rate of fish. As demonstrated in Chapter 6, we employ an economic model of recreation demand

(Phaneuf, 2002) to estimate how water quality affects recreation choices and the economic value of those visits.

Aesthetic benefits from odor reductions are primarily expected to benefit residents living in near hog farms. People choose where to live based on a wide range of characteristics of the housing such as the number of bedrooms, size, age, acreage and other locational attributes such as the quality of local schools, proximity of parks, shopping opportunities, and other amenities. This represents the hedonic model of consumer choice first introduced by Rosen (1974). Numerous studies of housing demand have shown that environmental attributes can affect the demand for and price of housing, with negative disamenities such as landfills leading to value reductions (see Smith and Huang [1995] for a review). Of particular relevance for this study is a hedonic analysis by Palmquist, Roka, and Vukina (1997), which found that proximity to swine operations in North Carolina (particularly larger ones) can reduce residential property values. Similar effects of swine operations on property value have been found in other areas of the country (Ready and Abdalla, 2003; Herriges, Secchi, and Babcock, 2003), and odors from the swine operations are asserted to be a primary reason. Therefore, these hedonic findings provide a basis for estimating odor reduction benefits for local residents.

Some of the health benefits associated with alternative waste management techniques may result from improvements in air quality. As indicated above, swine operations emit ammonia (NH_3) which can deposit to land and water surfaces in the surrounding areas or be transformed to ammonium (NH_4) based fine particulates. Human exposure to fine particulates in the atmosphere is associated with a number of potential health effects, as described in Chapters 3 and 6. Reduction in ammonia emissions from swine operations can reduce the generation of fine particulates and the corresponding health effects, thereby benefiting those who might otherwise incur the health effects and associated economic costs.

Other health benefits may accrue through improvements in drinking water quality. Swine operations in North Carolina manage wastes in part by spraying nutrient-rich (nitrogen and phosphorous) water from the holding lagoons into surrounding fields for fertilization. If some of these nutrients are not taken up in the plants, they can be exported from the site and work their way into surface and groundwater systems. The associated concern with groundwater is that nitrogen loadings may contribute to elevated nitrate concentrations in drinking water, which, if high enough, may be considered unhealthy. In several economic studies, people have demonstrated a preference for reductions in nitrate concentrations in their drinking water (see Bergstrom, Boyle, and Poe [2001]).

Theoretically Consistent Economic Values for Environmental Improvements[7]

Economic value refers to the notion that improvements in the environment enhance the well-being of people in ways that are, in principle, economically measurable.

Step 2, at the beginning of this section, identifies the need to develop "appropriate" measures of economic value. *Economic value* refers to the notion that improvements in the environment enhance the well-being of people in ways that are, in principle, economically measurable. The economic measures should reflect how much people are *willing to pay* for given improvements in environmental quality or how much they are *willing to accept* for reductions in environmental quality. The idea is to evaluate how people do or would choose between different situations that vary by levels of environmental quality and monetary compensation.

In some cases, the referenced economic value can be estimated by observing actual behavioral responses to changes in environmental quality (e.g., recreational site choices, housing purchases, job selection). These are referred to as *revealed preference* (RP) approaches. RP approaches are, in some sense, ideal because they are based on real, rather than hypothesized, behavioral responses. However, RP approaches have some limitations, as summarized by Henscher, Louviere, and Swait (1999):

- difficulty in valuing new attributes or features for which there is no RP history, and/or for which one cannot safely forecast by analogy to existing products or services;
- key RP explanatory variables may exhibit little variability or applicability to the case being evaluated; and
- often RP data fail to satisfy assumptions underpinning economic theory and/or contain statistical irregularities.

An alternative to RP approaches for economic valuation is the use of *stated preference* (SP) methods, in which individuals are asked to make trade-offs between changes in environmental quality and monetary compensation. Examples of SP methods are the

[7] Dr. F. Reed Johnson provided valuable comments in this section.

contingent valuation method and conjoint analysis. SP methods overcome some of the problems identified with RP above by observing choices from statistically defined choice sets that systematically vary all the current and future attributes of interest. In contrast to RP data, collecting SP data also may require fewer resources. However, experimental control over the SP choice context may come at the expense of potential *hypothetical* bias. Although SP data reflect trade-offs considered during the decision-making process, there often is little basis for determining to what extent SP subjects would actually do what they say they will do. Some studies have observed discrepancies between stated preferences and subsequent observed behavior (Morwitz, Steckel, and Gupta, 1997). Although SP approaches are somewhat controversial, economists have used them for more than 30 years, and the methods have undergone substantial refinement in the last decade to address problems such as hypothetical response bias, the recognition of budget constraints, and other factors to impose more realism on the economic choices. The analysis contained in this report relies heavily on RP measures (i.e., through housing choices, recreational choices, and labor choices) but also includes SP measures when RP measures are either unavailable or inappropriate.

This study examines these benefits and uses a combination of original research and results from other studies to develop empirically based, customized estimates of the benefits of reducing environmental residuals from swine operations in North Carolina.

This study examines these benefits and uses a combination of original research and results from other studies to develop empirically based, customized estimates of the benefits of reducing environmental residuals from swine operations in North Carolina. The methods used to estimate these benefits are based on approaches that satisfy peer-review standards employed by regulatory agencies of the federal government and are standard operating practices in the environmental and natural resource economics profession. However, the monetized estimates included in this report are solely for the purposes of informing the technology comparisons. Although these estimates have been developed using rigorous methods applied to the best available data and can therefore provide important information on the relative benefits of technology alternatives, they are not intended to be, nor should they be interpreted as, complete and precise monetized estimates of the total benefits of reducing swine-related environmental residuals in the state of North Carolina. In particular, these estimates should not be construed as a monetized natural resource damage assessment

(NRDA) of current practices.[8] NRDA is typically applied as an intensive study of a specific event at a specific location. That does not match the scope, intention, or budget of this study, which is not tied to any specific event and is broadly aggregated for a large area in eastern North Carolina. Therefore, the monetized estimates should be interpreted as rough indicators of benefits, rather than as precise dollar values.

1.2.3 Affected Swine Operations

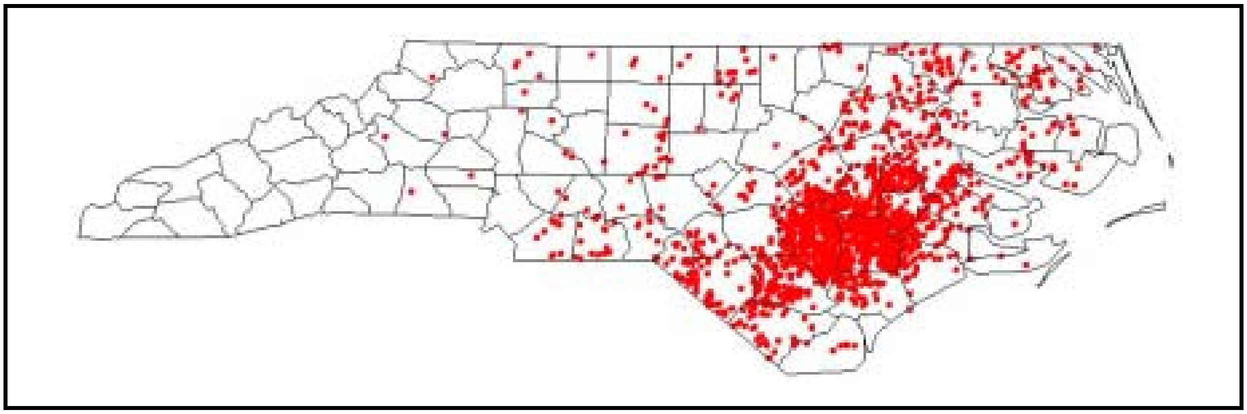
This analysis uses the population of approximately 2,300 permitted swine operations identified by the North Carolina Division of Water Quality (DWQ) as the universe of potentially affected facilities.

The Agreement specifically requires the adoption of environmentally superior waste treatment technologies on swine operations owned by Smithfield Foods and Premium Standard Farms. An independent group of North Carolina swine operators called Front Line Farmers has also agreed to adopt the approved technologies on a voluntary basis. Together these groups constitute a minority of the swine farms currently operating in North Carolina. However, the Designee requested that this analysis evaluate the potential benefits if essentially all swine operations in the state were to adopt the technologies. Therefore, this analysis uses the population of approximately 2,300 permitted swine operations identified by the North Carolina Division of Water Quality (DWQ) as the universe of potentially affected facilities. The environmental benefits assessment model developed for this project and described later in the report, however, does allow for the benefits assessment to be confined to subsets of the universe of North Carolina swine operations, as defined by facility type, size, ownership and watershed location.

Figure 1-3 shows the location of all swine operations within North Carolina as tracked by the DWQ database referenced above. The concentration of hog producers in the eastern part of the state is evident from the figure. Duplin and Sampson counties are the two largest hog-producing counties in the country (Pork Facts, 2000), and four of the top seven producing counties are in North Carolina. Consequently, the environmental and economic impacts are

[8] In an April 30, 2002 meeting of parties to the Agreement and the RTI economics team, James Gulik, Senior Deputy Attorney General for the Environment Division for the North Carolina Attorney General's Office indicated that any use of the results of this study for purposes other than to inform the technology assessment decision of the Designee would be considered a misuse of the results.

Figure 1-3. Locations of North Carolina's Hog Producers



Source: North Carolina Department of Environment and Natural Resources (NCDENR). 2002. North Carolina Division of Water Quality 1997 Survey of Animal Feeding Operations—Database. Raleigh, NC: NCDENR.

concentrated in this region as well. Reflecting this geographic emphasis, the environmental fate and transport models and economic valuation models are customized to the conditions in eastern North Carolina.

1.2.4 Residual Reduction Scenario Evaluation

At this juncture, the Designee has identified 18 swine waste treatment systems that are candidates for designation as environmentally superior. Those technologies are listed in Table 1-1. This report was originally intended to serve as a comparative analysis of the environmental benefits of each of the candidate technologies. However, at the time of this report, these systems are in various states of completion, ranging from full steady state operation to recent completion of construction to construction not yet started. As a result, the availability of complete environmental residuals data for all technologies is approximately 2 years away.[9] When the expected delay in technology-specific data became apparent, the focus of the research effort shifted course in response. After we consulted with the Designee, the environmental benefits assessment project changed from a technology-by-technology evaluation to the development of an automated integrated benefits modeling system that is flexible enough to evaluate the technology-specific environmental residuals inputs when they do become available in the future. In the meantime, this

[9]The projected completion date for environmental residuals monitoring data for all 18 systems is middle to late 2005.

Table 1-1. Candidate Technologies and Evaluation Setting

Technology	Evaluation Type[a]
1. In-ground ambient temperature anaerobic digester/energy recovery/greenhouse vegetable production system	F
2. High-temperature thermophilic anaerobic digester (TAnD) energy recovery system	F
3. Solids separation/constructed wetlands system	F
4. Sequencing batch reactor (SBR) system	F
5. Upflow biofiltration system	F
6. Solids separation/nitrification-denitrification/soluble phosphorus removal/solids processing system	F
7. Belt manure removal and gasification system to thermally convert dry manure to a combustible gas stream for liquid fuel recovery	P
8. Ultrasonic plasma resonator system	P
9. Manure solids conversion to insect biomass (black soldier fly larvae) for value-added processing into animal feed protein meal and oil system	P
10. Solids separation/reciprocating water technology system	S1
11. Micro-turbine cogeneration system for energy recovery	S2
12. Belt system for manure removal	P
13. High-rate second generation totally enclosed Bion system for manure slurry treatment and biosolids recovery	F
14. Combined in-ground ambient digester with permeable cover/aerobic blanket—BioKinetic aeration process for nitrification-denitrification/in-ground mesophilic anaerobic digester system (this project represents three farm sites)	S2
15. Dewatering/drying/desalinization system	P
16. Solids separation/gasification for energy and ash recovery centralized system (this project represents three farm sites)	S1
17. High solids high-temperature anaerobic digester system	T
18. Solids separation/mesophilic anaerobic digestion/membrane filtration—reverse osmosis system	F

[a]Key:
F = Single farm-scale system
P = Pilot-scale unit process, evaluated in laboratory or field-site setting
S1 = Technologies 10 and 16 are combined as one complete system, evaluated at three farms
S2 = Technologies 11 and 14 are combined as one complete system, evaluated at three farms
T = Offsite waste treatment process

tool can be used to evaluate scenarios defining ranges of possible outcomes associated with adoption of environmentally superior technologies. This scenario analysis is performed and described in Chapter 7.

With this as background, the main objectives of the report are to

- describe the analytical approach, building on and substantially expanding the methods described in our 2002 methodology document (RTI, NCSU-ARE, 2002);
- describe the scientific foundation, logical structure, and data underlying the analytical models used to quantify the environmental quality impacts and monetized benefits of swine-related changes in residuals;
- in the absence of actual technology-specific data at this time, evaluate a range of possible residual reduction scenarios to estimate and report the corresponding magnitude of changes in swine-related environmental residuals on water and air quality measures, within the study area of eastern North Carolina;
- estimate the monetized economic benefits of the changes in environmental quality;
- describe how the integrated assessment tool developed for this project can be used to estimate environmental quality benefits when technology-specific data and Designee-specified adoption scenarios become available in the future; and
- discuss and address analytical uncertainties, gaps in the analysis, future research needs, and other caveats in interpreting the results of this analysis.

1.3 ORGANIZATION OF THE REPORT

The remainder of this report is organized along the lines of the research activities and modeling components referenced above. Chapter 2 describes the methods, data, and results of the ammonia dispersion and deposition analysis. This process draws on a GIS-based characterization of all swine facilities in the state and unitized deposition factors to simulate the effects of farm-level changes in ammonia emissions on aggregate deposition levels. Chapter 3 pivots off of the ammonia gas results in Chapter 2 by simulating the interaction of ammonia gas that is not wet or dry deposited with other atmospheric compounds to form aerosol particulates (ammonium (NH_4^+) sulfate, ammonium bisulfate, ammonium nitrate, and ammonium chloride). Chapter 4 presents the modeling approach, data, results, and validation for the surface water quality modeling component of the study. That chapter quantifies the impact of swine and nonswine sources on the nitrogen and phosphorous loadings in the region's surface water stream network. Chapter 5 describes the empirical analysis of groundwater well sample data used to estimate the magnitude of the effect of swine operations on groundwater nitrate concentration levels in the study region. The analysis employed different data sets, model specifications, and estimation methods to examine these effects. Chapter 6 presents the economic methods, data, and models used to estimate the monetized value of changes in environmental quality quantified by the environmental quality models of Chapters 2 through 5. These estimates are evaluated for a range of possible residual reduction scenarios that could be engendered by adopting environmentally superior waste management alternatives. The report concludes with a chapter describing the development of an integrated assessment tool that can be used to evaluate a wide range of farm- and technology-specific residuals reduction scenarios.

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2 Atmospheric Dispersion and Deposition of Ammonia Gas

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In this chapter, we present our approach for and results from modeling the dispersion and deposition of ammonia (NH_3) air emissions from swine facilities onto a designated study area. To estimate ammonia emissions, dispersion, and deposition under baseline conditions and with simulated reductions in emissions (addressed in Chapter 7), we used existing emission factors; designed a variety of model units (e.g., lagoons) to accommodate differing capacities and types of swine operations; and used an existing, proven dispersion-deposition model. The estimated deposition loading is integral to RTI's evaluation of the expected change in surface water nitrogen and phosphorus concentration and loadings associated with swine waste management technologies under study in eastern North Carolina (see Chapter 4).

This chapter presents RTI's approach to estimating ammonia emissions, the method used to model dispersion and deposition of ammonia, results of baseline modeling, quality assurance measures taken, and uncertainties associated with this component of the overall environmental analysis.

2.1 AMMONIA AIR EMISSIONS ESTIMATION

Measuring the change in ammonia emissions attained by implementing alternative waste management technologies requires establishing a standardized baseline. "Baseline" represents ammonia emissions and subsequent ammonia deposition from waste management processes and practices currently in use at each facility in the study area. To establish the baseline, a profile of each concentrated animal feeding operation (CAFO) was obtained using the North Carolina database of approximately 2,295 swine operations. We assigned each of the 2,295 operations to one of 12 model CAFOs (i.e., a CAFO representing one of three meteorological regions in North Carolina with one of four model acreages). RTI modeled each of the 12 model CAFOs to predict atmospheric transport and deposition, using the U.S. Environmental Protection Agency's (EPA) Industrial Source Complex-Short-Term 3 (ISCST3) model, which assumed an emission rate of 1 mg NH_3/square meter/second from the CAFO property (commonly referred to as a "unitized rate"). Based on the facilities' type of operations, each CAFO was further characterized by assigning it to one of five growth stages (e.g., wean to feed, wean to finish). This yielded a choice of 60 settings (12 model CAFOs x 5 growth stages) to which each of the study's CAFOs could be assigned and a unique annual average emission factor could be applied. In the end, the deposition (a.k.a. "loading") over a 50 km radius for each CAFO is represented as

$$NH_3 \text{ Loading} = \text{Unitized Deposition Rate} \times \text{Emission Factor} \times \text{CAFO Capacity.} \quad (2.1)$$

We expressed baseline ammonia emissions input data, in terms of steady state live weight (SSLW). (This approach was consistent with the NCSU-ARE research team estimating the cost of technology adoption.)

The following sections describe how we collected and derived data for modeling inputs and developed model CAFOs.

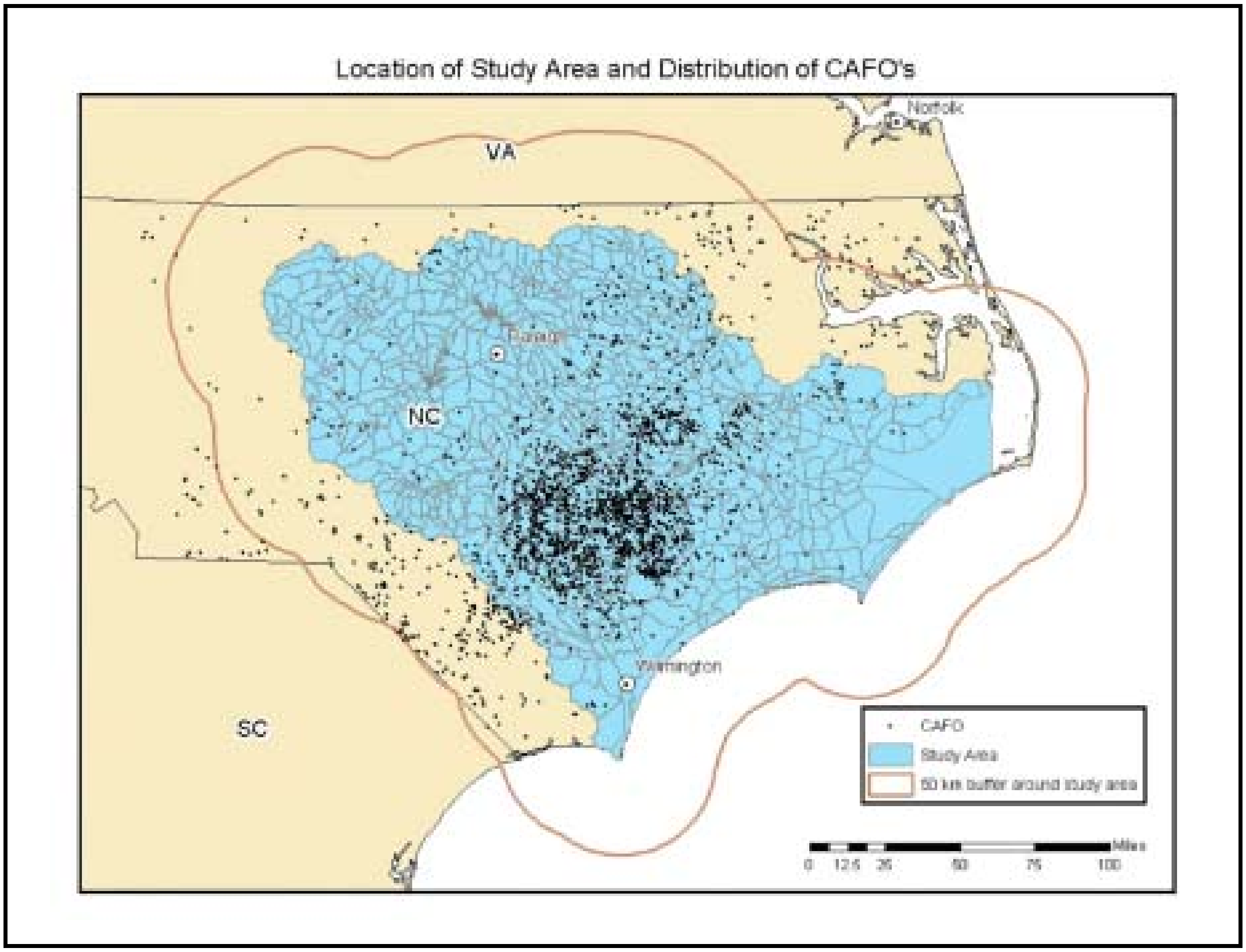
2.1.1 Data Collection

RTI collected data and developed the input parameters needed to estimate ammonia dispersion and deposition from swine CAFOs in the study area. The primary data sources are discussed in more detail in this section. We obtained data from existing databases, other research projects, the literature, and other sources as warranted. RTI used geographic information system (GIS) tools to create the required data inputs for the dispersion and deposition characterization and for the water quality modeling.

North Carolina CAFO Inventory/Study Area

The study area for the air dispersion/deposition analysis is defined primarily by the water quality study area: Tar-Pamlico, Neuse, Cape Fear, White Oak, and New River basins. This study area contains 2,295 CAFOs as identified in North Carolina's inventory of swine operations (NCDENR, 2002) (see Figure 2-1).

Figure 2-1. Study Area and Distribution of CAFOs



Emission Factors Selection and Derivation by Growth Stage

Modeling the impacts of ammonia deposition from CAFOs requires a measure of ammonia emissions from CAFO sources. This section addresses the ammonia data available from CAFO monitoring studies and the method of selecting emission factors from the data set. We present recommended emission factors along with associated uncertainties.

Operation Type. Swine operations were organized into the five categories designated for the overall economic analysis:

- farrow to wean,
- wean to feed,
- farrow to feed,
- farrow to finish, and
- feed to finish.

Emission Sources. To the extent possible, RTI selected emission factors specific to these animal feeding operations for the three primary emission sources:

- animal confinement housing,
- lagoon, and
- sprayfield.

Data Collection. RTI reviewed available data in the literature to determine appropriate emission factors for these sources by operation category. A recent EPA publication (EPA/ORD, 2002b), which provides a comprehensive compilation of available emission data, was used as the predominant literature for this analysis. Emission data were available for the confinement housing for different types of operations; however, only feed-to-finish or farrow-to-finish data were available for lagoons. Nearly all "emission factors" for sprayfields in the literature were developed using a mass balance approach based on the ammonia-nitrogen of the excreted waste, the emissions from the confinement housing and lagoon, and an assumed fraction of ammonia released during spray application and from the unabsorbed sprayfield residue after application. As such, the reported "emission factors" for the sprayfields are dependent on the emission factors assumed for the confinement housing and the lagoon. Because each sprayfield emission factor in the literature is mass balance derived from a unique CAFO's housing and lagoon emissions, RTI did not use the factors reported in the literature for sprayfields (on a per-pig basis) directly. Instead, we used the same mass balance approach applied in the literature for each of the five operating categories' housing and lagoon emission factors. To apply this mass balance approach, data were needed regarding the nitrogen generation rates for each of the five swine operating categories.

Data Collected. *Confinement Housing Emission Factors.* The emission factors collected for confinement housing from the EPA (2002) literature review are summarized in Table 2-1. The units used to express the emission factors vary considerably across the literature. In attempts to put the emission factors in common units of measure, emission factors reported on a per-pig basis were converted to a per-mass basis (e.g., per 500 kg live weight and per 150 lb "standard" pig). For the farrow-to-wean operations, an average pig weight of 400 lbs was used (presumably counting sows only); for finishing operations, an average pig weight of 150 lbs was used (EPA, 2001).

From the analysis of the data in Table 2-1, RTI derived emission factors for the five animal operation types as shown in Table 2-2. These emission factors are treated as annual averages.

Comparison of emission factors derived from the literature to monitoring results at the baseline operations. Over the course of this research, the OPEN Team monitored ammonia emissions for multiple seasons at two sites operating traditional confinement houses, lagoons, and sprayfields (referred to as "baseline"). However, no data were available for comparison in the Settlement Agreement's Third Annual Progress Report (http://www.cals.ncsu.edu/waste_mgt/apwmc.htm) on ammonia emission monitoring results for confinement housing at the two baseline operations of Stokes and Moore operations.

Lagoon Emission Factors. The best data available for lagoon emission factors were for finishing operations, where two independent research teams (Aneja, Chauhan, and Walker, 2000; Harper and Sharpe, 1998) conducted lagoon emission measurements over the course of a year. One of these studies (Harper and Sharpe, 1998) also reported measurements made for a lagoon at farrow-to-wean operations. No data were available for the wean-to-feed operations. The lagoon emission factors are summarized in Table 2-3.

Table 2-1. Confinement Housing Emission Factors

Researcher	Animal Type	House Type	mg NH_3/ hr/500 kg l.w.	Mg NH_3/ hr/pig	kg NH_3/ 500 kg l.w./yr	kg NH_3/ pig/yr	g NH_3/pig/ day
Farrow to Wean							
Groot Koerkamp et al., 1998	Sows, litter	England	**744**	**303**		2.65	
		Netherlands					
		Denmark					
		Germany	**3,248**	**1,298**		11.37	
		Mean					
	Sows, slats	England	**1,049**	**503**		4.41	
		Netherlands	**1,282**	**535**		4.69	
		Denmark	**1,701**	**730**		6.39	
		Germany	**1,212**	**325**		2.85	
Steenvoorden et al., 1999	Gestating sows—standard individual confinement		1,318[a]	479[a]		**4.2**	
	Gestating sows—narrow manure gutter with metal slatted floor		753[a]	274[a]		**2.4**	
	Farrowing sows—standard fully slatted floor		2,606[a]	947[a]		8.3	
	Farrowing sows—shallow manure pit with gutter		1,256[a]	457[a]		4	
Mean (std. dev.)			**1,517** 812			5.0764 2.6926	
Median			1,234			4.41	
Mean without 11.37 (std. dev.)			1,311 565			4.447 1.793	
Farrows Without Sows (Wean to Feed)							
Groot Koerkamp et al., 1998	Farrows, slats	England	**1,047**	**26**	2.5	0.22	
		Netherlands	**786**	**27**	2.72	0.24	
		Denmark	**1,562**	**46**	4.54	0.4	
		Germany	**649**	**22**	2.16	0.19	
Mean		Mean	**1,011**			0.26	

(continued)

Table 2-1. Confinement Housing Emission Factors (continued)

Researcher	Animal Type	House Type	mg NH_3/ hr/500 kg l.w.	mg NH_3/ hr/pig	kg NH_3/ 500 kg l.w./yr	kg NH_3/ pig/yr	g NH_3/pig/ day
Finishing (Farrow to Finish, Feed to Finish)							
Groot Koerkamp et al., 1998	Finishers, litter	England	**1,429**	**108**		0.95	
		Netherlands					
		Denmark	**3,751**	**394**		3.45	
		Germany					
		Mean				2.2	
	Finishers, slats	England	**2,592**	**185**		1.62	
		Netherlands	**2,076**	**385**		3.37	
		Denmark	**2,568**	**319**		2.79	
		Germany	**2,398**	**308**		2.7	
		Mean				2.62	
Demmers et al., 1999	Finishing		**1,980**	**270**	**47**	2.36	
Steenvoorden et al., 1999	Finishing, 50% slatted		2,093[b]	285[b]		**2.5**	
	Finishing, separate manure gutters		1,507[b]	205[b]		**1.8**	
	Finishing, slopping floors		837[b]	114[b]		**1**	
Harris and Thompson, 1998	Finishing—Nov. 1997 ventilated		2,294[b]	313[b]		2.74	**7.5**
	Finishing—Jan 1998 ventilated		3,968[b]	541[b]		4.74	**13**
	Finishing—May 1998 ventilated		2813[b]	384[b]		3.36	**9.2**
Harris, 2001	Farrow to finish—summer		4,027[b]	549[b]		**4.81**	
	Farrow to finish—annual		3,089[b]	421[b]		**3.69**	
1985 Emission Inventory, annual (Warn, Zelmanowitz, and Saeger, 1990, EPA-600/7-90-014—NAPAP emission inventory)			1,632[b]	223[b]		**1.95**	

(continued)

Table 2-1. Confinement Housing Emission Factors (continued)

Researcher	Animal Type	House Type	mg NH_3/ hr/500 kg l.w.	mg NH_3/ hr/pig	kg NH_3/ 500 kg l.w./yr	kg NH_3/ pig/yr	g NH_3/pig/ day
Asman, 1992; annual (Euro.)			2,110[b]	288[b]		**2.52**	
Battye et al., 1994; annual			3,357[b]	458[b]		**4.01**	
Van der Hoek (European Community), 1998; annual			3,089[b]	421[b]		**3.69**	
Harris, Shores, and Jones, 2001; annual			2,244[b]	306[b]		**2.68**	
Mean			**2,493**				
Mean—U.S. only—finishing			**2,577**				

[a]Converted to a mass basis, assuming an average of 400 lbs/pig (i.e., primarily sows).
[b]Converted to a mass basis, assuming an average of 150 lbs/pig.
Note: Bold data are data as reported in the literature.

Table 2-2. Recommended Annual Average Emissions Factors for Confinement Housing

Animal Type	Emission Factor (mg NH_3/hr/500 kg l.w.)	Emission Factor (kg NH_3/std.pig/yr)[a]
Farrow to wean	1,517	1.81
Wean to feed	1,011	1.21
Farrow to feed	1,517	1.81
Farrow to finish	2,493	2.98
Feed to finish	2,493	2.98

[a]Standard pig = 150 lbs

Table 2-3. Annual Average Lagoon Emission Factors

Researcher	Animal Type	kg NH_3/ operations/ day	kg NH_3/ pig/day	kg NH_3/ pig/yr
Farrow to wean				
Harper and Sharpe (1998), North Carolina Farm 20	Farrow to wean	**14.8**	0.0027[a]	0.97[a]
Farrows without sows (wean to feed)				
No data				
Finishing (farrow to finish, feed to finish)				
Aneja, Chauhan, and Walker (2000) North Carolina Farm 10	Farrow to finish	**66.8**	0.0060[b]	2.19[b]
Harper and Sharpe (1998) North Carolina Farm 10	Farrow to finish	**31.3**	0.0028[b]	1.03[b]
Mean		49	0.0044	**1.61**

[a]Emission rate converted to an "equivalent" 150 lb finishing pig. Farm 20 has 2,352 piglets (25 lbs) + 1,940 sows (400 lbs) = 834,800 lbs or 5,565 equivalent 150 lb finishing pigs (834,800/150 = 5,565 equivalent finishing pigs).

[b]Value treated as an "equivalent" finishing pig. That is, it presumes three finishing pigs = weight of one sow and considers the weight of piglets negligible (0). Result: 7,480 finishing pigs + 3,636 equivalent pigs (i.e., 1,212 sows) = 11,116 equivalent finishing pigs.

Note: Bold data are data as reported in the literature.

Reviewing Harper and Sharpe's (1998) results, there appears to be no significant difference in the lagoon emission factors developed for the farrow-to-wean and finishing operations. Furthermore, there is more confidence in the lagoon emission factors developed for the finishing operation because two independent measurements are available. Therefore, we calculated the average emission factor of the two long-term studies (Aneja, Chauhan, and Walker, 2000; Harper and Sharpe, 1998) for the finishing operation and used this emission factor for all operation types. The Harper and Sharpe results indicate that this is a reasonable assumption for farrow-to-wean operations. The applicability of this emission factor to wean-to-feed operations is somewhat questionable considering the anticipated nitrogen loading rates to the lagoon (as estimated in the following section), but it is unavoidable because of the lack of lagoon emission measurement data at wean-to-feed operations.

Comparison of emission factors derived from the literature to monitoring results at two baseline operations. Over the course of this Agreement's research, the OPEN Team monitored ammonia emissions for multiple seasons at two sites operating traditional confinement houses, lagoons, and sprayfields (referred to as "baseline" operation). The monitoring results for the Stokes and Moore operations' lagoons are reported in the Settlement Agreement's Third Annual Progress Report (http://www.cals.ncsu.edu/waste_mgt/apwmc.htm).

Modeling reported in this final report uses emission factors derived from the literature. It was necessary to take this approach given that monitoring data were not available in a time frame consistent with contract milestones and completion.

The lagoon emission factors applied in the modeling were a mean of the Aneja, Chauhan, and Walker (2000) findings and the Harper and Sharpe (1998) findings. The OPEN Team states in the Third Annual Progress Report that the "conventional statistical ammonia flux model developed from the Stokes and Moore results was in very good agreement with previous published analysis" (i.e., Aneja, Chauhan, and Walker [2000], p. 91).

The emission correlation presented in Third Annual Progress Report was compared to the emission factor derived from the literature data. This correlation is

$$\mathrm{Log}_{10}\,(\text{A flux/ton}) = 3.8655 + 0.04491\,T_{lagoon} - 0.05946\,D. \qquad (2.2)$$

where

A flux/ton = emission rate (ug-N/min/1,000 kg live weight)

T_{lagoon} = temperature of the lagoon liquid (°C)

D = 0 when $T_{lagoon} > T_{air}$ and $D = T_{air} - T_{lagoon}$ otherwise.

We assumed for this analysis that D = 0, which would provide a reasonable but slightly high estimate of emissions, because D would otherwise reduce the predicted emissions. The average annual ambient air temperature for two of the three meteorological stations is 15.5°C and the average annual ambient air temperature for the third meteorological station is 17.5°C. The average annual lagoon temperature can be estimated from these annual average air temperatures. Therefore, Eq. (2.2) can be used to estimate the annual average ammonia emissions factors appropriate for North Carolina lagoons. For the two meteorological stations with annual average temperatures of 15.5°C, the correlation predicts an average emission factor of 1.59 kg NH_3/(150 lb pig)/yr. For the one meteorological station with annual average temperatures of 17.5°C, the correlation predicts an average emission factor of 1.95 kg NH_3/pig/yr. The emission factor used in this analysis was 1.61 kg NH_3/pig/yr, which agrees very well with these emission factors calculated from the correlation presented in Third Annual Progress Report.

Therefore, it is presumed that the model's use of the mean of Aneja et al. and Harper and Sharpe's numbers approximates the emissions from traditional North Carolina lagoons, with some possibility that these numbers slightly underestimate emissions.

Sprayfield Emission Factors. Sprayfield emission factors are generally developed using a fraction of nitrogen lost during and after spray application and a nitrogen balance (amount of nitrogen produced by the pigs less what volatilizes from the confinement house and lagoon). Therefore, rather than directly using the emission factors reported in the literature, RTI used this same generalized mass balance approach for estimating these emission factors. To accomplish this, we developed nitrogen excretion rates for each operation type. RTI used the housing and lagoon emission factors developed for those operations (as previously described) and assumed that 100 percent of the nitrogen excreted is converted to ammonia during storage and treatment. As described in Cure, Southerland, and Wooten (1999), we assumed that 25 percent of the ammonia remaining in the lagoon effluent is emitted during spray application, and an additional 30 percent of the ammonia that is applied during spray application subsequently volatilizes from the soil surface (rather than taken up by the vegetation).

The State of North Carolina uses certain average pig weights and production assumptions to estimate the SSLW on a per-sow basis for operations with multiple growth stages (see http://www.soil.ncsu.edu/certification/Manual/a/chapter3A.htm#table3-1). The North Carolina average steady-state live weight values for various growth stages and for operations with multiple growth stages are presented in Table 2-4. Nitrogen excretion rates used for this analysis, as developed by EPA (2001), are also summarized in Table 2-4.

Table 2-4. Summary of Relevant Information by Animal Type

a. Data for specific growth stages

Animal Type	Average Live Weight (lb/pig)[a]	Nitrogen Excretion Rate (lb/yr/1,000 lbs)[b]
Sows (gestating)	400[c]	70
Sows (lactating)	400[c]	171
Farrow to wean	10[d]	219
Wean to feed	30	219
Feed to finish	135	153
Boars	400	55

b. Assumptions and data for multiple growth stages operations

Parameter/Operation Type	Average Live Weight (lb/sow)[a]	Parameter Value[a]
Number of farrow/litter		10
Number of litters/year		2
Weanling age, days		21
Farrow to wean	433	
Farrow to feed	522	
Farrow to finish	1,417	

[a]Values used by North Carolina in the North Carolina swine operation survey, unless otherwise noted.
[b]U.S. Environmental Protection Agency (EPA). 2001. Emissions from Animal Feeding Operations Draft Report. U.S. EPA Contract No. 68-D6-0011. August 2001. Table 8-8.
[c]North Carolina does not distinguish between gestating and lactating sows.
[d]Not reported by North Carolina. Used value reported in EPA (2001), Table 8-9.

To perform the nitrogen mass balance for operations with multiple growth stages, we needed to assess the time-weighted average mixture of hogs within each growth stage for that multiple growth stage operation. For the most part, these values can be "back-calculated" from the data in Table 2-4. Additionally, because lactating sows have much higher nitrogen excretion rates, we also needed to estimate the relative number of lactating sows to the total number of sows on-site. Table 2-5 summarizes the assumptions and calculations used to estimate the average number of hogs in a given growth stage per 100 sows. The mixture of animals presented in Table 2-5 yields the North Carolina average steady-state live weights for multiple growth stage operations as presented in Table 2-4.

Table 2-5. Average Number of Animals On-Site for a Farrow to Finish Operation

Animal Type	Days/Event	Average Number of Head per 100 Sows
Gestating sows	295[a]	81[b]
Lactating sows	70[a]	19[b]
Boars	365[a]	5[c]
Farrow to wean	21	115[d]
Wean to feed	55[c]	301[d]
Feed to finish	121[c]	663[d]

[a]Days/year. Sows and boars are assumed to remain on-site year-round. Sows are assumed to be lactating (or have nitrogen excretion rates like lactating sows) for 35 days/litter with two litters per year.

[b]There are 70/365 or 19 percent of sows lactating on average.

[c]Values selected to achieve North Carolina steady-state live weights values for multiple growth stage operations.

[d]Calculated based on two litters/year, 10 farrows/litter, and relative duration of growth stage on-site. Example, the operation would produce 20 farrows/sow x 100 sows x 21/365 = 115 farrows on average.

The information in Tables 2-4 and 2-5 can be combined to calculate the nitrogen excretion rates for each of the model operation types. The results of this calculation are summarized in Table 2-6.

Once the ammonia generation rate is estimated, the basic algorithm to calculate the sprayfield emissions is as follows:

$$\text{Sprayfield emissions} = \text{spray emissions} + \text{field emissions} \qquad (2.3)$$

where

Spray emissions	=	25 percent of lagoon effluent rate
Lagoon effluent rate	=	NH_3 generation rate – Housing emissions factor – Lagoon emissions factor
Field emissions	=	30 percent of (lagoon effluent rate – spray emissions)

(Note: All emission factors or rates are in units of lbs_{NH_3}/yr/std.pig)

Applying these equations with the ammonia generation rates (Table 2-6) and the selected emission factors for confinement housing (Table 2-2) and lagoons (Table 2-3) for each of the five operating categories yields the sprayfield emission factors presented in Table 2-7.

Table 2-6. Nitrogen and Ammonia Production Rates by Type of Operation

Model Operation	Average Nitrogen Excretion Rate (lbs/yr/klbs SSLW)[a]	Ammonia Generation Rate (lbs NH_3/yr/std.pig)[b]
Farrow to wean	91	7.54
Wean to feed	219	18.13
Farrow to feed	113	9.37
Farrow to finish	138	11.45
Feed to finish	153	12.67

[a]SSLW = steady-state live weight.
[b]Assumes 100 percent of the nitrogen excreted is converted to ammonia, assumes “standard pig” weight of 150 lbs/pig, and accounts for increased molecular weight of ammonia compared to elemental nitrogen.

Table 2-7. Annual Average Composite Emission Factors

Emission Source	Animal Operation Category Emission Factors (kg NH_3/std.pig/yr)				
	Farrow to Wean	Wean to Feed	Farrow to Feed	Farrow to Finish	Feed to Finish
Confinement housing	1.81	1.21	1.81	2.98	2.98
Lagoon	1.61	1.61	1.61	1.61	1.61
Sprayfield	1.96	7.27	2.83	3.26	3.84
Total—operation	5.38	10.09	6.25	7.85	8.43
	Animal Operation Category Emission Factors (kg NH_3/SSLW/yr)				
Total—operation	0.036	0.067	0.042	0.052	0.056

Comparison of emission factors derived from the literature to monitoring results at two baseline operations. As included above, the OPEN Team monitored ammonia emissions for multiple seasons at two sites operating traditional confinement houses, lagoons, and sprayfields. In the Open Team’s portion of the Settlement Agreement’s Third Annual Progress Report (http://www.cals.ncsu.edu/waste_mgt/apwmc.htm), they reported that ammonia emission monitoring results for sprayfields at the two baseline operations of Stokes and Moore operations measured ammonia at nondetectable levels. The Team attributed this low level of ammonia emissions to the fact that no spraying had occurred at the field for 1 month. Because a major component of sprayfield emissions result while spray droplets are transported through the air and from wastewater as it rests on the soil and plants, it is logical that little emissions would be measured.

Model Operation Emission Factors. The emission factor for the entire CAFO is simply the sum of the emissions factors for each of the primary emission sources at the CAFO (i.e., the confinement house, lagoon, and sprayfield). Table 2-7 presents the emission factors for each of these three emission sources and the cumulative (i.e., “composite”) emission factor for the CAFO for each of the five operating categories.

Because the North Carolina Division of Water Quality (NCDWQ) survey information on operating capacity is reported both in terms of steady-state live weight and on the number of head (sows) on-site, the total model operation emission factors presented in Table 2-7 were also converted and presented in terms of steady-state live weight. These emission factors (in terms of steady-state live weight) can be used directly with the NCDWQ survey information to calculate the total ammonia emissions from each operation.

Uncertainties Encountered in Emission Factor Derivation. The following is a list of sources of uncertainty encountered in developing ammonia emission factors:

1. Lack of data led to the assignment of one category’s lagoon emission factor to all operational categories as a default.
2. In some circumstances, not all categories had available emission data. Therefore, emission factors were assigned from a known category that was more similar in animal weight (e.g., for confinement housing, the farrow-to-finish, and feed-to-finish categories were considered similar in average weight per animal on-site, so these two categories used the same emission factors).
3. Confinement housing factors varied in part due to housing design variations (i.e., waste collection designs varied). A portion of the emission factors also was available from European operations, which may not be most representative of U.S. operations.
4. Sprayfield emissions are based on mass balance with the assumption that nitrogen released is 100 percent ammonia; therefore, emissions may be overestimated. Data on nonammonia nitrogen composition were difficult to determine.

5. Some assumptions were needed to estimate the nitrogen production rate. The primary assumption needed was the relative time in which a sow was lactating (or had nitrogen excretion rates similar to a lactating sow). Other assumptions were also needed for this calculation; however, these assumptions were bounded by the fixed steady-state live weights for multiple growth stage operations. As such, these assumptions should add very little uncertainty to the calculated nitrogen production rates or the final ammonia emission factors.

Meteorological Regions Selection and Data Processing

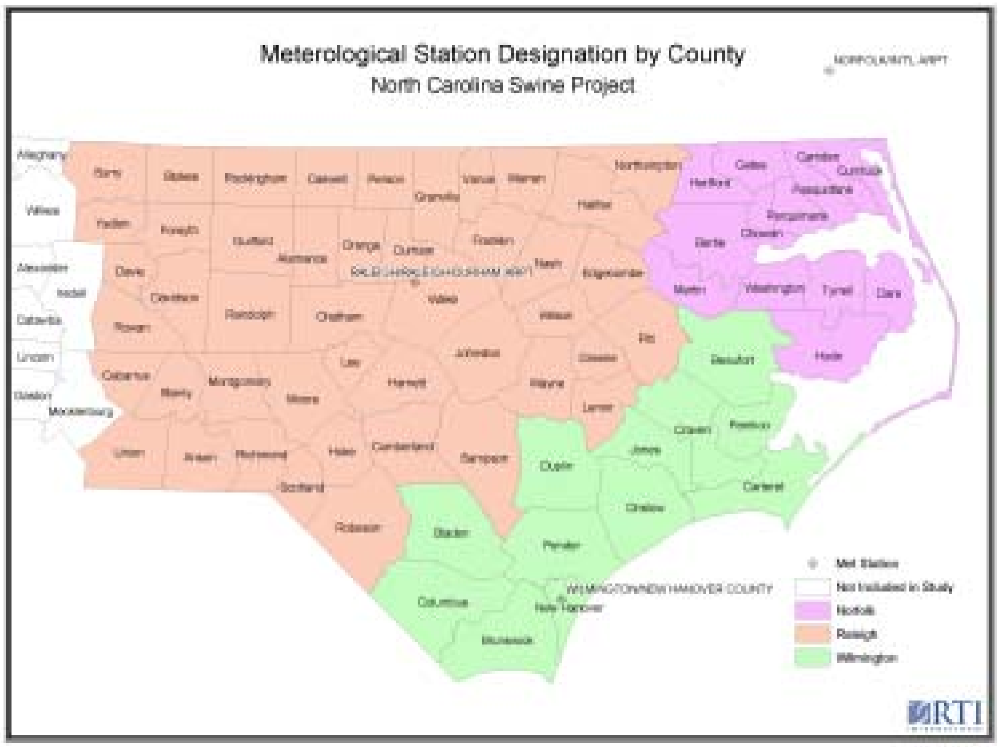
Three national weather service (NWS) meteorological stations were selected for the air modeling analysis in this study. They are Raleigh-Durham, North Carolina; Wilmington, North Carolina; and Norfolk, Virginia. Greensboro and Charlotte NWS meteorological stations are also in the study area but are located close to the edge of the study area. Only a few swine operations are located in the Greensboro and Charlotte area. The windrose from Raleigh-Durham shows similarity to those from Greensboro and Charlotte. Therefore, Raleigh-Durham's meteorological data were used to represent the Greensboro and Charlotte area due to project budget limitations.

The next step was to divide the study area into three subareas. Each subarea is most similar in meteorological conditions to those measured at the meteorological station. In general, the primary delineation of areas is based on geographic features affecting synoptic winds, including mountain ranges and plains. The secondary delineation is based on features affecting mesoscale (several hundreds kilometers) winds, including coastal regions and basic land cover classifications of forest, agriculture, and barren lands. Because the study area is located in the eastern North Carolina plain and its size ranges about several hundreds of kilometers, the features that affect mesoscale winds should be considered.

The North Carolina Department of Environment and Natural Resources (NCDENR) provides its recommendation on what NWS meteorological station should be used for ISCST3 modeling for each county of North Carolina. The recommendation is listed in Table 4-2 of "Guidelines for Evaluating the Air Quality Impacts of Toxic Pollutants in North Carolina" (NCDENR, 1999). In this analysis, we carefully examined NCDENR's recommendations and decided to adopt NCDENR's station assignments.

Because we decided to use Raleigh-Durham's NWS stations for the Greensboro and Charlotte areas, those counties that were assigned to use Greensboro and Charlotte data by NCDENR were assigned to use Raleigh-Durham's data. Two coastal counties, Tyrrell and Dare, were not assigned to any NWS by NCDENR, so RTI chose to use Norfolk's data because their surrounding counties were assigned to Norfolk. Figure 2-2 shows the area covered by each meteorological station.

Figure 2-2. Meteorological Station Designation by County



Growth Stage

To compute deposition by each facility in the study area, we established the following steps:

1. Assign the facility to one of three meteorological stations: Raleigh, North Carolina; Wilmington, North Carolina; or Norfolk, Virginia.

2. Assign the facility to an acreage category.
3. Assign the facility to a growth stage.

Swine are managed at numerous levels that are driven by growth stage and optimum combinations of growth stages for growers. Because 19 unique combinations of growth stages were reported on North Carolina's CAFO Survey database, it was necessary to condense the number to a size that is manageable for modeling. We reduced the list of 19 to 5 combinations (for which emission factors were available) and assigned each of the 19 combinations to the most representative of the five categories below:

A. Farrow to wean
   (1) gilts
   (2) farrow to wean, gilts
   (3) boar, stud
B. Farrow to feeder
   (1) wean to feed, farrow to wean, gilts
   (2) wean to feed, farrow to wean
   (3) farrow to wean, farrow to feeder
   (4) farrow to feeder, gilts
C. Farrow to finish
   (1) wean to feed, feeder to finish, farrow to wean, boar stud or gilts
   (2) wean to feed, feeder to finish, farrow to wean
   (3) farrow to feeder, feeder to finish
   (4) feeder to finish, farrow to feeder
   (5) feeder to finish, farrow to finish
   (6) farrow to wean, farrow to finish
   (7) farrow to wean, farrow to feeder, farrow to finish
   (8) farrow to feeder, farrow to finish
   (9) feeder to finish, farrow to wean
D. Wean to feeder
E. Feeder to finish
   (1) wean to feed, feed to finish
   (2) feeder to finish, boar stud

Once operations were assigned to one of five growth stage categories, growth stage-specific emission factors were applied to each operation:

1. farrow to wean
2. farrow to feeder
3. farrow to finish
4. wean to feeder
5. feeder to finish

There are a total of 60 combinations of representative meteorological locations/operation sizes (12) and operating categories (5).

Thus, Eq. (2.4) computes emission rates per CAFO:

$$\text{Emission rate} = \frac{\text{(Emission factor x no. head)}}{\text{acres (expressed as meter square)}} \tag{2.4}$$

In summary, RTI performed the following tasks:

1. Labeled each CAFO as 1 through 12 to reflect the appropriate model CAFO:

Meteorological Station	50 Acres	100 Acres	260 Acres	500 Acres
Norfolk	1	2	3	4
Wilmington	5	6	7	8
Raleigh	9	10	11	12

2. Assigned each CAFO to one of five production (growth stage) categories.
3. Identified the emission factor for that production category.
4. Computed the emission "rate":

$$\frac{\text{Emission Factor X No. Head Pigs for Each CAFO}}{\text{Assigned Model Acreage}}$$

2.1.2 Model CAFOs Development

Selection of Acreage to Apply to Model Swine Operations for Ammonia Dispersion and Deposition Modeling

Acreage Assumptions. The goal of this modeling exercise is to estimate the amount of ammonia deposited onsite and within a 50 km radius of the swine facility's boundary based on a unit emission rate of 1 mg/s/m^2. The ISCST3 model requires that site acreage be selected to represent the extent of the area emission source (i.e., the facility and its waste management operations).

In this section, we describe the development of assumptions used for acreage from which the unitized emission rate emanates. We reviewed readily available North Carolina information sources to determine the average number of acres in a North Carolina swine CAFO (described below).

North Carolina Division of Water Quality Information. North Carolina DENR's DWQ maintains a database of swine operations; however, the database does not contain information on the acreage of each swine operation in North Carolina. Therefore, no statistics could be computed on the range of acreage relative to swine population on North Carolina sites.

RTI telephoned the NCDWQ to learn about any information on average swine CAFO acreage and confirmed that no such data were recorded. However, based on his experience and discussion with an industry representative, the DWQ representative determined that 42 to 52 acres is a reasonable range of acreage for a 3,000- to 3,500-head swine CAFO in North Carolina. (These CAFOs were in operation prior to North Carolina's adoption of buffer requirements [Ramsay, 2002]). Therefore, we chose one of the model facility sizes to be 50 acres for a 3,000-swine operation.

North Carolina Agriculture Census Data. The North Carolina Agricultural Census (NCDA&CS, 1999) reports acreage ranges for swine operations but does not relate the acreage to number of swine. (See Table 2-8 [Table 4-9 of the 1997 Agricultural Census] [NCDA&CS, 1999].) However, the Agricultural Census does quantify the number of swine CAFOs by swine population grown. (See Table 2-9 [Table 31 of 1997 Agricultural Census].)

From the distribution in Table 2-8, RTI selected three model acreages from which to emit the unitized emission factor:

- 100 acres (50th percentile)
- 260 acres (75th percentile)
- 500 acres (90th percentile)

Table 2-8. Number and Acreage Range of North Carolina Swine Operations

Acre[a]	Number of Operations[a]	Cumulative Number of Operations	Percentile (Number of Operations)
1–9	255	255	
10–49	609	864	
50–69	224	1,088	
70–99	238	1,326	50th (1,333 operations)
100–139	267	1,593	
140–179	159	1,752	
180–219	120	1,872	
220–259	101	1,973	75th (1,999 operations)
260–499	300	2,273	90th (2,300 operations)
500–999	226	2,499	
1,000–1,999	114	2,613	
≥ 2,000	53	2,666	
Total	2,666	2,666	

[a]North Carolina Department of Agriculture & Consumer Services (NCDA&CS). 1999. "1997 Census of Agriculture—North Carolina." <www.agr.state.nc.us/stats/census/htm>. See Table 49 of 1997 Census of Agriculture—Volume 1: North Carolina, State and County Tables.

Conclusions Based on DWQ and Census Data. We concluded that four model acreages are representative and appropriate for conducting the ISCST dispersion and deposition model:

- ➤ 50 acres (Ramsay [2002] recommendation for 3,000 to 3,500 head CAFO)
- ➤ 100 acres (50th percentile of 1997 North Carolina Agricultural Census)
- ➤ 260 acres (75th percentile of 1997 North Carolina Agricultural Census)
- ➤ 500 acres (90th percentile of 1997 North Carolina Agricultural Census)

We believe that, by applying these acreages to the model, suitable emission rates can be estimated for each of the five, population-based categories designated for analysis:

- ➤ 0 to 500 animal units (AU) 0 to 1,250 hogs
- ➤ 500 to 1,000 AU 1,251 to 2,500 hogs

Table 2-9. Population and Number of North Carolina Swine Operations

Number of Head of Hogs and Pigs	Number of Operations in 1997	Cumulative Number of Operations	Percentile (Number of Operations)
1–24[a]	944		
25–49[a]	170		
50–99[a]	125		
100–199[a]	72		
Subtotal	1,311 (without 1–24 head = 367)		
200–499	77	77	
500–999	132	209	25th percentile (418 operations)
1,000–1,999	236	445	50th percentile (837 operations)
2,000–4,999	648	1,093	75th percentile (1,507)
5,000 and more	582	1,675	
Total	2,986[b]	986[b]	

[a]Rows excluded because fewer than the 250-head trigger for North Carolina Department of Environmental and Natural Resources (DENR) permitting.

[b]2,986 operations is 320 greater than the 2,666 in Table 49 (NCDA&CS, 1999). 320 operations is comparable (less than 15 percent difference) to the 367 unregulated small operations in Table 31's subtotal (NCDA&CS, 1999).

Source: North Carolina Department of Agriculture & Consumer Services (NCDA&CS). 1999. "1997 Census of Agriculture—North Carolina." <www.agr.state.nc.us/stats/census/htm>. See Table 49 of 1997 Census of Agriculture—Volume 1: North Carolina, State and County Tables.

- 1,000 to 1,500 AU 2,501 to 3,750 hogs
- 1,500 to 2,0000 AU 3,751 to 5,000 hogs
- 2,000 to 2,500 AU 5,001 to 6,250 hogs

Alternative Approach: Selecting Model CAFO Acreage Based on Reported Number of Animals. Neither the North Carolina inventory of swine operations nor the North Carolina Agricultural Census (NCDA&CS, 1999) contains a comparison of acres to number of animals grown. In the absence of such data, the only source available was the DENR/industry opinion that 42 to 52 acres is a reasonable range of acreage for a 3,000- to 3,500-head swine CAFO in North Carolina. Presuming a linear relationship in population and acreage would mean that a 21,000-head CAFO would be approximately 350 acres. However, no data are readily available to support the presumption of a linear relationship.

Testing the Acreage Against Sprayfield Size Requirements. Based on a review of literature, RTI considered the minimum area required for a 3,000-head CAFO sprayfield to be 20 acres. This assumption is supported by data presented in Tables 2-10 and 2-11 for grasses such as fescue, Carolina Bermuda, and hay. Applying a general rule that houses, lagoons, and sprayfields occupy approximately equal areas at a CAFOs (i.e., each consuming one-third of the total CAFO area), a 3,000-head CAFO would be about 60 acres in size (within 20 percent of the 50-acre model size recommended), and a 21,000 head CAFO (requiring a minimum 140-acre sprayfield) would be about 520 acres in size (within 5 percent of the 500-acre model size that represents the 90th percentile based on Agricultural Census data).

Recommendations. Given the lack of information relating swine populations to swine CAFO acreage, we recommend relying on North Carolina Agricultural Census data relating the number of swine CAFOs to acreage in conjunction with estimated population-based acreage needs for sprayfields. The only exception is the solicited expert opinion of the North Carolina DWQ in conjunction with representatives of the swine industry, which relates an average North Carolina swine operation acreage for an average North Carolina swine operation. Thus,

- If 0 to 3,750 hogs, assign 50-acre model
  (Source: North Carolina DWQ)
- If 3,751 to 6,250 hogs, assign 100-acre model
  (Source: North Carolina Agricultural Census supported by sprayfield acreage needs)
- If 6,250 to 21,000 hogs, assign 260-acre model
  (Source: North Carolina Agricultural Census supported by sprayfield acreage needs)
- If greater than 21,000 hogs, assign 500-acre model
  (Source: North Carolina Agricultural Census supported by sprayfield acreage needs)

This approach is reinforced by testing the reality of acreage assumptions using general acreage demand assumptions for sprayfields based on swine population size. Where application of a model acreage appears unrepresentative due to population- and crop-based acreage requirements for spray application, the acreage assumption was not applied to the population category for the modeling analysis. For example, it appears unrepresentative for a 0 to 1,250-swine operation to require a 500-acre facility size to function.

Table 2-10. Acreage Requirements for Animal Waste Application

Anaerobic Lagoon % Nitrogen Loss	Acres: Finishing Only Per Head (135 lbs) Fescue	Acres: Finishing Only Per Head (135 lbs) Carolina Bermuda	Acres: Farrow to Feeder per Sow (522 lbs) Fescue	Acres: Farrow to Feeder per Sow (522 lbs) Carolina Bermuda	Acres: Farrow to Finish per Sow (1,417 lbs) Fescue	Acres: Farrow to Finish per Sow (1,417 lbs) Carolina Bermuda
20%	0.05	0.03	0.18	0.12	0.33	0.17
50%	0.03	0.02	0.11	0.08	0.31	0.21
75%	0.15	0.01	0.06	0.04	0.16	0.10
85%	0.01	0.006[a]	0.03	0.02	0.09	0.06
Mean	0.0263	0.0165	0.095	0.065	0.222	0.13

[a]0.006 ac per head of finishing pig x 3,000 head per CAFO = 18 acres.

Source: Barker, J.C. 1980. "Land Area Guidelines for Livestock Waste Application." AG-199. North Carolina Agricultural Extension Service.

Table 2-11. Acres Under Application by North Carolina Swine Producers

Crop	Number of Swine: Less than 250	Number of Swine: 250–999	Number of Swine: 1,000+	Under Contract: Yes	Under Contract: No
Corn	42	37	39	32	44
Soybeans	11	19	19	16	16
Hay or alfalfa	17	25	40	40	20
Other crop(s)+	40	50	42	46	43
Mean acres under application	25	67	66	67	45
Median acres under application	10	20	35	24	20
Number of respondents	65	72	70	83	129

Note: Asked only of those who apply waste to their land.

Source. Hoban, T.J., and W.B. Clifford. 1995. "Managing North Carolina's Livestock Waste—Challenges and Opportunities." Raleigh, NC: North Carolina State University.

Capacities

RTI applied the set of model CAFO SSLW categories that was developed for the NCSU-ARE cost analysis (see Chapter 1) to the emission modeling. As shown in Table 2-12, these model CAFOs are intended to represent the variety of operations growing most of the pigs in North Carolina.

Table 2-12. Standard Swine CAFO Types and SSLW Categories Used in This Study

Swine CAFO Type	Swine CAFO SSLW Capacity (1,000 lbs)				
	0–500	**500–1,000**	**1,000–5,000**	**1,500–2,000**	**2,000+**
Farrow-Wean	•	•	•	•	•
Farrow-Feeder	•	•	•	•	•
Farrow-Finish	•	•	•	•	•
Wean-Finish	•				
Feeder-Finish	•	•	•	•	•

Note: Grey-shaded area indicates CAFO categories that are not represented in the population and that were not analyzed.

In summary,

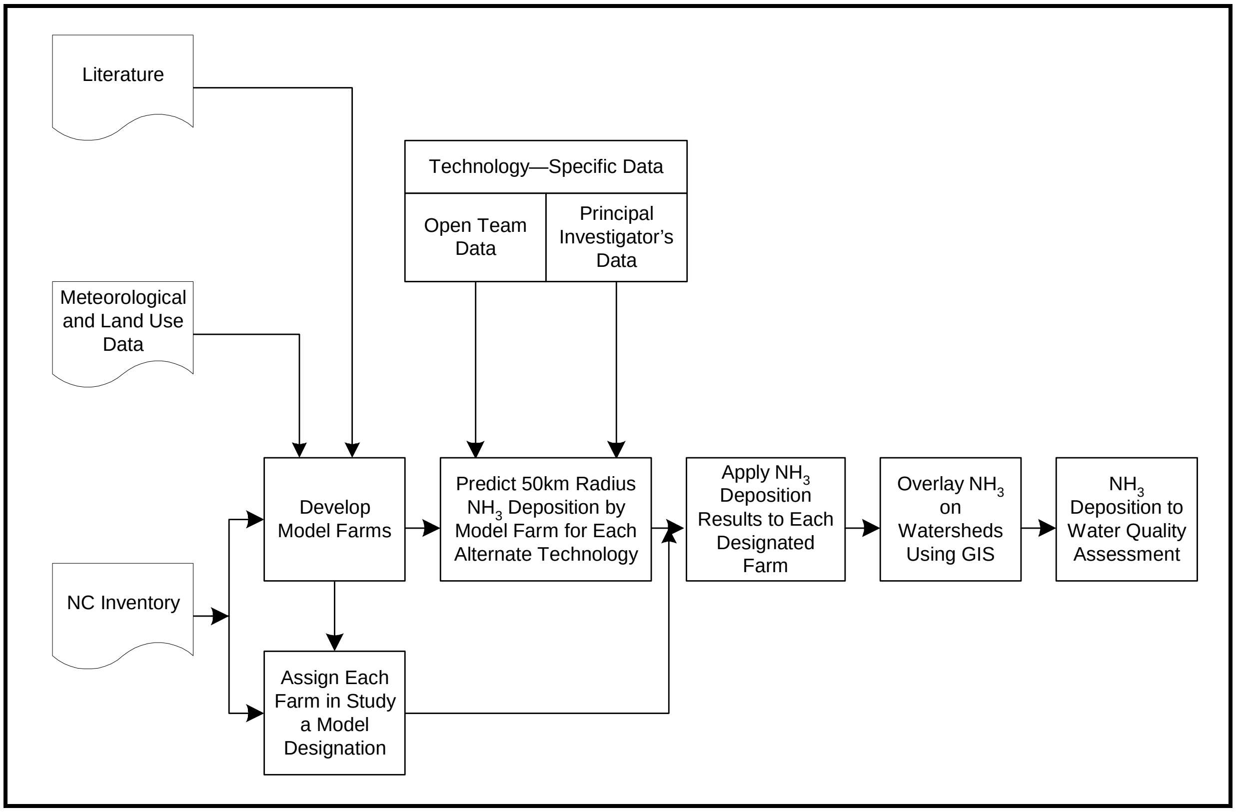
- ➤ If an operation's SSLW capacity was 19,440 to 499,230 lbs, it was modeled as a 50-acre CAFO.
- ➤ If an operation's SSLW capacity was 500,400 to 996,960 lbs, it was modeled as a 100-acre CAFO.
- ➤ If an operation's SSLW capacity was 1,005,750 to 1,986,300 lbs, it was modeled as a 260-acre CAFO.
- ➤ If an operation's SSLW capacity was 2,030,400 lbs to 10,182,400 lbs, it was modeled as a 500-acre CAFO.

2.2 AMMONIA AIR DISPERSION AND DEPOSITION PREDICTION

This section describes how RTI modeled ammonia emissions' dispersion and deposition in the study area. Figure 2-3 depicts how available emission factors from traditional housing, lagoon, and sprayfield technologies were modeled to predict ammonia's dispersion and deposition (up to 50 km radius from the edge of the facility). The deposition from each facility in the study area was in turn mapped and incorporated in the surface water quality model.

The following subsections describe our unitized deposition modeling, post-processing to apply modeling results to the study area's 2,295 CAFOs, GIS applications, and deposition modeling results for the study area.

Figure 2-3. Ammonia Atmospheric Dispersion-Deposition Modeling Approach



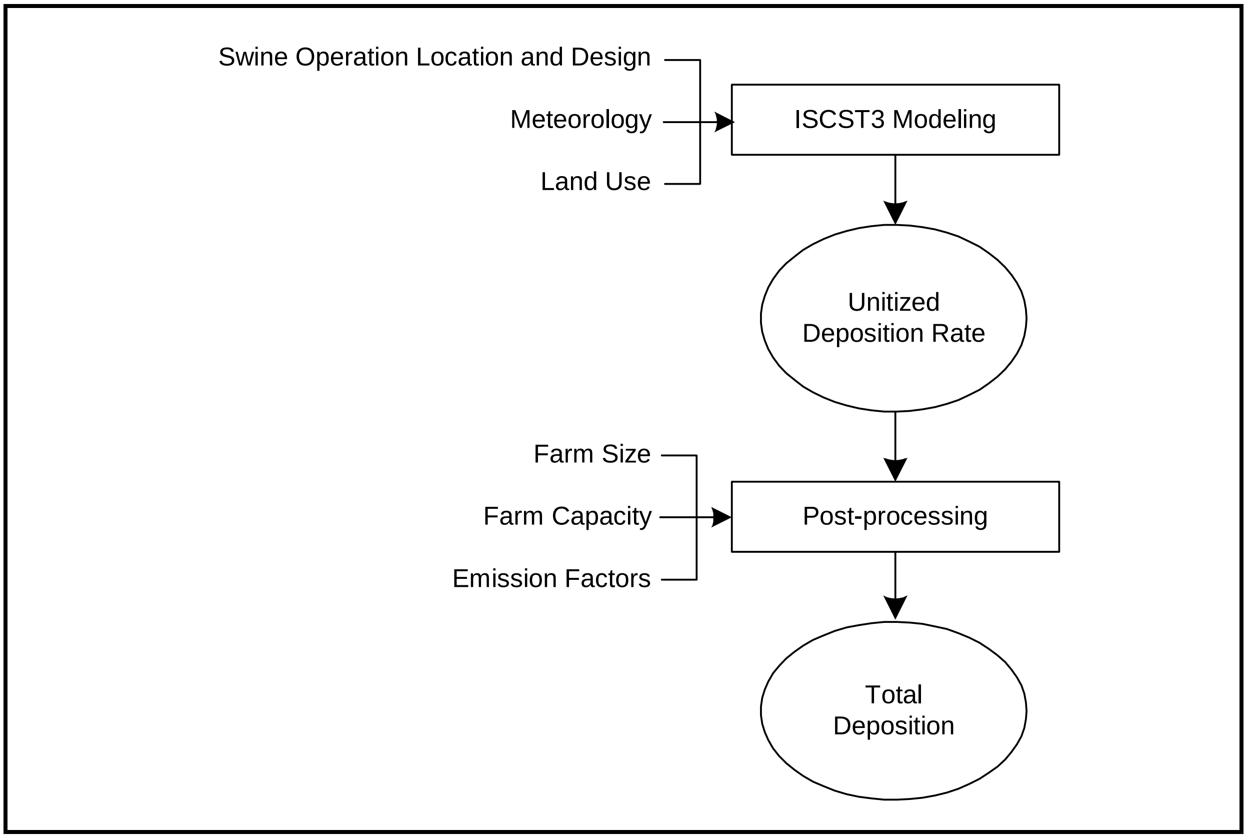
2.2.1 Unitized Deposition Modeling by Model CAFO

Application of the ISCST3 Model

The modeling effort described in this section characterizes the dispersion and deposition of ammonia as a gas. (Gaseous ammonia deposits faster [nearer field] than other ammonia species, such as fine particulate ammonia sulfate that may form following ammonia's release to the atmosphere.) The model runs assume that the emission rate is 1 mg per second per square meter (a unit emission rate) and the chemical composition is 100 percent ammonia.

RTI used ISCST3, version 02035, to model the dispersion and deposition of ammonia. This model is a standard, EPA-approved model for predicting atmospheric dispersion and deposition of specific chemical species (up to about 50 km from the source). Figure 2-4 shows the components of the system and the approach RTI used to estimate ammonia deposition on swine operations.

Figure 2-4. Approach to Estimating Total Ammonia Deposition from Swine Operations



The inputs depicted in Figure 2-4 require a variety of data. These requirements and the data sources are presented in Table 2-13.

To compute dispersion and deposition, we first assumed that a model swine facility contains three ammonia-emitting sources: animal houses with their waste collection systems; waste lagoon(s); and sprayfield(s) emitting 24 hours per day, 365 days per year. It is understood that these units do not operate full time. In some cases, lagoons may be temporarily closed for sludge removal. Given the variability in facility dimensions and layout across the state, we chose not to locate and model each emission source on the facility site individually. Rather, the model facility was treated as a single-area emission source. RTI also assumed that the area emission source was located at ground level, which is characteristic of swine waste management operations such as lagoons and sprayfields. The selection of the model CAFOs was described in previous sections.

Table 2-13. Overview of Primary Data Requirements

Data Type	Data Description/Specification	Data Source
ISCST3 Data	ISCST3 version 02035	EPA (1995a, 1995b, 2002a)
Chemical modeled	Ammonia (NH_3)	
NH_3 dry deposition velocity	1 cm/s	Median of two sources Hov and Hjollo (1994) (0.6 to 5 cm/s); Sutton, Moncrieff, and Fowler (1992) (4 to 5 cm/s)
NH_3 scavenging rate coefficient	6E-5 hour/(mm-s) for liquid precipitation 2E-5 hour/(mm-s) for frozen precipitation	Default value of HNO_3 in CALPUFF model. Jonson and Berge (1995) also show the wet scavenging coefficients for HNO_3 and NH_3 are the same. EPA suggests that frozen precipitation is one-third of the liquid value.
Meteorological station data (5 years of data)	Raleigh, North Carolina Wilmington, North Carolina Norfolk, VA (each operation assigned to one of three stations based on location)	National Weather Service
Setting	Average land use in each meteorological station region	RTI
Site shape	Circular	RTI
Site receptor grid	Onsite: 3 concentric receptor rings with 108 receptor points Offsite: 27 concentric receptor rings up to 50 km from edge of site boundary	RTI
Site sizes	50 ac 100 ac 260 ac 500 ac	RTI derived from Agricultural Census Data
Site emission character	Area source	RTI
Operating time	24 hr/d, 365 d/yr	
Land use		Geographic Information Retrieval and Analysis System (GIRAS) (USGS, 1990) converted into ARCINFO GIS (EPA, 1994)
Post-processing Data		
Swine operation's location, type, capacity		NCDWQ
Growth categories	Farrow to wean Wean to feed Farrow to feed Farrow to finish Feed to finish	NCSU APWMC (2002) recommended for economic analysis
Emission factors	Derived for each growth category where possible	Literature and ongoing NCSU studies

Once all inputs for the ISCST3 unitized model were prepared, the model was run to predict the dispersion and deposition of 1 mg/s/m^2 of ammonia of 12 model CAFOs. The results of this “unitized modeling” are presented in Table 2-14 and Figures 2-5 through 2-7.

Table 2-14. ISCST3 Unitized Ammonia Deposition Results for Each Model CAFO[a]

12 Model CAFOs				
NWS Station (represents meteorological region)	**CAFO Acreage**	**Total Deposition (Mg/yr)**	**Dry Deposition (Mg/yr)**	**Wet Deposition (Mg/yr)**
Norfolk, VA	500	33,522	33,177	345
Raleigh, NC	500	39,956	39,566	390
Wilmington, NC	500	36,846	36,360	486
Norfolk, VA	260	17,531	17,390	142
Raleigh, NC	260	20,916	20,756	161
Wilmington, NC	260	19,286	19,083	202
Norfolk, VA	100	6,823	6,784	39
Raleigh, NC	100	8,155	8,110	44
Wilmington, NC	100	7,517	7,460	57
Norfolk, VA	50	3,457	3,441	16
Raleigh, NC	50	4,138	4,120	18
Wilmington, NC	50	3,813	3,790	23

NWS = National Weather Service
[a]Assumes source’s emission rate of 1 mg/s/m^2.

2.2.2 Post-ISC Site-Specific Data Processing

Once we completed the unitized modeling, we applied a post-processing program that multiplies an annual average composite site emission rate by the unitized deposition rate. To develop a composite baseline site emission rate, we reviewed the literature to find annual emission factors for each source (house, lagoon, and sprayfield) (see Section 2.1.1). We used the sum of the three emission factors as a single “composite” emission factor for the entire facility site. The composition of that area emission source was designed so that emission factors for individual sources could be adjusted or substituted based on alternative technology scenarios at a later time. (Chapter 7 discusses RTI’s method for computing impacts of emission reduction scenarios.)

Figure 2-5. Maximum ISCST3 Unitized Ammonia Deposition Rate at Downwind Distances for the Raleigh Durham Meteorological Region

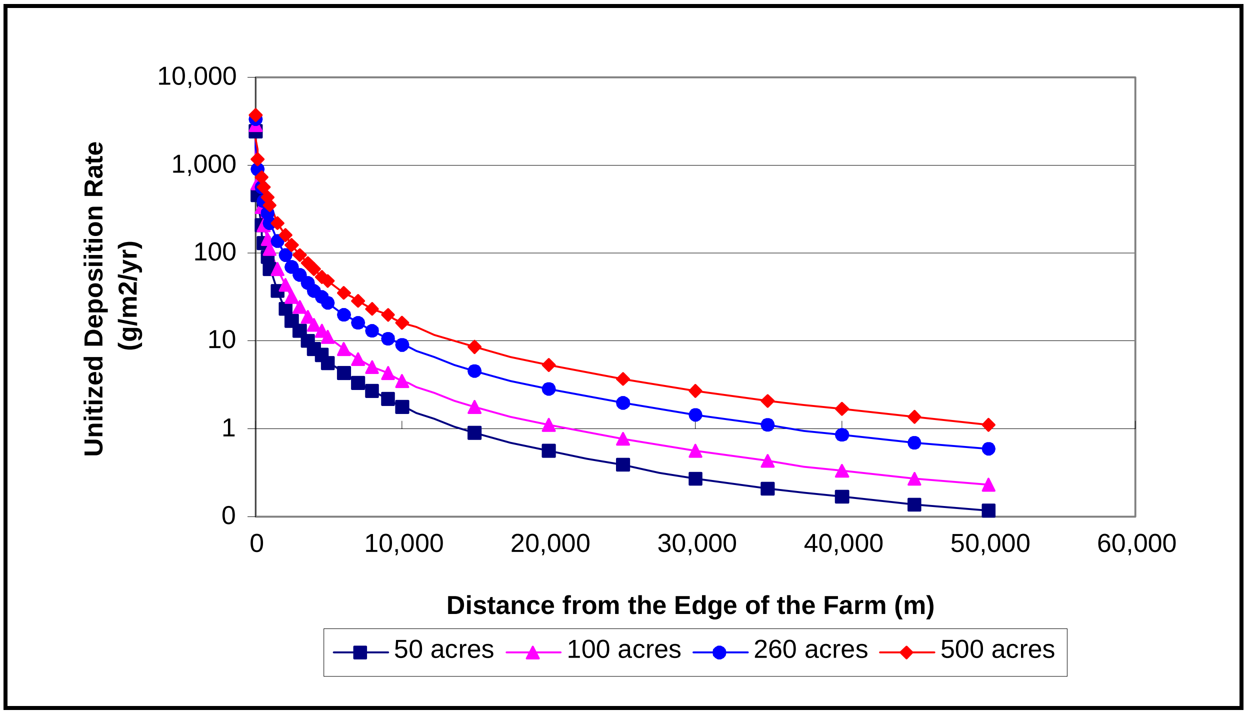


Figure 2-6. Maximum ISCST3 Unitized Ammonia Deposition Rate at Downwind Distances for the Norfolk Meteorological Region

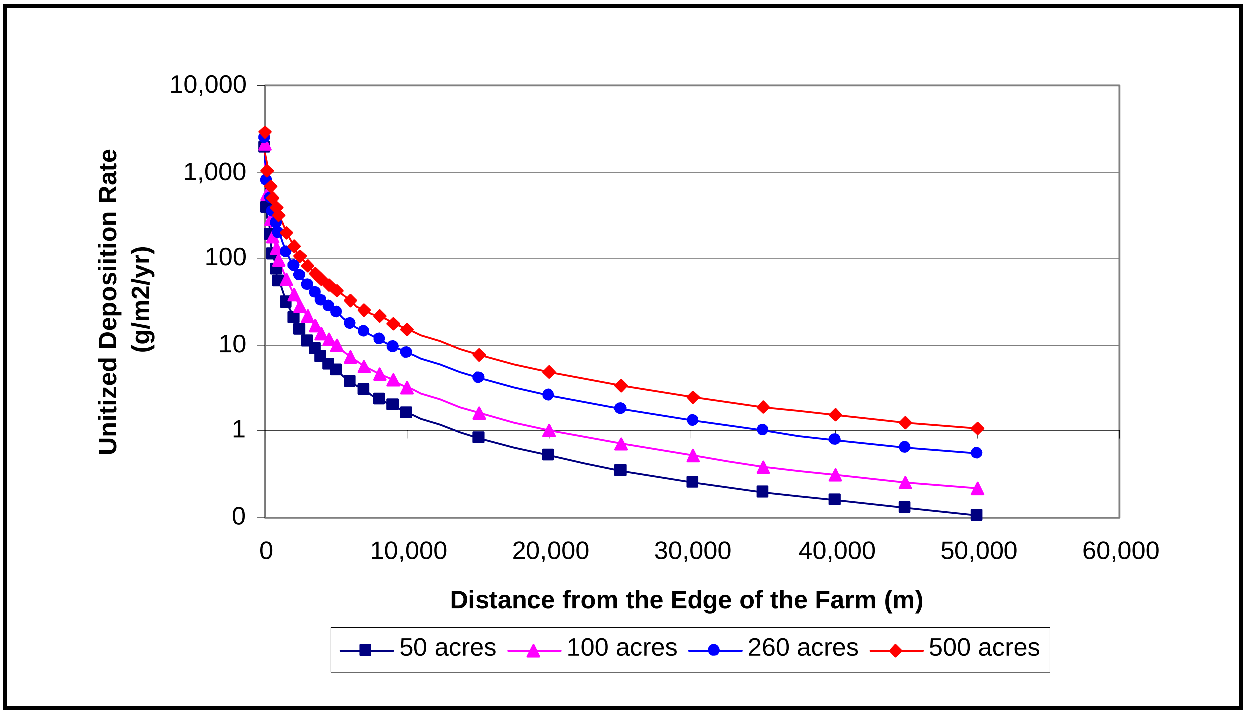
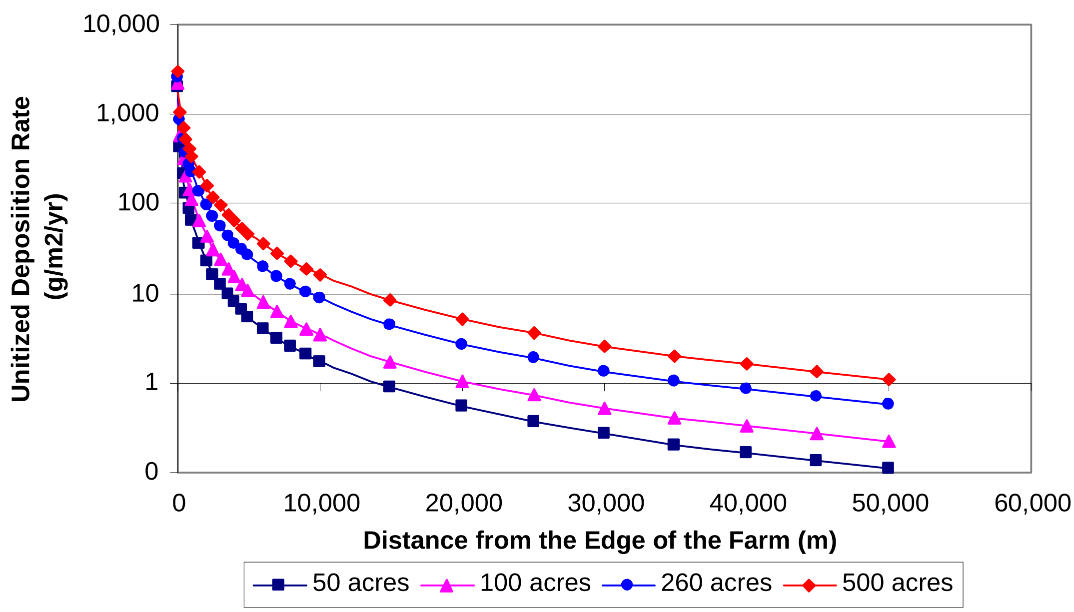


Figure 2-7. Maximum ISCST3 Unitized Ammonia Deposition Rate at Downwind Distances for the Wilmington Meteorological Region



Post-processing computed the predicted actual deposition on and around each swine operation (up to 50 km from edge of operation) in the study area.

Post-processing estimated the actual deposition on and around each swine operation (up to 50 km from edge of operation) in the study area. To perform this estimation, RTI calculated the total deposition at each receptor point in the facility's modeling grid. Using the 550+ watershed boundaries provided from water quality modeling, we computed the total load of all operations within each watershed. If a 50 km radius covers more than one watershed, the deposition crossing the watershed boundary was accounted for in that adjacent watershed. If the 50 km radii of two or more facilities overlap in a watershed, the total load of the facilities in the overlapping area was computed. RTI's air modeling team presented GIS-formatted deposition by watershed to RTI's water quality modeling team for input in their modeling analysis.

2.2.3 GIS Application Approach

A GIS was used to model the ammonia deposition for all 2,295 CAFOs in the study area. The ISCST3 air modeling program produced air dispersion rate files for 12 different deposition scenarios. RTI modeled four swine operation sizes (50, 100, 260, and 500 acres) for three meteorological regions (Wilmington, North Carolina; Raleigh, North Carolina; and Norfolk, Virginia) for a total of 12 combinations (model CAFOs). Thus, each CAFO was assigned to one of the meteorological regions based on its geographic location and to one of the CAFO sizes based on the number of head and growth stage of its livestock.

Once RTI assigned each CAFO to a "model CAFO type," we calculated the actual amount of dry, wet, and total ammonia deposition at each point. This was done by creating Thiessen polygons around each modeled deposition point. Because the value produced by the ISCST3 model was a deposition ***rate*** based on a unitized emission factor, the actual deposition amount was calculated by multiplying the area of the Thiessen polygon by the emission factor assigned to the CAFO by the unitized emission rate. This resulted in a total for each Thiessen polygon in grams/year for total, dry, and wet deposition. These totals were then calculated back onto the point contained within the Thiessen polygon.

Each air dispersion file contained Cartesian coordinates for each of the 1,087 modeled deposition points. Each CAFO had a geographic location (latitude and longitude) for its origin. Once we assigned a given CAFO to a model CAFO type, we placed the deposition pattern on the ground (georeferenced) using the latitude and longitude of the origin. Each model operation was placed in turn until all 2,295 swine operations (each with 1,087 points) had been georeferenced to the earth's surface. This resulted in a very large point coverage of 2,494,665 individual deposition points.

RTI then overlaid this large point coverage with the hydrologic unit (a.k.a. HUC) boundaries for those HUCs in the study area. The 14-digit HUC ID that each point fell into was transferred to each of the 2,494,665 points. RTI calculated summary statistics for total, dry, and wet deposition on a HUC-by-HUC basis.

2.2.4 Deposition Results for the Study Area

Results of baseline modeling showed that, when accounting for deposition only in the 50 km radius of each CAFO, about 34,000 megagrams (over 37,000 short tons) of 2,295 CAFOs' ammonia emissions were deposited in the study area in 1 year. These CAFOs deposited an additional 7,800 megagrams (about 8,600 short tons) of ammonia outside the study area where some CAFOs' 50 km radii crossed over the study area boundary. In total, the model predicted the 2,295 CAFOs deposited approximately 43,000 megagrams of ammonia. These values represent the sum of each of the 2,295 CAFOs' deposition within a 50 km radius of each CAFO, excluding any ammonia that transports and deposits beyond the 50 km radius. As mentioned before, this modeling exercise presumes ammonia remains gaseous throughout its atmospheric dispersion within a 50 km radius. However, a fraction of ammonia may convert to an ammonium salt that is an aerosol (fine particulate). We address the destiny of aerosol ammonium in Chapter 3.

In total, the model predicted the 2,295 CAFOs deposited approximately 43,000 megagrams of ammonia. These values represent the sum of each of the 2,295 CAFOs' deposition within a 50 km radius of each CAFO, excluding any ammonia that transports and deposits beyond the 50 km radius.

Figures 2-8 and 2-9 depict the range of deposition by county and by hydrologic unit, respectively. (There are over 550 HUCs [or small watersheds] in the study area.) The greatest deposition occurs in Sampson and Duplin counties. The 10 HUCs estimated to have the greatest ammonia deposition are all within the Cape Fear River basin (see Table 2-15). These 10 HUCs are highlighted in Figures 2-8 and 2-9. The 10 HUCs total about 5,700 Mg/yr of ammonia deposition, which is about 17 percent of the study area's total annual ammonia deposition.

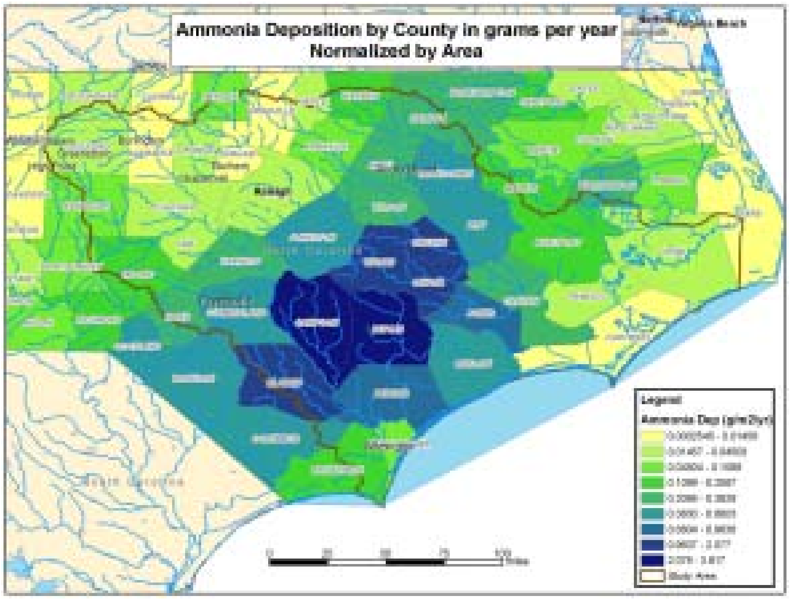
The ranking of counties and HUCs is a function of animal density and the density and proximity of CAFOs to one another. As Figure 2-8 demonstrates, when CAFOs are located near one another, the ammonia deposition for each CAFO's 50 km radius can overlap with another CAFO, thus multiplying the ammonia deposition/loading to HUCs.

RTI used this baseline ammonia deposition data in the Environmental Benefits Assessment Model described in Chapter 7. These baseline data are reduced according to an alternative waste management scenario's effectiveness in reducing CAFO ammonia emissions.

2.3 DATA AND MODEL QUALITY ASSURANCE

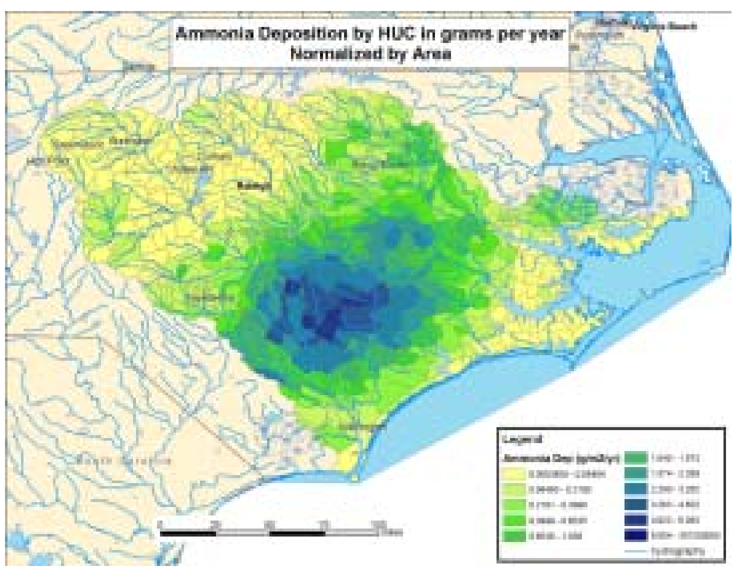
As indicated above, ISCST3 is a standard, EPA-approved model for predicting unitized atmospheric dispersion and deposition of specific chemical species. Below is a list of the QA/QC activities associated with the application of the ISCST3 modeling effort:

Figure 2-8. Modeled Ammonia Gas Deposition from 2,295 Swine Operations (by County) in Grams per Year Normalized by Area[a]



[a]Modeling used the NCDWQ survey (of swine operations) database, NCDA&CS 1997 Ag Census data and published emission factors in conjunction with EPA's ISCST3 model, and NWS meteorological data for three North Carolina regions.

Figure 2-9. Modeled Ammonia Gas Deposition from 2,295 Swine Operations (by 14-digit HUC) in Grams per Year Normalized by Area[a]



[a]Modeling used the NCDWQ survey (of swine operations) database, NCDA&CS 1997 Ag Census data and published emission factors in conjunction with EPA's ISCST3 model, and NWS meteorological data for three North Carolina regions.

Table 2-15. 10 HUCs Modeled as Receiving the Greatest Ammonia Deposition

HUC	Total NH_3 Deposition (Mg/yr)	Dry NH_3 Deposition (Mg/yr)	Wet NH_3 Deposition (Mg/yr)	River Basin
3030007050010	722	689	32	Cape Fear
3030006110040	694	665	29	Cape Fear
3030007090010	685	659	26	Cape Fear
3030007010020	642	616	25	Cape Fear
3030006110010	602	580	23	Cape Fear
3030006090060	509	486	23	Cape Fear
3030006110020	508	485	23	Cape Fear
3030006090040	464	450	22	Cape Fear
3030007020010	455	434	22	Cape Fear
3030006100020	437	428	22	Cape Fear

- Verify for each site that clipped area land use/cover codes match the original data set, areas in 3 km radius add up to the total area, and predominant land use/cover code as reported in the spreadsheet is the predominant land use/cover code in the original data set.
- Check that the Anderson-type land use code corresponds to the correct PCRAMMET code for the three meteorological stations using data provided by GIS to ensure programs are functioning correctly.
- Check the extracted data against the original data on a SAMPSON CD to ensure the data-extracting program is functioning correctly (randomly selected lines of data from first three pages of the printout for each site). Verify all columns are correct. Verify all 5 years of data are included. Once surface data are downloaded, run a data QA/QC program (SQAQC) to identify missing data.
- Check to make sure that the missing data were filled in correctly.
- Check the PCRAMMET input files against input data tables to make sure the data were transferred correctly.
- Check the PCRAMMET warning and error files to make sure there were no error and warning messages.
- Check all modeling options and parameters in the input files.
- Conduct a reasonableness check to make sure the results make sense.

Once we completed the unitized modeling, we applied a post-processing program that multiplies an annual average composite site emission rate by the unitized deposition rate. A GIS was used in the post-process. Below is a list of the QA/QC activities in the post-processing step:

- Check that the ammonia deposition rates at each geographic coordinate matched those in the ISC output files.
- Check that the correct model CAFO was applied to each swine operation.
- Check that the correct emission factor was applied to each CAFO.
- Check that the total ammonia deposited at a point representing the surrounding Thiessen polygon equaled the deposition rate per square meter X number of square meters in the Thiessen polygon.
- Check that the total ammonia deposited reported by HUC agreed with manual calculations. RTI selected all deposition points within a HUC and summed them using the statistics tool in ArcGIS.
- Check that all CAFOs within 50 km of the study area were included in the total ammonia deposition calculations.

2.4 MODELING ASSUMPTIONS AND UNCERTAINTY

Because of the complexity of this study as well as data and resource limitations, there are inevitable sources of uncertainty in the air dispersion and deposition analysis described above. As a result, the analysis requires a number of modeling assumptions and simplifications. Below we highlight five key sources of model uncertainty and explain default assumptions for these areas. Proposed sensitivity analyses are also described, so that one can examine how model results are influenced by these default selections.

1. Land use—RTI used the average land use for each of three meteorological regions (Raleigh, Wilmington, and Norfolk). This assumption could be tested by selecting a smaller, more specific area's land use versus the average of a meteorological region.
2. Meteorological station—Some sites in the meteorological North Carolina Piedmont region could have been assigned to a Greensboro meteorological station; however, these were consolidated with Raleigh because of the similarity in weather patterns.

3. Ammonia transformation—ISC does not compute chemical transformation, for example, from ammonia gas to ammonium sulfate fine particulates. Particulates are known to transport farther than ammonia gas before deposition. Information on this transformation rate was found. A conservative approach was to assume 100 percent ammonia available for dispersion and deposition. The impact of this assumption could be tested by assuming a select percentage reduction in available ammonia due to transformation (e.g., 20 percent conversion to ammonium sulfate).
4. Circular site—This assumption could be tested by assuming the model site is another shape (e.g., square, rectangle, or a randomly selected irregular shape).
5. Flat terrain—The ISC program models an area source as flat terrain. It does not account for elevated receptors.

More detailed descriptions of modeling assumptions for CAFO acreage assignment and emission factors are in the respective sections of this chapter.

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3 Generation of Ammonium (NH_4^+) Salt Fine Particulate Matter

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3.1 BACKGROUND

Ammonia, as a gas, is known to dry deposit efficiently to wet surfaces and vegetation. Ammonia gas that is not wet or dry deposited is available for reaction with sulfuric, nitric, and hydrochloric acids present in the atmosphere to form ammonium (NH_4^+) sulfate, ammonium bisulfate, ammonium nitrate, and ammonium chloride molecules. These molecules exist in the atmosphere as aerosol particulates. The dominant inorganic secondary aerosols species in the atmosphere comprise a fraction of $PM_{2.5}$ (also known as PM_{Fine}, a particulate matter with aerodynamic diameter less than 2.5 µm) and include sulfate, nitrate, and ammonium (Ansari and Pandis, 1998). The presence, concentration, and physical form of atmospheric ammonia are important given the health concerns regarding exposure to fine particulate matter. Limited data suggest that $PM_{2.5}$ is more toxic to humans than the larger particulate species (PM_{10}) (Lippmann, 1998).

As shown in Chapter 2, implementation of environmentally superior swine waste management technologies may reduce ammonia and subsequently reduce the potential for ammonium salt aerosol formation (i.e., PM_{Fine}). Implementation of environmentally superior technologies would, therefore, not only benefit water quality via reduced atmospheric ammonia gas deposition but also human health via reduced levels of PM_{Fine}. In the spring of 2003, the Smithfield-Premium Standard Settlement Agreement's Technical Advisory Panel requested that RTI compute a general estimate of the benefits achieved by reducing the atmospheric ammonium salt concentration (PM_{Fine}) in the study area. To accomplish this, we first needed to estimate the baseline level of ammonium PM_{Fine} that CAFOs may generate. With that estimate, we can perform a benefits analysis.

This chapter addresses how the PM_{Fine} estimate was calculated and the outcome of modeling in the study area.

3.2 INITIAL ASSESSMENT

RTI reviewed a significant amount of literature to identify a conversion factor for ammonia gas to ammonium salt aerosol. We also contacted U.S. Environmental Protection Agency (EPA) experts regarding this matter, but, because of the complexity of the atmospheric chemistry of ammonium formation, a simple conversion factor was not available. (See Dennis [2003]; Edney [2003]; and Gipson [2003].) In the end, we arrived at an estimate that was based on two measurements from studies conducted in the North Carolina study area: Baek and Aneja (2003) and Robarge et al. (2002). Both studies suggested ammonia gas (NH_3) comprises more than 70 percent of total ammonia (NH_x) (i.e., NH_3 gas plus NH_4^+ salt aerosol in the atmosphere during all seasons).

3.2.1 Ammonia to Ammonium Conversion

Gas-to-particle conversion can be accomplished by condensation, which adds mass onto pre-existing aerosols, or by direct nucleation from gaseous precursors that form aerosols (Baek and Aneja, 2003). Gas-to-particle conversion strongly depends on the concentration of acid gases and water vapor in the atmosphere. Ammonia reacts with sulfuric, nitric, and hydrochloric acid gases to form aerosols such as ammonium sulfate, ammonium bisulfate, ammonium nitrate, and ammonium chloride. Ammonium salts formed by these reactions can exist as solid particles or liquid droplets depending on the amount of water vapor in the atmosphere. Ammonia preferentially reacts with sulfuric acid (H_2SO_4) to form ammonium bisulfate (NH_4HSO_4) and ammonium sulfate ($[NH_4]_2SO_4$) through the process defined by Eqs. (3.1) and (3.2):

$$NH_3\ (gas) + H_2SO_4\ (liquid) \rightarrow NH_4HSO_4\ (liquid) \qquad (3.1)$$

$$NH_3\ (gas) + NH_4HSO_4\ (aq) \rightarrow (NH_4)_2SO_4\ (solid\ or\ liquid) \qquad (3.2)$$

Ammonia can also undergo an equilibrium reaction with gas-phase nitric acid (HNO_3) in the atmosphere to form ammonium nitrate (NH_4NO_3) as shown in Eq. (3.3):

$$NH_3\ (gas) + HNO_3 \leftrightarrow NH_4NO_3\ (solid)\ or\ (liquid) \qquad (3.3)$$

The low vapor pressure of sulfuric acid (H_2SO_4) allows it to condense easily on particle and droplet surfaces. Because the rate of condensation depends on the amount of water vapor in the atmosphere, sulfuric acid is seldom found in the gas phase. However, nitric acid is much more volatile than H_2SO_4 and not likely to form particles by homogeneous or heteromolecular nucleation. Therefore, because of its volatility, particulate nitrate is believed to be lower in concentration than sulfate (Seinfeld and Pandis, 1998; Pacyna and Benson, 1996). However, particulate nitrate can be the dominant species in fine particulate matter when in sulfate-limited regimes.

3.2.2 Ammonium Concentrations in the Study Area

The measurements conducted by Baek and Aneja (2003) consisted of two measurement sites at a commercial swine operation in eastern North Carolina. The north site was located approximately 50 meters northeast of the swine waste storage and treatment lagoon, and the south site was located approximately 400 meters south-southwest of the waste lagoon. Both measurement sites were either on the farm or very close to the farm. Samples were collected using the annular denuder systems (ADS) from April to July 1998 at the north site and from April 1998 to March 1999 at the south site. The measured data were not only used for analyzing the general characteristics of ammonia, sulfuric acid, and nitric acid, but also for examining the gas-to-particle conversion between ammonia and acid gases.

Annual average ammonia concentration at the south measurement site was 17.89 $\mu g/m^3$. Annual average ammonium concentration at the same site was 1.64 $\mu g/m^3$. This shows that ammonia comprises more than 90 percent of total ammonia (NH_x) at or near the farm. From seasonal variations, the measured particulates (e.g., ammonium) showed larger peak concentration during summer, suggesting that the gas-to-particle conversion was efficient during summer.

The measurements by Robarge et al. (2002) were taken at the Clinton Horticultural Crops Research Station located approximately 5 km north and east of Clinton, North Carolina. Three swine production facilities are located between 1.5 and 3.2 km to the east/northeast and east/southeast of the site. In addition, three swine production facilities are between 3.2 and 5 km northwest of the site. Chemical and meteorological measurements were collected from October 1998 to September 1999. Chemical species were collected using an ADS, which was similar to what Baek and Aneja used in their measurements.

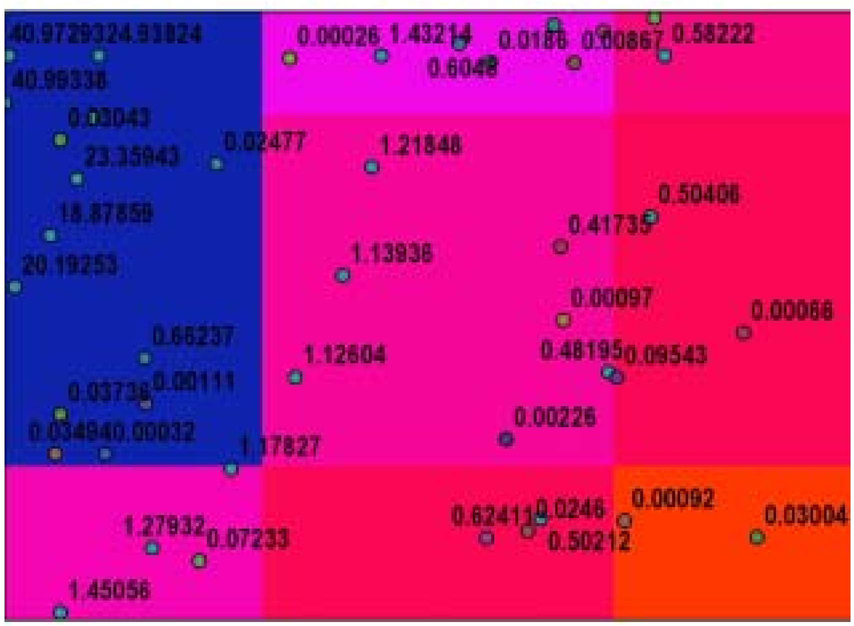
Unlike the measurement by Baek and Aneja at the swine operation site, Robarge's measurements represent a more ambient condition. Ammonia concentrations observed in this study follow a log normal distribution, with an annual mean ammonia concentration of 5.55 $\mu g/m^3$. Ammonia comprises more than 70 percent of the total ammonia NH_x during all seasons. The annual mean ammonium concentration from this measurement site was 1.44 $\mu g/m^3$.

The two studies show the sensitivity of the conversion rate of ammonia to ammonium in that the rate increases with the increase in ammonia concentration and the distance from the swine operation. Both measurements show ammonia and ammonium display seasonal cycles with higher concentrations occurring during the summer for both species. The ratio of ammonia to total ammonia (NH_x) from Robarge's ambient measurements) was greater than 70 percent, while the same ratio from Baek and Aneja's measurements (adjacent to the swine operation) was greater than 90 percent. This suggests more ammonia was converted to ammonium during its transport in atmosphere.

3.3 APPROACH FOR ESTIMATING PM_{Fine} EFFECTS IN STUDY AREA

The ISCST3 model run by RTI for gaseous ammonia deposition analysis generated unitized ammonia gas air concentration (postdeposition) at 1,080 discrete X,Y locations for each of 12 model farms (see Figure 3-1). There were four CAFO sizes (50, 100, 260, and 500 acres) at three different meteorological regions (Raleigh, Wilmington, and Norfolk) resulting in the 12 model CAFOs. (See Chapter 2 for further details.) One of the model CAFOs was assigned to each of the approximately 2,295 swine operations in the study area and placed at the actual geographic coordinates of the operation. The actual ammonia gas concentration value at each of the 1,080 X,Y locations was calculated by multiplying the unitized concentration by the emission factor for the CAFO. The ambient ammonia gas concentration remaining at each deposition point was calculated after the unitized deposition value at that point was subtracted.

Figure 3-1. Raster Grid Cell with Contributing CAFO Points



Note: Each point location from each CAFO contains an ammonia concentration value. A representative value is computed for each cell for each CAFO, and then the concentrations from all CAFOs are added together.

Once the actual ambient ammonia air concentration value was computed, a Triangular Irregular Network (TIN) was created for each CAFO using a geographic information system (GIS). This TIN was converted to a raster dataset (cell based) with the value field containing the ammonia concentration value. Each rasterized CAFO was then added to a master raster dataset, so that the ammonia concentration value was additive for each cell. Summary statistics (such as the mean) of ambient ammonia gas concentrations emanating from swine operations were then generated for the master raster dataset on a county-by-county basis (see Table 3-1). Based on measurements from the two studies previously discussed, RTI decided to apply Robarge's ambient monitoring findings and assume that ammonia gas comprises 70 percent of total ammonia species and that ammonium salt (NH_4^+) comprises 30 percent of total ammonia species in this analysis. With this 30 percent value, we calculated a county annual average ammonium salt (PM_{Fine}) concentration by multiplying the county average of ambient ammonia gas by 30 percent.

It should be noted that this exercise estimates only ammonium salt PM_{Fine} resulting from swine operation emissions. It does not estimate background ambient PM_{Fine} resulting from other emission sources.

3.4 PM_{Fine} ESTIMATION RESULTS

Table 3-2 lists the estimated baseline ambient ammonium (NH_4^+) concentrations attributable to swine operations for each of the counties in the study area. These data serve as input to the integrated benefits analyses described in Chapters 6 and 7.

The average ammonium concentration inside the study area was estimated as 0.592 $\mu g/m^3$. The county with the maximum estimated annual average ammonium salt PM_{Fine} concentration was Duplin County at 3.576 $\mu g/m^3$.

(For a point of reference, North Carolina's $PM_{2.5}$ (or PM_{Fine}) standard is 15.0 $\mu g/m^3$ [NCAC 2D.410(a)]).

Table 3-2 also compares the modeled ammonium PM_{Fine} concentration to the ambient PM_{Fine} measured by North Carolina in its monitoring program from July 1999 to December 2001. Duplin County's averaged monitored Pm_{Fine} for that period was 12.6 $\mu g/m^3$, implying that ammonium PM_{Fine} originating from swine operations may comprise 28 percent of the county's ambient PM_{Fine}. Duplin County's monitored ambient concentration of 12.6 $\mu g/m^3$ is below the 15.0 $\mu g/m^3$ standard.

Table 3-1. Estimated Average Annual Swine-Generated Ammonia Gas Concentration (µg/m^3) Remaining after Deposition before the Gas Reacts to Form Ammonium Salts (descending order)[a]

North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonia Gas (NH_3) Concentration (µg/m^3)	North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonia Gas (NH_3) Concentration (µg/m^3)
Duplin	11.919	Randolph	0.345
Sampson	10.382	Franklin	0.324
Greene	6.904	Warren	0.303
Wayne	6.383	Person	0.288
Lenoir	5.169	Moore	0.272
Bladen	4.071	New Hanover	0.170
Jones	3.049	Tyrrell	0.164
Johnston	2.175	Wake	0.158
Pitt	2.150	Montgomery	0.129
Pender	2.074	Lee	0.125
Cumberland	2.044	Vance	0.110
Edgecombe	1.623	Rockingham	0.100
Onslow	1.475	Pamlico	0.072
Wilson	1.360	Hyde	0.072
Harnett	1.083	Guilford	0.063
Robeson	1.075	Chatham	0.062
Halifax	0.946	Orange	0.051
Nash	0.920	Alamance	0.036
Craven	0.756	Granville	0.027
Beaufort	0.638	Carteret	0.023
Washington	0.561	Caswell	0.018
Martin	0.554	Forsyth	0.014
Columbus	0.529	Durham	0.013
Brunswick	0.497	Dare	0.002
Hoke	0.475		

[a]Estimates are based on results of modeled ammonia gas deposition as reported in Chapter 2. That modeling effort used the NC DWQ Survey (of swine operations) database, NCDA&CS 1997 Ag Census data and published emission factors in conjunction with the US EPA's ISCST3 model, and NWS meteorological data for three North Carolina regions.

Table 3-2. Estimated Average Annual Ammonium (NH_4^+) Concentration ($\mu g/m^3$) for Counties in the Study Area (descending order)

North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonium (NH_4^+) Concentration [a] ($\mu g/m^3$)	Weighted Annual Arithmetic Mean Ambient Concentration from All Sources[b] ($\mu g/m^3$)	North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonium (NH_4^+) Concentration [a] ($\mu g/m^3$)	Weighted Annual Arithmetic Mean Ambient Concentration from All Sources[b] ($\mu g/m^3$)
Duplin	3.576	12.6	Randolph	0.104	NA
Sampson	3.115	NA	Franklin	0.097	NA
Greene	2.071	NA	Warren	0.091	NA
Wayne	1.915	15.3 (S)	Person	0.086	NA
Lenoir	1.551	12.3 (S)	Moore	0.081	NA
Bladen	1.221	NA	New Hanover	0.051	NA
Jones	0.915	NA	Tyrrell	0.049	NA
Johnston	0.652	NA	Wake	0.047	15.3
Pitt	0.645	13.7 (S)	Montgomery	0.039	NA
Pender	0.622	NA	Lee	0.038	NA
Cumberland	0.613	15.4 (S)	Vance	0.033	NA
Edgecombe	0.487	NA	Rockingham	0.030	NA
Onslow	0.442	12.1	Pamlico	0.022	NA
Wilson	0.408	NA	Hyde	0.022	NA
Harnett	0.325	NA	Guilford	0.019	16.3
Robeson	0.323	NA	Chatham	0.019	13.4 (S)
Halifax	0.284	NA	Orange	0.015	14.3
Nash	0.276	NA	Alamance	0.011	14.6 (S)

(continued)

Table 3-2. Estimated Average Annual Ammonium (NH_4^+) Concentration ($\mu g/m^3$) for Counties in the Study Area (descending order) (continued)

North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonium (NH_4^+) Concentration [a] ($\mu g/m^3$)	Weighted Annual Arithmetic Mean Ambient Concentration from All Sources[b] ($\mu g/m^3$)	North Carolina Study Area County	Estimated Average Annual Swine-Generated Ammonium (NH_4^+) Concentration [a] ($\mu g/m^3$)	Weighted Annual Arithmetic Mean Ambient Concentration from All Sources[b] ($\mu g/m^3$)
Craven	0.227	NA	Granville	0.008	NA
Beaufort	0.191	NA	Carteret	0.007	NA
Washington	0.168	NA	Caswell	0.005	14.5 (S)
Martin	0.166	NA	Forsyth	0.004	16.1
Columbus	0.159	NA	Durham	0.004	15.3
Brunswick	0.149	NA	Dare	0.001	NA
Hoke	0.142	NA			

[a]Estimates are based on results of modeled ammonia gas deposition (as report in Chapter 2) in conjunction with the findings of Robarge et al. (2002) which led to an assumption that ambient air in eastern North Carolina has a 70% ammonia (NH3) gas to 30% ammonium salt aerosol ratio.

[b]Source: North Carolina Division of Air Quality's ambient PM2.5 ambient monitoring data analysis for the period from July 1999 to December 2001. The state is monitoring PM2.5 to determine if a county exceeds the new 15.0 µg/mg ambient standard, signifying nonattainment. The notation "S" indicates that the mean is suspect (albeit valid) because at least one calendar quarter had less than 75 percent of the expected number of valid samples. http://daq.state.nc.us/monitor/datapm2ppt5.

NA = Not available.

Counties in the study area monitored at nonattainment with the new standard are Wayne, Cumberland, Wake, Guilford, Forsyth, and Durham. Wayne County is ranked fourth in the modeled swine-generated ammonium PM_{Fine}, which is estimated to comprise 12 percent of the ambient PM_{Fine} concentration. Cumberland County (ranked eleventh in modeled swine-generated ammonium PM_{Fine}) is estimated to comprise 4 percent of the county's ambient PM_{Fine} concentration.

Among the 11 counties ranked by modeling to contribute the most swine-generated ammonium PM_{Fine} in the study area, five have ambient PM_{Fine} monitoring data available. Data indicate the following:

County	**Percentage of swine-generated ammonium contribution to ambient PM_{Fine} concentration**
Duplin	28%
Wayne	12%
Lenoir	13%
Pitt	5%
Cumberland	4%

3.5 QUALITY ASSURANCE/QUALITY CONTROL AND VALIDATION

The final product of this step was a series of ammonia concentration values (and, therefore, PM_{Fine} concentrations) by county. These results were verified by conducting the following activities:

- Comparing the concentration values of the X,Y locations in the ISC output files with the concentration values in the GIS point coverages.
- Checking that the correct emission factor was applied to each CAFO.
- Comparing the rasterized concentration values against the concentration values at the X,Y locations in the GIS point coverages.
- Making sure that all raster cells within the study area were given a value (making sure there were no no-data cell values).
- Spot checking to make sure that the values of the summed grid cells were equal to the contributions of all the individual CAFO grid cells.

- Checking that the mean ammonia concentration values by county agreed with the values calculated when doing the mean by hand. The grid cells that fell within the county were selected, and then the mean was calculated using the zonalstats command in ArcGIS. Values were randomly sampled to make sure that the mean was in the correct range.
- Comparing the modeled ammonium concentrations to the measured concentrations. The county-wide average modeled ammonium concentration was about 3.11 $\mu g/m^3$ for Sampson County if we assume ammonia comprises 70 percent of NH_x. Measurements were taken at the Clinton Horticultural Crops Research Station in Sampson County. The measurement site is located 1.5 km to 5 km from a few swine production facilities. The measured concentration was 1.44 $\mu g/m^3$. The modeled and measured concentrations are roughly the same magnitude.

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4 Surface Water Quality Effects of Changes in Nitrogen and Phosphorus Loadings

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In this chapter, we present our approach for and results from modeling nitrogen and phosphorus sources and stream delivery in river basins in eastern North Carolina. To estimate nitrogen and phosphorus stream loadings and concentrations under baseline conditions and with simulated reductions in both airborne and waterborne emissions, RTI compiled and integrated data for source terms and model algorithms and parameters for both swine and nonswine sources. Stream concentration predictions serve as a primary input to the recreation demand model presented in Chapter 6.

4.1 BACKGROUND

Previous studies have addressed the potential for surface water quality impacts from livestock manure sources in eastern North Carolina. For example, Kellogg (2000) concluded that the Cape Fear River basin is the highest priority river basin in the country, and the Neuse/Pamlico basins rank 15th, in which “EPA and USDA could be targeted first to quickly meet the goals of protecting watersheds from contamination by manure nutrients” (p. 8). This study documented risks and potential impacts from livestock manure, but stopped short of estimating actual surface water impacts.

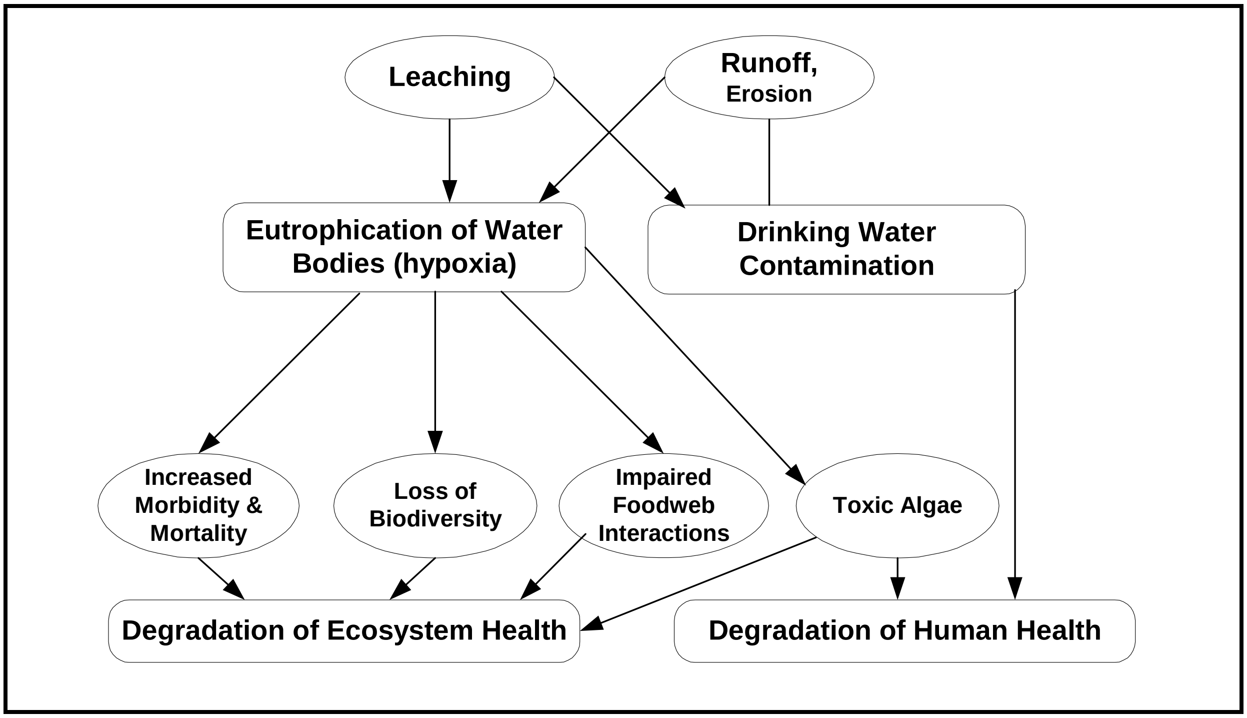
Consideration of impacts of manure nutrients on surface water starts with a conceptual understanding of the relevant processes and characteristics (see Figures 4-1 and 4-2).

The discharge of nutrients to water bodies is recognized as a predominant anthropogenic impact on coastal ecosystems and as the major cause of coastal eutrophication (Nixon, 1995; ESA, 2000). After reaching aquatic systems, nutrients may have many different effects on both planktonic and rooted aquatic plants, the extent of waters with low concentrations of dissolved oxygen, and other ecological processes.

Understanding of the complexity of responses by coastal ecosystems to human-caused nutrient enrichment has grown considerably in the past decade. Earlier conceptual models focused on direct responses of coastal waters, such as stimulation of phytoplankton blooms. The contemporary conceptual model reflects a growing awareness of the complexity of the problem, including recognition that the attributes of specific water bodies create enormous variations in their responses to nutrient loading, and that a cascade of direct and indirect effects can occur. Furthermore, experience has shown that appropriate management actions to reduce nutrient inputs can reverse some of the degradation caused by enrichment (ESA, 2000; NRC, 2000).

The temporal and spatial context of these impacts is complex, because surface water quality risks may be localized or regional, associated with infrequent or relatively catastrophic events, or longer-term, more chronic delivery of pollutants. An additional challenge in evaluating swine waste impacts is distinguishing swine effects from those associated with other activities in the watershed or river basin such as crop production, urban runoff, and wastewater discharge. The transport pathway, especially for agricultural nitrogen, can occur via surface or subsurface runoff (Figure 4-3) and through atmospheric emissions, transport, and deposition, as discussed in Chapter 2. Impacts can be realized relatively “near-field” or “far-field,” depending on a variety of factors.

Figure 4-1. General Pathway for Nutrient Impacts on Human and Ecosystem Health

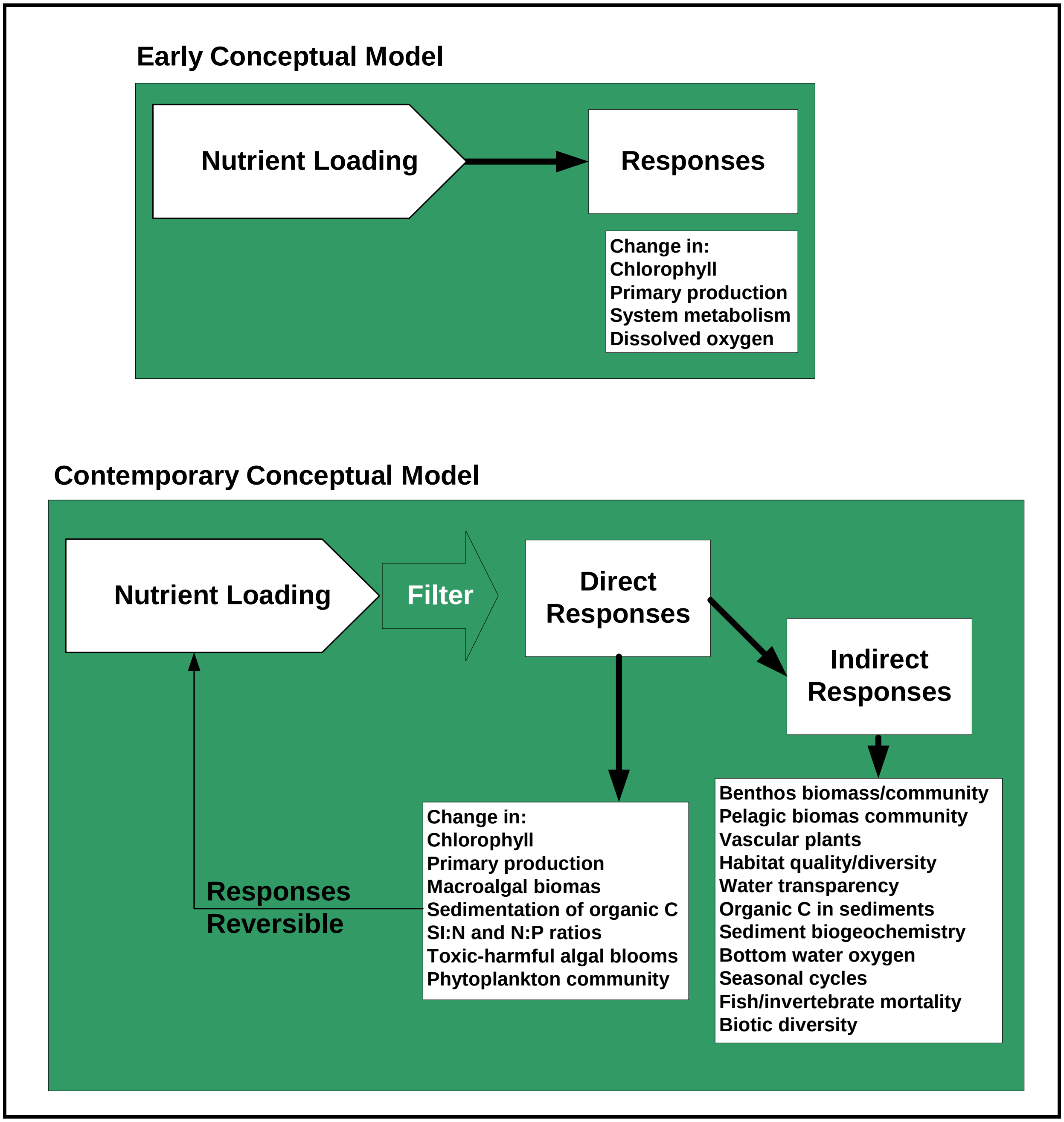


Source: Ecological Society of America (ESA). Fall 2000. "Nutrient Pollution of Coastal Rivers, Bays, and Seas." *Issues in Ecology, #7*. Available at <http://www.esa.org/sbi/sbi_issues/issues_pdfs/issue7.pdf>.

Excess nitrogen in waste, fertilizer, and precipitation can leach or run off as organic nitrogen, ammonium (NH_4^+) or nitrate (NO_3), and can also be transported atmospherically. Several chemical and biological reactions can transform much of the excess nitrogen in runoff to nitrate. Nitrate is also a major form of nitrogen that plants take up and convert back into organic nitrogen as plant tissue. Nitrate often moves via groundwater and exits terrestrial systems as base flow to the streams, although there may be a substantial lag time because groundwater flows much more slowly than surface water. Denitrification in the subsurface can transform the nitrogen to gaseous form, which then diffuses into the atmosphere.

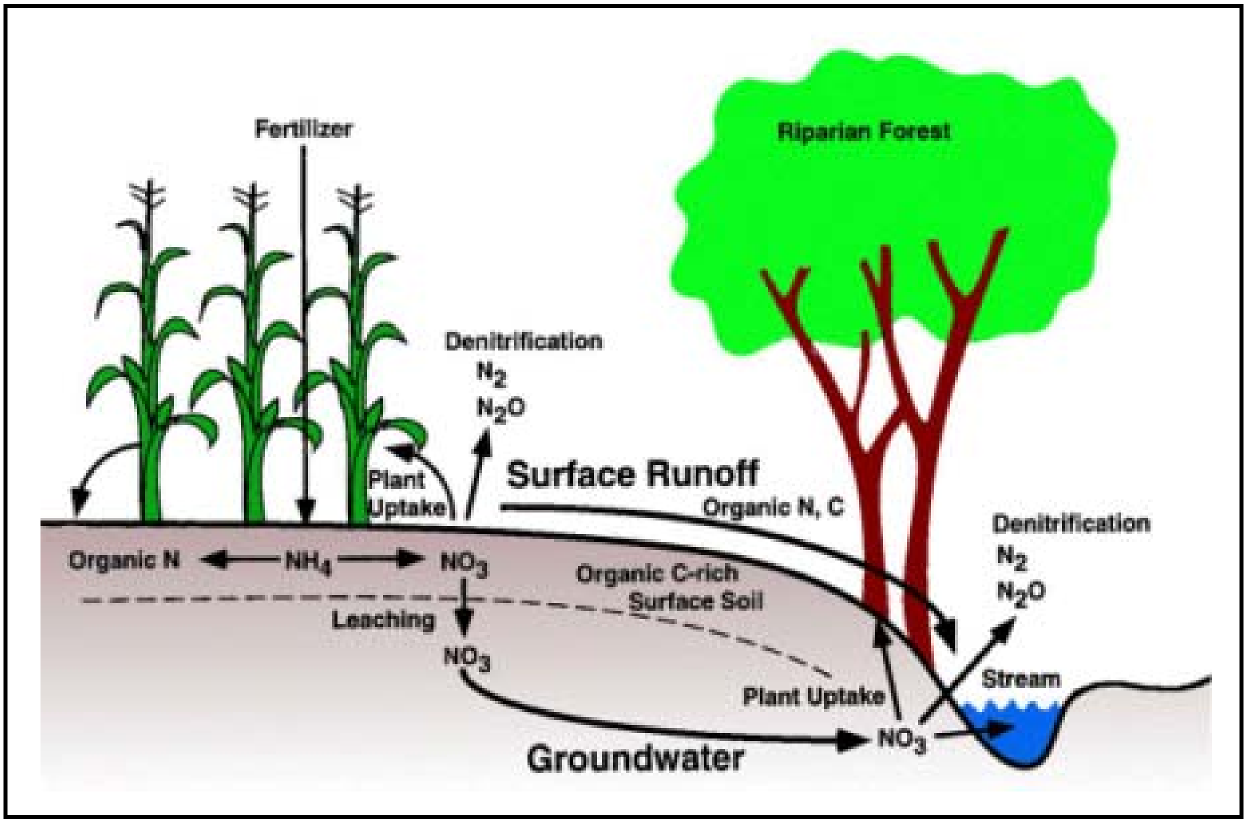
One of the challenges in managing nutrient inputs to coastal waters in North Carolina has been developing predictive frameworks for quantifying nutrient inputs and delivery from multiple sources and landscapes via complex natural systems. A considerable amount of relevant work has been completed at the field, small watershed, or river basin scale or in association with more generalized landscape and agricultural impacts or programmatic efforts (see, for example,

Figure 4-2. Early and Contemporary Conceptual Models for the Relationship Between Nutrient Loading and Ecological Responses



Source: Ecological Society of America (ESA). Fall 2000. "Nutrient Pollution of Coastal Rivers, Bays, and Seas." *Issues in Ecology, #7*. Available at <http://www.esa.org/sbi/sbi_issues/issues_pdfs/issue7.pdf>.

Figure 4-3. Simple Conceptual Model of Nitrogen Movement in the Environment



Source. Agricultural Nitrogen Pathways, U.S. Environmental Protection Agency (EPA), Office of Research and Development, National Exposure Research Laboratory. December 1999. *Mid-Atlantic Stressor Profile Atlas*. EPA/600/C-99/003. Washington, DC. Available at <http://www.epa.gov/reva/StressorAtlas/sa1.pdf>.

McMahon, Qian, and Roessler [2002], [In press]; McMahon and Woodside, [1997]; NCDWQ [1999]; RTI [1995, 1997]; Spruill et al. [1996]; WRRI [2000]; EPA [2003]). However, development of a comprehensive analytical framework focusing on surface water impacts specifically from swine operations on broad spatial scales in North Carolina has not been completed to date.

It is beyond the scope of this project to consider the entire spectrum of relationships between swine waste and surface waters or to model the details of the cycling and speciation of nutrients. Rather, the focus is on the aggregated environmental movement of total nitrogen and total phosphorus from swine operations downstream through natural stream drainage pathways to the estuarine interface. Because nitrogen releases to the atmosphere can eventually affect surface waters, focusing on nitrogen allows for the integration of atmospheric and surface water assessments. Finally, and most importantly for this study, focusing on nitrogen and phosphorus can provide meaningful information for quantifying economic benefits associated with potential reductions in future surface water inputs.

4.2 TECHNICAL APPROACH

The study area is defined by the river basin boundaries of the Tar-Pamlico, Neuse, Cape Fear, White Oak, and New River basins (Figure 4-4).

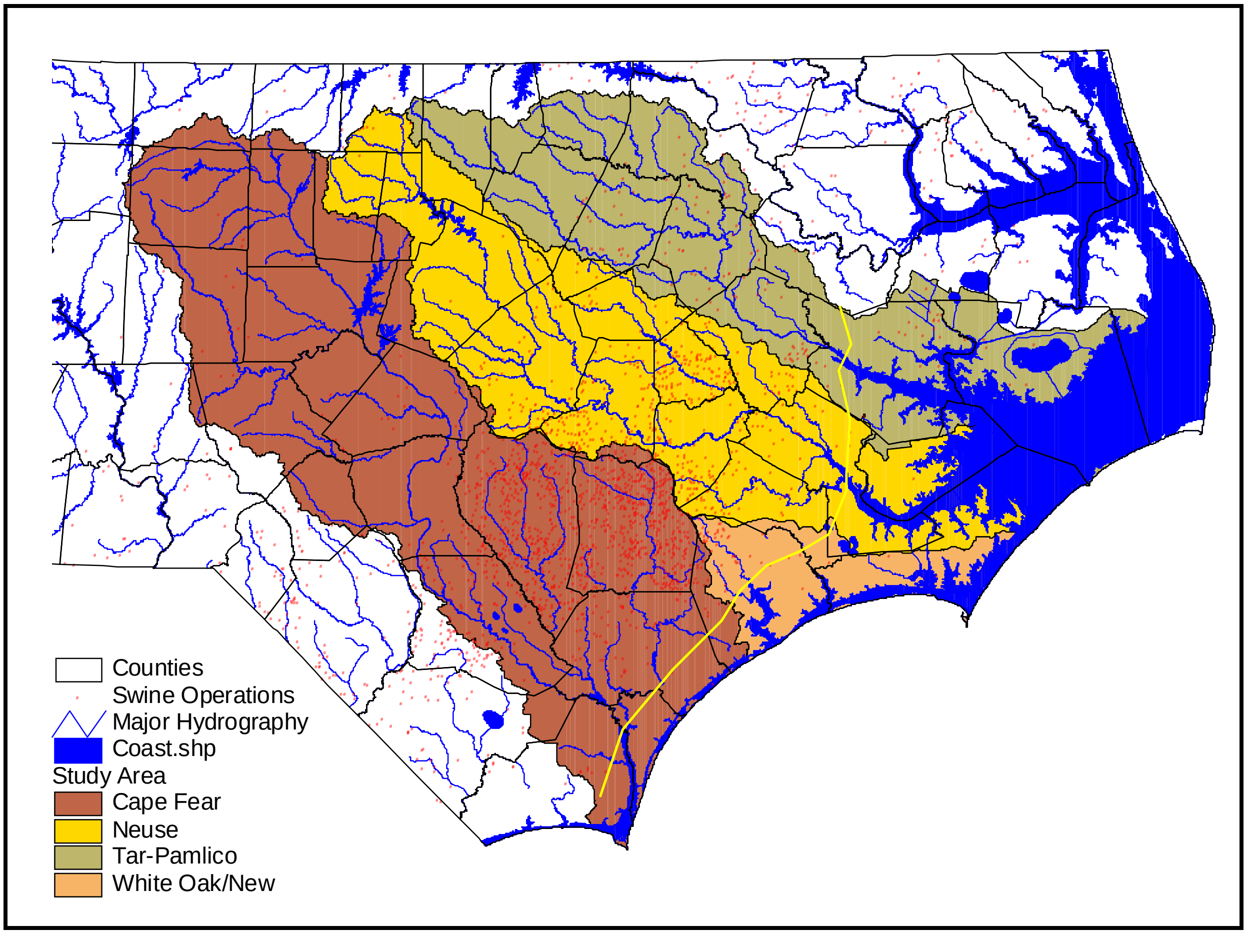
To address nitrogen and phosphorus release and movement via swine waste management, we integrated a broad array of data and analytical components into a systematic framework. We drew upon both state and national studies and data resources, as discussed below. The temporal context of the analysis works from a baseline of "current conditions," which we created by obtaining the most up-to-date data available for the respective components. The study area is defined by the river basin boundaries of the Tar-Pamlico, Neuse, Cape Fear, White Oak, and New River basins (Figure 4-4), the watersheds most directly affected by swine operations in the state. The downstream extent of the model application approximated waters draining to the tidal boundary or "head of estuary." We also estimate direct deposition of atmospherically derived nitrogen to each of these estuaries. Modeling complex estuarine processes is beyond the scope of this study.

4.2.1 Stream Model Specification and Development

We developed a predictive modeling approach to estimate nutrient loading and delivery. Key steps in developing the approach included setting up a database that characterizes the stream network and routing system; predicting stream flow and channel hydraulics; creating input files of historical and projected wastewater treatment plant (WWTP) effluent characteristics and nonswine nonpoint source inputs; quantifying swine inputs and developing a methodology to determine delivery of swine inputs to surface waters; and modeling the instream attenuation of nitrogen and phosphorus. Each of these model components is summarized in Table 4-1 and discussed below. Technical details are provided in Appendix A.

The fundamental unit for model calculations was a stream reach. (A stream reach is a segment of a stream identified in the Reach File database, averaging approximately one mile in length.) Model outputs were reported at a 14-digit hydrologic unit (HUC) level, using watershed boundaries defined by agencies in North Carolina.

Figure 4-4. Map of the Surface Water Assessment Study Area



Note: The yellow line indicates the approximate downstream extent of the stream routing and delivery

(These watersheds have an average size of 31 square miles.) Primary model input terms were expressed as annualized mass loadings (kg/yr) of nitrogen and phosphorus, and output was expressed as both loadings as well as instream concentrations (mg/l). Loadings and concentrations were compiled for the watershed outlet reach for each of 565 HUCs, and subsequently used as inputs to the economic benefits analysis described in Chapter 6. The initial scope of 676 HUCs in the study area was reduced because the stream reach network could not be built for coastal HUCs with nondendritic drainage patterns as represented in the stream reach data.

Stream Network

A database of the study area watershed stream network was developed to form the structural and functional backbone of the

Table 4-1. Primary Model Specifications

Model Component	Approach
Spatial domain	One-dimensional advective stream model for single channel, natural drainage, free-flowing streams
Temporal domain	Steady state, annualized. Baseline source data from 1995-present
State variables	Stream flow Total phosphorus and total nitrogen
Stream flow	Calculated for each reach based on analysis completed as part of the development of the National Water Pollution Control Assessment Model (RTI, 2001a)
Hydraulics	Assume stable channel, channelized flow Include stream width, depth, sinuosity, slope, velocity, time-of-travel (RTI, 2001a)
Instream kinetics	First order decay, variable by stream flow, based on analyses by the U.S. Geological Survey (USGS)
Swine sources	Study-specific method based on ammonia modeling results; information provided by NCSU researchers (Norwood, 2002; 2003a,b); source-to-stream runoff delivery algorithm based on distance of facility to stream reach; and indirect ammonia deposition delivery based on watershed land cover
Nonswine sources	Point source data from EPA; runoff inputs using export coefficients and land cover; atmospheric inputs based on monitoring data and indirect deposition delivery based on watershed land cover
Computational element	Stream reach based on EPA Reach File Version 3 (RF3)

model. The stream network database required spatial referencing to associate tributaries and reaches with other locational data such as swine land application areas, watershed outlets, wastewater treatment plant outfall, and instream monitoring sites and to estimate travel distances and times. It also required routing capabilities, or "navigational intelligence," to determine the direction of flow at stream junctions. Based on these functional requirements and recent studies, EPA's Reach File Version 3 (RF3) was chosen as the principal data source and framework for the stream network. A total of almost 25,000 stream reaches were included in the enhanced RF3 database created for the project.

Streamflow and Hydraulics

Streamflow and hydraulic characteristics were modeled based on methods described in RTI (2001a). Time-of-travel estimates were a key requirement for estimating nutrient loss or attenuation during delivery via the stream network. We estimated time of travel based on standard engineering methods for stable stream channels (see Appendix A). These methods involved estimating stream depth, width, roughness, and sinuosity for each reach. In combination with flow and reach length data, we use these hydraulic parameters to calculate velocity and time of travel.

Sources of Nitrogen and Phosphorus

The analysis comprehensively addresses inputs of nitrogen and phosphorus from numerous sources and quantifies swine sources independently from other sources. Further differentiation was warranted because of the emerging understanding of both runoff and atmospheric pathways for delivery of nitrogen from swine operations to surface waters. Therefore, we developed methods to estimate watershed inputs for swine facilities for both runoff of nitrogen and phosphorus from land application of waste and deposition of ammonia. Additionally, we estimated municipal and industrial wastewater sources and runoff of both nitrogen and phosphorus and deposition of nitrogen from nonswine nonpoint sources.

Inputs from Swine Facilities. To address nitrogen inputs from swine facilities via runoff, we derived land application rates (kg N/yr) for each facility using facility inventory data received from the North Carolina Division of Water Quality and methods described in the ammonia emissions and deposition modeling assessment (Chapter 2). Sprayfield application rates were adjusted based on volatilization assumptions included in the ammonia air emissions estimation. Fifty percent of liquid nitrogen applied was assumed to be harvested in crops and not available for runoff (Norwood, 2002). The method for identifying land areas receiving waste is discussed in Appendix A. Nitrogen delivery from sprayfields to edge of field was calculated based on methods used by Schwabe (1996) and Norwood (2003a), as described in Appendix A. We estimated edge of field to stream reach delivery using methods described in RTI (2001a), with an adjusted default global rate of 0.4/km employed based on qualitative calibration (or 40 percent loss per kilometer).[1] The principal determinant in field-to-stream delivery, therefore, was the estimated distance from the land cover cell associated with the facility to the nearest stream reach in the RF3 database.

To address atmospheric nitrogen inputs from swine facilities, we compiled output from the ammonia emissions and deposition modeling (Chapter 2) for each 14-digit watershed. Deposition rates were assumed to be uniformly distributed within each watershed.

Direct deposition onto water surfaces for freshwater was calculated based on the watershed's deposition rate and the area in the watershed identified as water based on land cover data. Indirect deposition was calculated based on an assumption of delivery (pass through) rates for different land cover categories (Table 4-2). This assumption was based on two primary sources (EPA, 2001; Valigura et al., 1996) and secondary sources (Atkinson, 2003; Chesapeake Bay Program et al., 2000; Fisher and Oppenheimer, 1991; Hinga, Keller, and Oviatt, 1991; Linker et al., 1999; Tyler, 1988). Valigura et al. (1996) point out that there is little information for determining this quantity. There is admittedly significant possibility for variability with respect to delivery based on a host of ecological considerations (see inset).

To address phosphorus inputs from swine facilities, we derived land application rates (kg P/yr) for each facility using facility inventory data received from the North Carolina Division of Water Quality and information provided by Norwood (2002; 2003b). This information indicated that a substantial portion of the phosphorus mass generated in feeding houses is either unaccounted for or accumulating in sludge (Table 4-3).

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[1] This term was the most difficult model term to determine in the methodology because of inherent environmental variability and complexity and the inability to develop methodology and supporting data within the project constraints. It is likely that a wide range of field to stream delivery rates occur based on factors besides distance to stream such as soil characteristics, riparian buffering and down gradient land use and cover, drainage patterns, groundwater recharge and discharge, and topography. RTI proposes to complete an additional model run with a higher rate prior to delivery of the final project report, and will update this chapter as deemed appropriate after review of the model results.

Table 4-2. Export Coefficients (kg/ha/yr) Used for Nonswine Nonpoint Source Runoff and Indirect Deposition Pass Through Rates (for Ammonia from Swine Operations and Nonswine Nitrogen Deposition)

Land Cover	Nitrogen	Phosphorus	Deposition Pass Through Rate	Percentage of Total Area
Forest (Piedmont)	2.75	0.25	0.1	29.9%
Forest (Coastal Plain)	2.75	0.2	0.1	21.6%
Row crops (Coastal Plain)	12	1.2	0.3	14.5%
Woody wetlands (Piedmont)	3.5	0.3	0.1	8.9%
Row crops (Piedmont)	12	1.6	0.3	5.7%
Woody wetlands (Coastal Plain)	2.75	0.3	0.1	5.5%
Pasture and hay (Piedmont)	7	1.1	0.2	4.8%
Low intensity residential	7	0.8	0.3	2.6%
Pasture and hay (Coastal Plain)	7	0.9	0.2	2.6%
Commercial and industrial	11	1.5	0.8	1.2%
Transitional (Coastal Plain)	7	1	0.5	0.8%
High intensity residential	9	1.1	0.5	0.7%
Urban recreational grasses	7	1.1	0.2	0.3%
Transitional (Piedmont)	7	1.5	0.5	0.3%
Emergent herbaceous vegetation	4	1	0.7	0.3%
Bare rock, quarries, sand, strip mines	10	0.5	0.8	0.2%

The fate and transport of nitrogen deposited on terrestrial systems is complex. Multiple factors, including ecosystem type (agricultural/forested/urban; upland/riparian), other air pollutants, soil chemistry, climate, history of nitrogen inputs, plant species differences in nitrogen demand and tolerance, and plant competition complicate and preclude definitive and widely applicable conclusions. For upland forests, many systems are nitrogen deficient and can therefore assimilate and benefit from additional inputs, up to perhaps 20 to 30 kg/ha yr. However, increased deposition can also alter fluxes and storage, soil chemistry, physiology, and community dynamics. The canopy is thought to "consume" roughly 40 to 50 percent of atmospheric input as evidenced by throughfall fluxes and can also indirectly affect availability of soil nitrogen. One specific concern is the nutrient imbalance resulting from preferential ammonia uptake and/or nitrate leaching. Nitrate leaching occurs more frequently when annual deposition rates (total nitrogen) exceed 10 kg/ha. Ammonium immobilization by plant roots or microorganisms and nitrification contribute to soil acidification, which can affect the export of nitrogen as well as the soil fertility and ecosystem health. On a river basin scale, much of the nitrogen input is thought to be denitrified and released to the atmosphere as nitrogen gas or nitrous oxide. There is likely a large spatial variability in the rates and occurrence of denitrification, as influenced by natural and anthropogenic factors such as topography, soils, climate, impoundments, channelization and draining, and alteration of riparian and wetland systems. Recent attention has focused on the ability of forested filter zones to remove 90 percent or more of nitrogenous compounds from surface runoff (see, for example, Verchot, Franklin, and Gilliam [1997]; NCSU [1997]).

Table 4-3. Phosphorus Swine Production and Application Rates (lb P/yr)

Facility Type	Unit	Barn Rate	Sprayfield Rate	Sludge Rate	Application Rate[a]	Percentage Accounted[b]
Farrow to feeder	Sow	7.12	1.28	5.406	6.686	93.90%
Farrow to finish	Sow	11.15	1.55	6.509	8.059	72.28%
Farrow to wean	Sow	40.25	6.19	17.67	23.86	59.28%
Feeder to finish	Animal	2.39	0.12	0.374	0.494	20.67%
Wean to feeder	Animal	4.95	0.58	1.871	2.451	49.52%

Note: The barn rate is the quantity of phosphorus in manure produced. The sprayfield rate is the quantity of phosphorus in spray applied. The sludge rate is the quantity assumed to accumulate in lagoon sludge. The application rate was used as the default rate for modeling and was calculated as the sum of the sprayfield and sludge rates. Percentage accounted is the quantity of phosphorus generated accounted for in the application rate. A “worst case” modeling scenario was also run in which all nutrients generated (barn rate) were assumed to be applied.

[a] Sprayfield Rate + Sludge Rate

[b] Application Rate/Barn Rate

Source: Norwood, F.B., North Carolina State University, Department of Agricultural Resource Economics. Personal correspondence with Randall Dodd. November 26, 2002.

For this study, the following assumptions were applied based on this information: sludge in lagoons is assumed to be at “steady state” (no net long-term accumulation of phosphorus in the lagoon), no net loss of phosphorus to air or groundwater is assumed, and both effluent and sludge are assumed to be applied in the land cover cells associated with the facility. A sensitivity analysis run was also completed to estimate potential loadings associated with environmental release of all phosphorus estimated to be generated at the barn. (The framework provided allows any of these assumptions to be easily modified.)

Seventy percent of the phosphorus applied was assumed to be harvested in crops and not available for runoff (Norwood, 2002). Phosphorus delivery from swine sources to edge of field was calculated based on methods used by Schwabe (1996) and Norwood (2003a), as described in Appendix A. This analysis relied on Universal Soil Loss Equation calculations completed by county and crop and assumed primary delivery of phosphorus in association with sediment. Edge of field to stream reach delivery was estimated using methods described in RTI (2001a). As with nitrogen, the principal determinant in field-to-stream delivery was the estimated distance and, hence, time of travel from the land cover cell(s) associated with the facility to the nearest stream reach.

Inputs from Nonswine Sources. We estimated wastewater inputs by using annualized effluent data for flow and nitrogen and phosphorus from each of the wastewater treatment plants in the study area based on monthly data from 1990 to 1999 compiled by EPA (2001). Annual loads were input into associated stream reaches.

The primary considerations in determining nonswine nonpoint source inputs are the large spatial area and the need to distinguish, on agricultural lands, swine inputs independently from other inputs. Nutrient export coefficients were derived based on several primary studies (RTI, 2001b; Dodd and McMahon, 1992; Linker et al., 1999; Alexander et al., 2001) (see Table 4-2) and then applied to land cover data to estimate nonpoint source runoff inputs. We estimated land cover in each land cover class for each 14-digit watershed in the study area using the USGS National Land Cover Dataset, each 14-digit watershed was assigned to one of three ecoregions occurring in the area, and export coefficients were estimated based on the three primary studies for each land cover class for each of the three ecoregions. The total nutrient loading was calculated as the product of the total area in each land cover class in each watershed and the associated export coefficient.

For nonswine atmospheric nitrogen inputs, we developed a method using available wet and dry deposition data from 1996 to 2000 from ambient monitoring sites (see Table 4-4), along with spatial interpolation.[2] Data were compiled for reduced (NH_4^+), oxidized (NO_3^- + HNO_3), and organic nitrogen within or near the study area. We calculated total deposition as the sum of wet and dry deposition. Several of the stations exhibited possible effects from swine operations based on a comparison of monitoring data and atmospheric modeling results; these stations were not used in calculating background nitrogen deposition rates. More abundant data were available for wet deposition (15 sites) as compared to dry deposition (three sites). Geospatial processing was used to create GIS coverages of 14-digit watershed deposition rates. No attempt was made to account for nitrogen deposition not reflected in this monitoring data or the variable bioavailability associated with the different species. To minimize double counting of deposition

[2] A previous study completed by Cowling et al. (1998) was considered but not used because of the earlier emphasis (data from 1989 and 1994) of the study.

Table 4-4. Ambient Nitrogen Deposition Data Used to Estimate Background Deposition (average value from 1996–2000 monitoring)

				Measured Deposition ($mg/m^2/yr$)					Modeled Deposition ($mg/m^2/yr$)		
				Wet			Dry			Wet	Percent of
Station ID	Location	Lat	Lon	NH_4^+	NO_3^-	DON	NH_4^+	NO_3[a]	14-digit HUC	NH_3	Measured[f]
1[b]	NCSU	35.78	–78.70	348.5	246.6	311.2			3020201090010	1.80	0.5
2[b]	Wake Co.	35.71	–78.67	336.7	276.2	286.9			3020201110020	2.60	0.8
3[b]	Clayton	35.65	–78.50	343.9	276.5	292.8			3020201110050	7.71	2.2
4[b]	Wilson	35.70	–77.95	263.9	218.6	196.6			3020203020030	11.70	4.4
5[b]	Smithfield	35.52	–78.35	352.2	308.2	330.7			3020201100050	13.28	3.8
6[b,c]	Goldsboro	35.33	–77.97	428.7	479.7	557.1			3020202010022	48.46	11.3
7[b,c]	Kinston	35.22	–77.53	390.1	339.1	459.2			3020202050010	62.37	16.0
8[b,c]	Trenton	35.07	–77.35	411.4	381.2	508.8			3020204010070	26.24	6.4
9[b]	New Bern	35.12	–77.05	321.3	308.6	454.4			3020204020010	5.80	1.8
10[b]	Bayboro	35.15	–76.72	244.0	318.0	352.9			3020105010030	1.04	0.4
11[b,d]	Beaufort	34.88	–76.61	300.1	276.3	475.9	33.8	147.7	3020204050050	0.16	0.05
12[c]	Marston	35.02	–78.28	378.0	251.0				3030006110020	119.36	31.6
13[d,e]	Jordan Ck/Candor	34.97	–79.52	183.5	280.7		49.0	166.8			
14[e]	Lewiston	36.13	–77.17	188.4	246.0						
15[e]	Prince Edward	37.17	–78.31	161.8	266.1		40.8	188.5			

[a]Dry NO_3 estimated as sum of NO_3 and HNO_3.

[b]Source: Whithall D., and H.W. Paerl. 2001. "Spatiotemporal Variability of Wet Atmospheric Nitrogen Deposition to the Neuse River Estuary, North Carolina." *Journal of Environmental Quality* 30(5):1508-1515.

[c]Stations not used in estimating nonswine deposition because of likelihood of contributions from swine sources. Other stations' data spatially interpolated for entire study area to create GIS coverage of nitrogen deposition.

[d]Source: CASTNET (http://www.epa.gov/castnet/).

[e]Source: NADP (http://nadp.sws.uiuc.edu/).

[f]Modeled NH_3/NH^{4+} (wet and dry) X 100

associated with swine operations, data from several of the stations located in relatively close proximity to areas of more concentrated swine activity that demonstrated higher percentages of modeled ammonia deposition rates to monitored ammonium deposition rates were not included in the analysis. We used an identical process to that described above for swine sources to estimate both direct and indirect deposition inputs to surface waters. Identical nitrogen pass through rates (Table 4-2) for indirect deposition were assumed for nonswine nitrogen sources.

Instream Nutrient Delivery

Experience from many water quality studies has demonstrated that a first-order decay process can be appropriate for simplified modeling of the physical, chemical, and biological processes affecting many constituents in water. This kinetic definition implies that the rate of loss of a constituent from the water column is a function of the initial concentration and time. Once we estimated time of travel, the primary challenge with this approach is selecting an appropriate rate of loss from the water column ("decay rate"). Empirically based rates from peer-reviewed studies (Alexander et al., 2001; Smith, Schwarz, and Alexander, 1997; McMahon Alexander, and Qian, In press) were used to estimate instream nutrient loss (see Table 4-5). These studies used spatially referenced regression equations to estimate instream delivery rates.

Software Platform

We implemented the methods described above by using industry standard GIS and relational database data preprocessing techniques to create input data tables in Microsoft Access 2000, developing model code in Microsoft Visual Basic, and conducting quality assurance tasks on all data and code. Model output was post-processed using standard routines in Microsoft Access, Excel, and ArcView 3.2.

4.2.2 Model Results

Results indicate that 90 percent of the predicted instream concentrations for watershed outlet reaches fall within the range of 1.5 and 5.1 mg/l (total nitrogen) and 0.09 and 0.24 mg/l (total phosphorus), with the distribution especially for total nitrogen

Table 4-5. Instream Decay Rates Employed

Stream Decay Rate	Value	Units	Reference
Total nitrogen (flow< 1.04 m^3/s)	0.05	1/km	McMahon, Qian, and Roessler (2002)
Total nitrogen (flow> 1.04 m^3/s)	0.002	1/km	McMahon, Qian, and Roessler (2002)
Total phosphorus (flow< 28.3 m^3/s)	0.268	1/day	Smith, Schwarz, and Alexander (1997)
Total phosphorus (28.3 m^3/s<flow<283 m^3/s)	0.0956	1/day	Smith, Schwarz, and Alexander (1997)
Total phosphorus (flow> 283 m^3/s)	0.3586	1/day	Smith, Schwarz, and Alexander (1997)
Total nitrogen (Jordan and Falls Lakes)	18.8	m/yr	McMahon, Qian, and Roessler (2002)

more skewed towards the lower end of the range.[3] This distribution is generally indicative of eutrophic conditions, based on stream classification work by Dodds, Jones, and Welch (1998). We compiled model outputs and aggregated them to assess both spatial patterns and relative contributions from the different source categories considered. Total nitrogen and phosphorus loading results for all sources (swine and nonswine) are summarized in Figures 4-5 through 4-10. With default input and delivery assumptions, we estimate swine waste accounts for 30 percent of the nitrogen and 11 percent of the phosphorus loading to coastal waters from inland, free-flowing streams and rivers in the study area.

Sixty-two percent of the swine delivery of nitrogen to free-flowing surface waters is estimated to occur via direct runoff, with the remainder through atmospheric deposition of ammonia upstream from estuarine waters.

Sixty-two percent of the swine delivery of nitrogen to free-flowing surface waters is estimated to occur via direct runoff, with the remainder through atmospheric deposition of ammonia upstream from estuarine waters. Impacts from swine facilities are most pronounced in a concentrated area in the upper Northeast Cape Fear and upper Black River basins in Duplin and Sampson counties (and to a lesser extent in adjacent watersheds in Bladen and Cumberland counties).

[3] The distribution for instream nitrogen concentrations is slightly higher than that reported by McMahon et al., as with the basin yields.

Figure 4-5. Total Nitrogen Loading by 14-Digit Watershed (kg/yr): All Sources

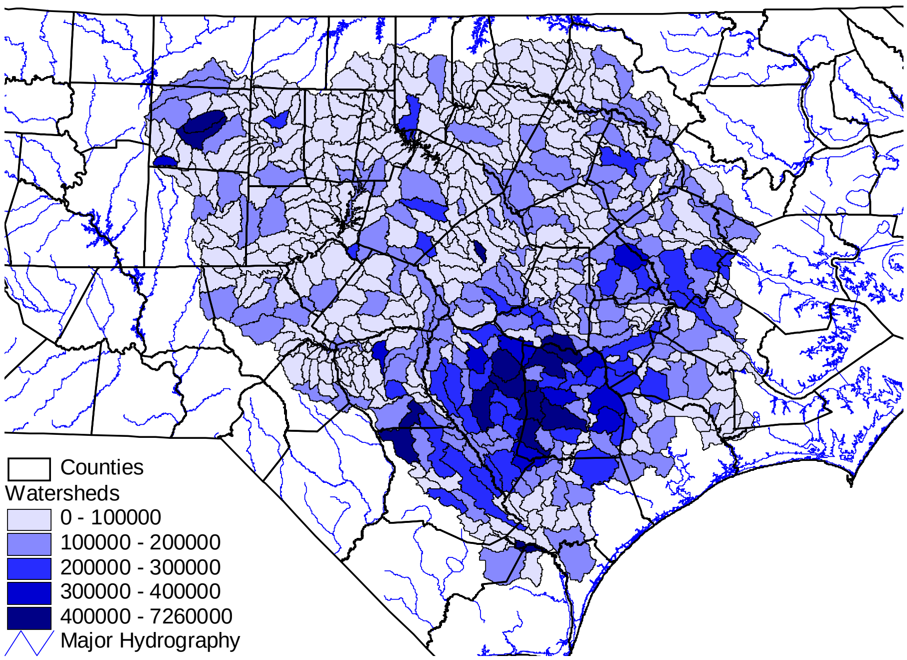


Figure 4-6. Total Phosphorus Loading by 14-digit Watershed (kg/yr): All Sources

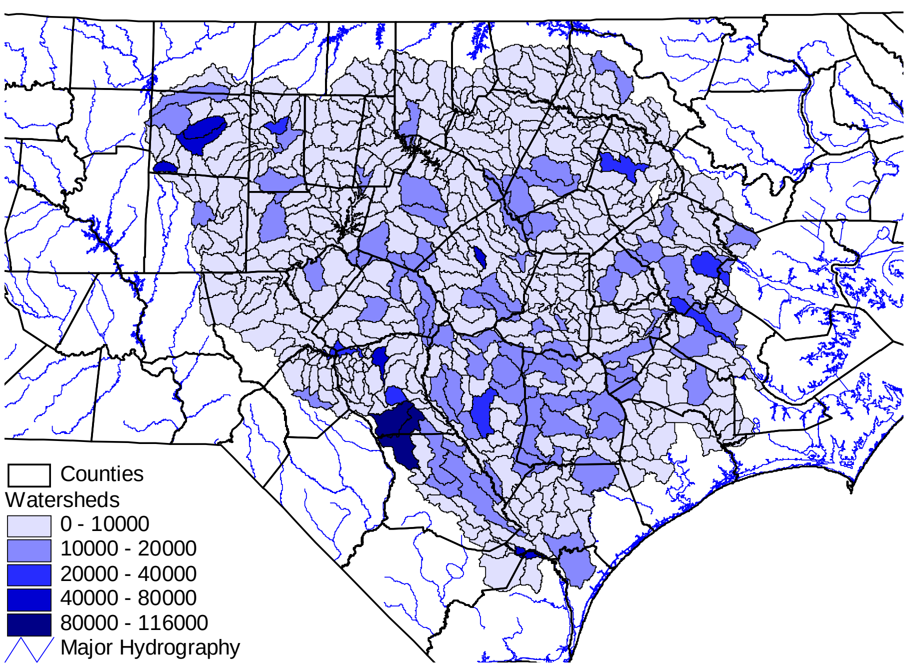


Figure 4-7. Modeled Swine Total Nitrogen Loading by 14-Digit Watershed (kg/yr)

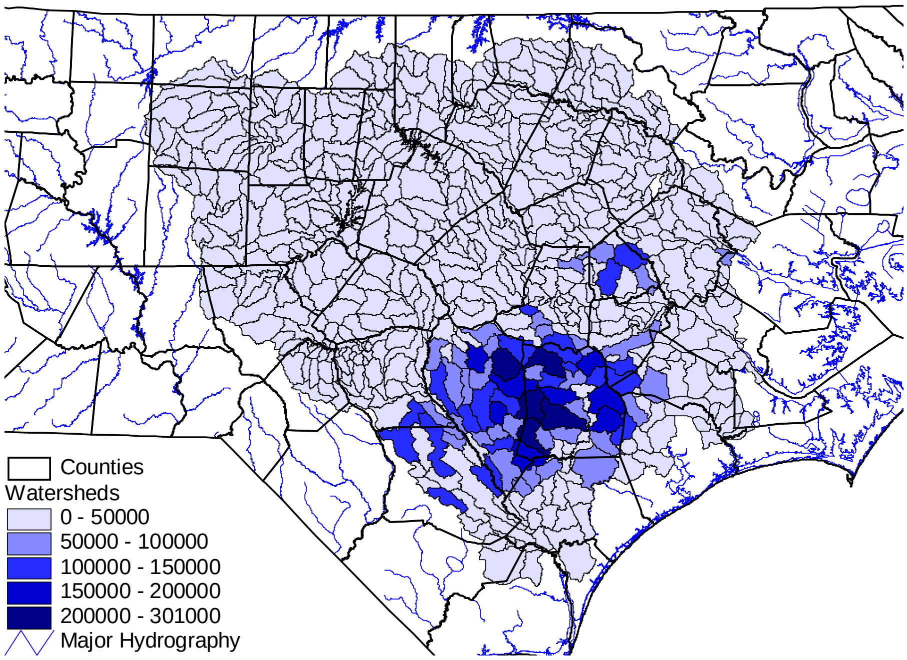


Figure 4-8. Modeled Swine Total Phosphorus Loading by 14-Digit Watershed (kg/yr)

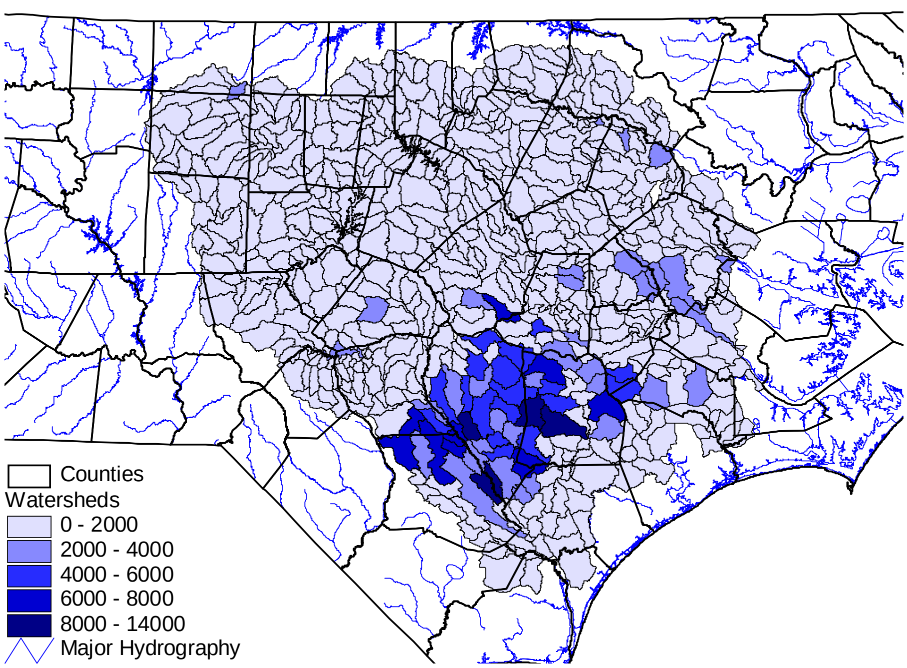


Figure 4-9. Total Swine Deposition Loading by 14-Digit Watershed (kg/yr)

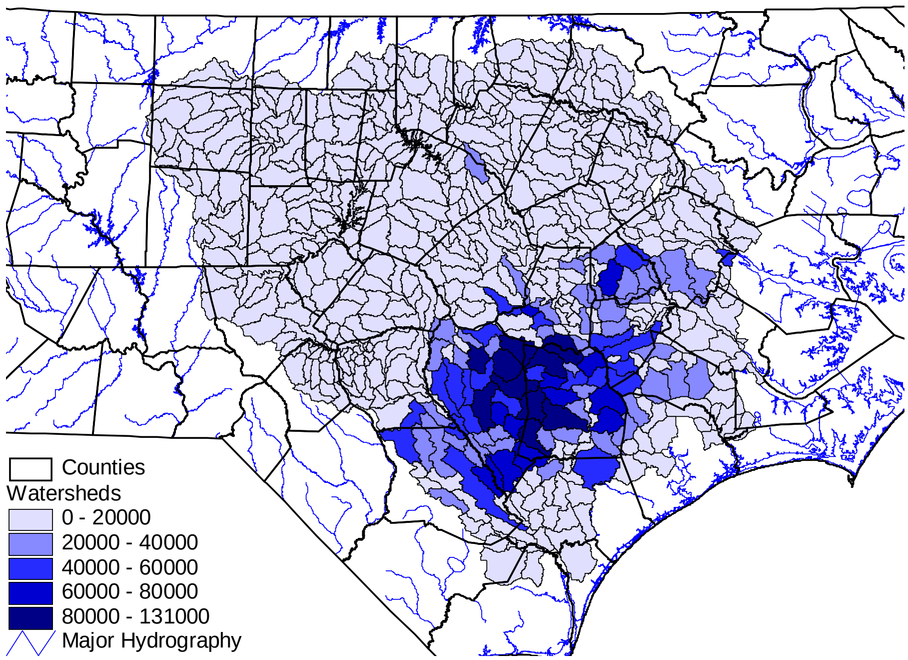


Figure 4-10a. Nitrogen Loading by Source Category

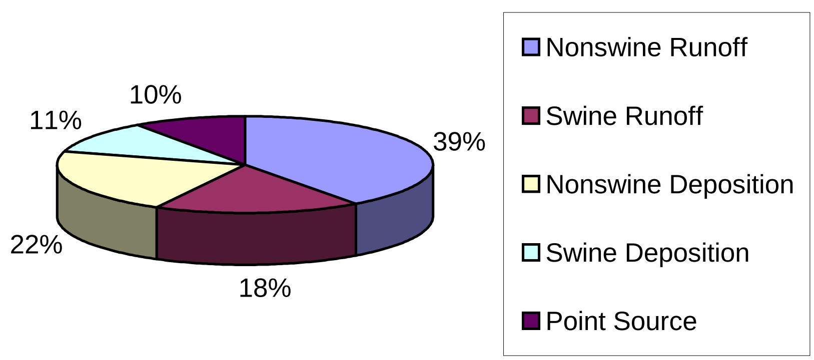


Figure 4-10b. Breakdown of Nitrogen Deposition by Source Category

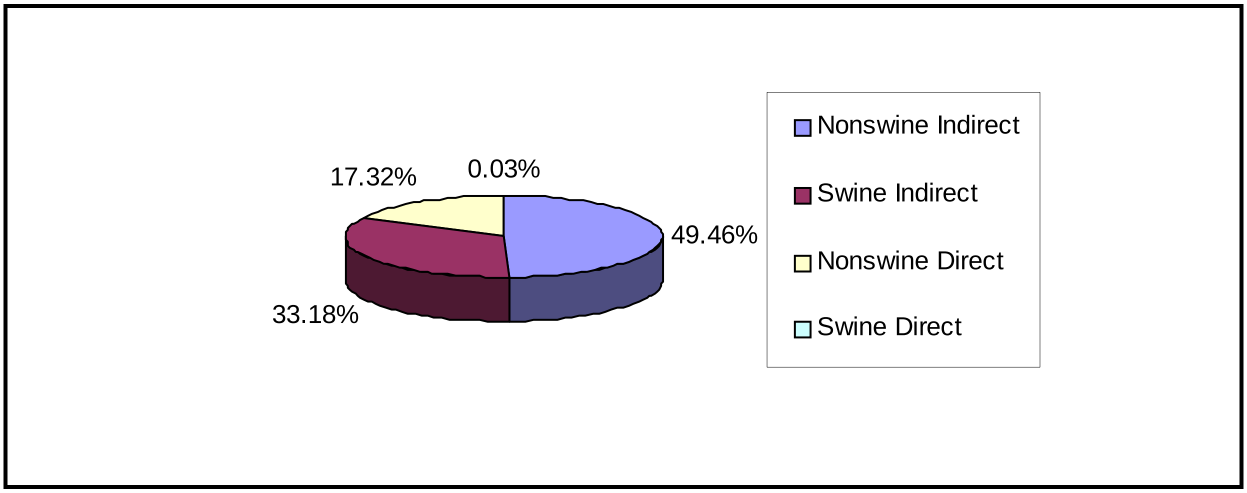
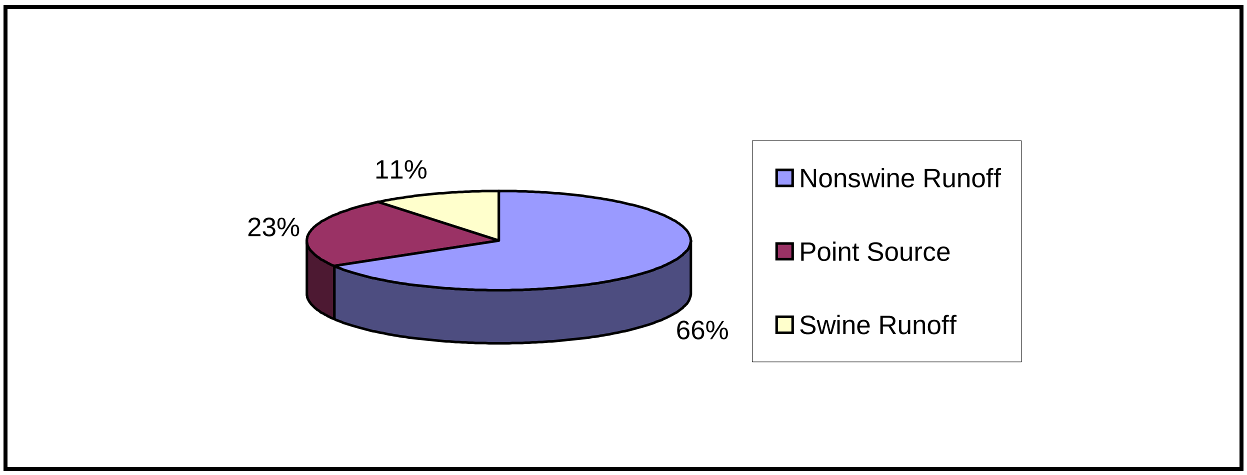


Figure 4-10c. Phosphorus Loading by Source Category



For the entire study area, swine facilities are predicted to contribute 28 percent of the atmospheric nitrogen inputs. Ammonia transported from swine facilities that deposits directly onto estuarine waters is estimated to deposit at rates of 0.01 to 0.04 kg/ha/yr for the different estuaries considered (Pamlico, Neuse, White Oak, and New), accounting for between 0.01 to 0.1 percent of the total estuarine loading. The rate of ammonia direct deposition to estuaries is estimated to be less than 1 percent of the estimated total nitrogen deposition rate accounting for nonswine sources, suggesting that local (indirect) ammonia gas transport and deposition is a more serious concern than ammonia transport directly to estuary waters. However, we cannot draw inferences about ammonium transport from swine facilities to estuaries because we did not attempt to model transport and deposition into the water system of swine waste as ammonium particles.

A sensitivity analysis increasing phosphorus application rates for swine waste based on a "worst case" scenario (see Table 4-3) resulted in an increase in phosphorus delivery to estuaries of almost 10 percent. We also completed a model run in which all swine input source terms for both runoff and deposition were set to zero, providing a hypothetical "zero swine discharge" scenario. This scenario reduced the areawide total nitrogen and phosphorus delivery to estuaries by 125 million kg/yr (N) and 2.4 million kg/yr (P) respectively, with variable reductions in more local stream concentrations and loadings depending on the relative influence from swine facilities. The largest change at a 14-digit watershed outlet level was an instream improvement of 6.5 mg/l of total nitrogen, with the median improvement across all watershed outlet reaches being 0.14 mg/l.

4.3 DISCUSSION

The analysis provides support for the foundation and analytical infrastructure that has been established for basin-scale surface water modeling of nitrogen and phosphorus from swine facilities in eastern North Carolina. Major advances of this study include the establishment of a detailed hydrologic and water quality modeling system linked to watershed definitions, swine facilities, land cover, point source, deposition, and stream monitoring data, and subroutines for estimating source inputs and delivery to surface waters that specifically differentiate between swine and nonswine sources.

The analytical framework created allows for technology assessment via adjustment of nitrogen and phosphorus sprayfield application rates, ammonia deposition rates, crop removal assumptions, and edge-of-field delivery coefficient adjustment. Other technology-related issues to test with the model would require either model respecification or a more involved refinement of some aspect of the model database and infrastructure.

Managing nitrogen and phosphorus in tandem is recognized as being important although management strategies differ between the two nutrients because of their different chemical properties and hydrologic controls. For phosphorus, surface runoff, with its associated water-borne sediment, is the primary mechanism for transport (Gburek, 2000). For nitrogen, the concern is often with subsurface flow of soluble nitrate (NO_3). Nitrogen-rich subsurface drainage waters and shallow aquifers discharge nitrogen into down gradient streams, although there can be a substantial lag from the time of environmental release to surface water contamination based on subsurface flow rates (e.g., Gburek, 2000). An element of coastal eutrophication that has been receiving broad and expanding attention in recent years is the atmospheric deposition, via precipitation and dryfall, and gaseous exchange of nitrogen directly onto water surfaces and indirectly onto land surfaces and then subsequently transported into water bodies. Research suggests that deposition may be responsible for 20 to 40 percent of the "new" nitrogen inputs to estuary waters on the eastern U.S. coast (see, for example, Chesapeake Bay Program et al. [2000]; Valigura et al. [1996]). The Ecological Society of America (2000) provides further assessment of nutrient and animal waste management issues.

The amount of agricultural nitrogen exported to surface waters in coastal North Carolina depends primarily on agricultural practices, the timing of rainfall in relation to fertilizer and manure applications, drainage, spatial relationships between agricultural lands and groundwater recharge and discharge areas, and riparian buffer characteristics (see, for example, Gilliam [2002]; Showers, Usry, and Gannon [2002]). Because of generally well- drained soils and high nitrogen input from fertilizer, manure, and atmospheric deposition, much of the area has potential for contamination of shallow groundwater by nitrate and subsequent movement to surface waters (Kellogg, 2000; Gilliam, 2002; USGS, 1996; Mew and Hofmockel, 2002, see Chapter 5). One recent observation in the Mississippi River basin is that the soil/ground water system can provide a long-term reservoir for nitrogen-rich waters, thereby complicating the ability to predict the rate of response of surface waters to changes in nitrogen inputs (Goolsby and Battaglin, 2000).

Our results demonstrate that ammonia deposition is a potentially significant component of surface water contributions. While the movement of ammonia via emissions, deposition, and watershed and stream delivery is a very new area of study, the potential for this pathway has been documented by previous research. For example, a doubling of ammonium deposition rates and elevated ambient concentrations of ammonia have been reported by Robarge, Cure, and Bode (1999) in Sampson County between 1980 and 1998. Measurements in forests adjacent (<3 km) to swine facilities showed enhanced ammonium deposition relative to sites located farther away (>5 km).

Because of the nature of nitrogen and phosphorus movement in the landscape, atmosphere, and waters, it is important as part of the technology assessment to recognize that mitigating nutrient impacts from swine facilities requires a sensitivity to the mobility of nutrients, perhaps especially with respect to nitrogen. Important factors affecting nitrogen delivery go beyond technological considerations alone, and include landscape position, watershed and riparian management, relationships between surface and groundwater hydrology, the relationship between new waste management systems and cropping and agronomic systems, and, ultimately, accounting for the fate of the large quantities of nutrients imported in animal feed. One potentially significant source of uncertainty and spatial heterogeneity is the degree to which the soil–groundwater system is providing an intermediate reservoir between swine sources and surface waters. How rapidly terrestrial and aquatic systems may react to systemic changes in inputs or new waste management practices is subject to considerable uncertainty. The study of airborne transport of nitrogen from animal operations is very young, and new insights will likely emerge in the coming years.

The existing surface water predictive model is operational, and at the same time, improvements are envisioned and could be pursued to enhance the utility and performance of the tool, as needs and resources warrant. An area for potential improvement is briefly mentioned below.

Based on limited stream monitoring data we have been able to compile and a relevant study by USGS (McMahon, Qian, and Roessler, 2002; McMahon, Alexander, and Qian, In press), our model may somewhat overpredict instream nitrogen concentrations with the current input data, algorithms, and parameters. In the near term, adjustments via either further qualitative assessment and/or more rigorous calibration could improve model performance. In the long term, collection of additional stream monitoring data, especially in the upper Northeast Cape Fear and/or Black River watersheds could be pursued to provide valuable information for model validation.[4] Two specific alternatives for immediate consideration are to test the model by adjusting (increase) small stream and land application (sprayfield) to stream decay rates to account for higher removal expected in these headwater systems than may be reflected in the decay rates currently being employed. A second possibility is to incorporate a riparian buffer model that more specifically accounts for watershed delivery based on land cover (and conceivably other riparian features) in riparian areas. RTI (1997) has already developed a riparian buffer model and processed land cover data, thereby facilitating testing and implementation.

4.4 SUMMARY

The purpose of the surface water assessment is to estimate the contribution of swine facilities to loading and instream concentrations of nitrogen and phosphorus in eastern North Carolina. To pursue this goal, a predictive model of nitrogen and phosphorus inputs and transport through streams to the estuarine boundary was developed. This model was based on an extensive database used to quantify model inputs and parameters and to perform a qualitative assessment of model performance. The results of the evaluation were made available for the economic benefits analysis as discussed in Chapter 6.

Watershed inputs were estimated for swine facilities for both runoff from land application and atmospherically derived ammonia deposition. Additionally, National Pollution Discharge Eliminations System (NPDES) point sources and runoff and deposition from other nonpoint sources were estimated so that all sources to surface waters were accounted for. Instream processing and delivery was also estimated with a stream delivery model. Data obtained generally represent conditions in the mid to late 1990s.

[4]Because of the current relative absence of necessary stream monitoring in the upper Northeast Cape Fear and Black River basins, a high degree of subjectivity in this process is inevitable.

With current input and delivery assumptions, swine waste is estimated to account for almost one-third of the nitrogen and 10 percent of the phosphorus inputs to free-flowing streams and rivers in the study area.

With current input and delivery assumptions, swine waste is estimated to account for almost one-third of the nitrogen and 10 percent of the phosphorus inputs to free-flowing streams and rivers in the study area. Most (62 percent) of the swine delivery of nitrogen to free flowing surface waters is estimated to occur via runoff, with the remainder through atmospheric deposition of ammonia upstream from estuarine waters. Ammonia transported from swine facilities that deposits directly onto estuarine waters is estimated to deposit at rates of 0.01 to 0.04 kg/ha/yr, accounting for about 0.01 to 0.1 percent of the total estuarine loading. The predicted rate of ammonia direct deposition to estuaries is estimated to be less than 1 percent of the estimated total nitrogen deposition rate when considering rates based on ambient data, suggesting that local (indirect) ammonia gas transport and deposition is a more serious concern than ammonia transport directly to estuary waters, and that other sources of atmospheric nitrogen provide a majority of the direct atmospheric loading. However, no inferences can be drawn about ammonium transport from swine facilities to estuaries we did not model transport and deposition of swine waste as ammonium particles, which are likely transported greater distances.

In addition to modeling work, a general conceptual framework is provided to help place the modeling work in context of the evolving empirical basis and scientific understanding of coastal eutrophication and nutrient management. Environmental models are abstractions of nature, and sound judgment in using modeling requires contextual consideration as well as understanding of model specification, assumptions, data sources, and important uncertainties. The goal in developing the framework has been to provide a flexible and adaptable tool that can address technology assessment needs.

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5 Groundwater Quality Assessment

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5.1 INTRODUCTION

The groundwater assessment described herein evaluates nitrate contamination of wells supplying drinking water in eastern North Carolina and the potential association with swine farm waste management. A statistical regression model is developed to evaluate the contributions of nitrogen sources, vulnerability factors, and swine farm data to measured groundwater-nitrate concentrations in eastern North Carolina. This model was used to evaluate impacts associated with swine farm sources in that region.

This chapter provides general background information about nitrate in groundwater in the North Carolina Coastal Plain and about documented impacts to groundwater from swine farm sources. We present the data supporting RTI's regression model and the modeling methodology. We then summarize the data and the results of the regression analysis and discuss potentially important factors that remain outside the scope of the analysis.

5.2 BACKGROUND

5.2.1 Groundwater Nitrate Migration in the North Carolina Coastal Plain

Nitrogen can exist in groundwater in multiple forms, including ammonium, nitrite, dissolved nitrogen gas, organic nitrogen, and the anion nitrate. The presence of nitrate in groundwater used as a drinking water source causes particular concern because of the potential health risks. Because of these health effects, the U.S. Environmental Protection Agency (EPA) and North Carolina have adopted 10 mg/L as the maximum allowable nitrate concentration in drinking water. In addition to acute health effects, some studies have identified links between nitrate in drinking water below 10 mg/L and certain forms of cancer (Weyer et al., 2001).

Numerous potential sources of nitrate in groundwater include decaying plant matter, soil organic matter, septic systems, lawn and garden fertilizer, cemeteries, agricultural fertilizer, land application of animal wastes, leaking sewer lines, and sanitary landfills. Heaton (1986) identifies three primary causes of nitrate contamination in groundwater: the release of nitrogen from soils during the conversion of uncultivated land to cultivated land, nitrate fertilizer application, and land disposal of concentrated animal or human waste. In regions where nitrate contamination is extensive, fertilizer application is typically identified as the primary cause (Freeze and Cherry, 1979).

Nitrogen applied to soils as fertilizer is typically in the form of nitrate. Organic wastes such as swine manure contain nitrogen in large organic molecules (amine groups) that can convert to ammonia and then nitrate through microbial activity (mineralization and nitrification). As rainwater percolates through the soil, plants absorb nitrogen. However, nitrogen not taken up by plants may migrate to groundwater. Nitrogen migration to groundwater often occurs during winter months with relatively little plant growth or during wet periods. Agronomic rates of fertilizer and organic waste application are based on estimated crop nutrient requirements. Exceeding these agronomic rates will often lead to groundwater contamination because the crops do not require and do not take up the excess nutrient.

Once in the groundwater system, nitrate is typically mobile. For example, it does not readily sorb to sediments; thus, its migration is usually at the same rate as the groundwater flow. Nitrate generally flows with groundwater from upland areas recharged by rainfall to lower lying discharge areas (e.g., streams, swamps, ditches). Through these pathways, groundwater nitrate impacts can lead to surface water contamination (see Chapter 4).

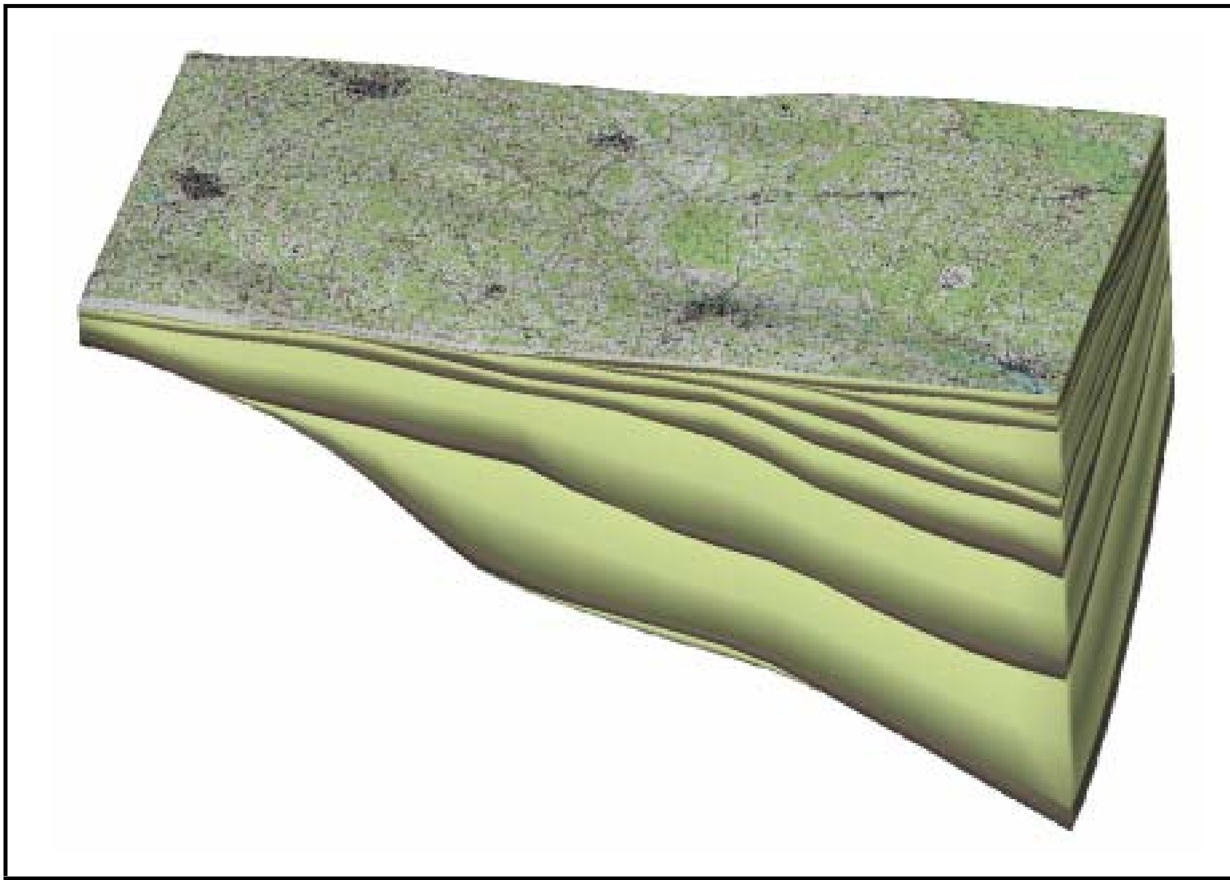
The North Carolina Coastal Plain is underlain by a layered series of aquifers and confining units that slope downward toward the coast, as illustrated in Figure 5-1. The aquifers are the relatively more permeable materials (e.g., limestone, sandstone) that produce the majority of groundwater used for drinking water in the region. The confining units are relatively lower permeability materials (e.g., clay, silt) that limit vertical flow between aquifers. Although regionally significant flow does occur through confining units, the rates of flow are much lower than the rates possible within the aquifers. The uppermost groundwater unit is the surficial aquifer. In the coastal plain, the surficial aquifer is typically underlain by low permeability confining units, which prevent significant downward migration into deeper, confined aquifer systems. To the extent that nitrate contamination is caused by surface releases such as from swine operations, it occurs predominantly within the shallow, surficial aquifer. Nitrate does not typically occur in the deeper, confined aquifer systems (Osmond, Gilliam, and Evans, 2002). However, exceptions can occur in aquifer recharge areas and/or where a confining unit is discontinuous.

5.2.2 Nitrate Contamination of Drinking Water Wells

Nitrate contamination of drinking water wells has been documented in the North Carolina Coastal Plain. A study of 1,719 drinking water wells in North Carolina showed that 1.8 percent of the wells had nitrate-nitrogen levels at or above the 10 mg/L drinking water standard (Miner, Jennings, and Wiggins, 1996). Approximately 20 percent of the wells contained between 4 and 9 mg/L nitrate. Coastal plain wells contained greater levels of nitrate, with 4.9 percent of the tested wells having 10 mg/L nitrate or higher.

A study conducted by the North Carolina Cooperative Extension Service (NCCES) in 1989 to 1990 showed 5 percent of 214 private wells with nitrate contamination above 10 mg/L (Gilliam et al.,

Figure 5-1. Aquifers (Yellow) and Confining Units (Brown) Underlying an Area of the North Carolina Coastal Plain



Notes: Geologic data are from the North Carolina Department of Environment and Natural Resources (DENR) geologic framework database. The topographic map is the USGS Rocky Mount quadrangle (1: 100,000 scale). The vertical exaggeration = 40x. Three-dimensional visualization by RTI.

1996). Follow-up testing of these wells in 1996 indicated relatively consistent results with 6 percent of the wells above 10 mg/L (NCCES, 1996).

The North Carolina Governor's Office initiated a water well testing program in 1995, whereby residents located near intensive livestock operations (including swine, turkeys, chicken, and cattle) could have their wells tested for nitrate (Warrick, 1996). This project included 948 wells in 50 counties. The results showed 34.8 percent of the wells with nitrate concentrations above 2 mg/L and 9.4 percent of the wells with nitrate at or above the 10 mg/L drinking water standard (Rudo, 1996).

In summary, nitrate contamination of drinking water wells in the North Carolina Coastal Plain has been documented in numerous studies. The percentage of wells with concentrations above 10 mg/L (the EPA drinking water standard) ranged from 4.9 percent (Miner, Jennings, and Wiggins, 1996) to 9.4 percent (Rudo, 1996) in the studies reviewed for this report.

5.2.3 Nitrate Impacts to Groundwater from Swine Farms

Potential and measured impacts to groundwater associated with swine farms have been well documented in the scientific literature. A regional-scale vulnerability analysis used animal populations and other indices to rank the Cape Fear River basin the most vulnerable in the country to contamination from manure nutrients, while the Neuse/Pamlico River basin ranked fifteenth (Kellogg, 2000).

Numerous studies have shown elevated nitrate concentrations in shallow groundwater below fields where swine waste was applied (Huffman et al., 1994; Gilliam et al., 1996; Israel and Showers, 2001). The study by Gilliam et al. (1996) investigated two sites in the North Carolina Coastal Plain. One site in Duplin County had been farmed with commercial fertilizer. The other site in Sampson County was a swine-waste sprayfield. High nitrate concentrations were observed in groundwater below the fields at both sites; however, no nitrate was found in the deeper aquifers below the confining unit at either site. Israel and Showers (2001) studied 90 test wells installed within sprayfields. Nitrate levels were as high as 50 mg/L, and the average concentration in the shallow groundwater was between 12 and 14 mg/L.

The U.S. Geological Survey (USGS), EPA, and the North Carolina Department of Environment and Natural Resources (NCDENR) have monitored nitrate at the Lizzie Research Site in Greene County (Spruill, Tesoriero, and Showers, 2002). In 1995 a field where fertilizer had been applied was converted to a sprayfield for swine waste. Before conversion to a sprayfield, nitrate concentrations in groundwater below the field were around 10 mg/L, and the isotopic signature of the nitrate indicated primarily an inorganic fertilizer and soil organic nitrogen source. After conversion to a sprayfield, nitrate concentrations increased to nearly 50 mg/L, and isotopic analysis indicated primarily a swine waste source. Before and after conversion to a sprayfield, the nitrate concentration in an upgradient well was approximately 5 mg/L. This analysis provided a unique opportunity to compare nitrate impacts associated with crop fertilizer application and swine-waste sprayfield operations at the same location.

Although nitrate contamination has been well documented in groundwater underlying sprayfields, direct causative links have been established between nitrate contamination in drinking water wells and swine farms in certain cases. The DENR Groundwater Section is responsible for regulatory oversight of permitted animal waste land application systems. Through this regulatory program, DENR has identified over 17 animal operations in eastern North Carolina in the vicinity of local private water supply wells with nitrate concentrations exceeding 10 mg/L (Mouberry, 2003).

One study performed isotopic analysis of nitrate-contaminated drinking water from wells in the vicinity of swine farms in Sampson County (Law Engineering, 1997). The isotopic analysis of 29 wells showed that two-thirds of the wells were contaminated primarily from a synthetic fertilizer source. The remaining wells indicated multiple contamination sources, including septic systems and organic nitrogen. Animal waste appeared to be a relatively minor influence in two wells.

5.3 METHODOLOGY

RTI developed an empirical statistical regression model to estimate potential groundwater impacts associated with swine waste management in eastern North Carolina. This model follows an approach used by EPA to evaluate its recently promulgated regulations on concentrated animal feeding operations (CAFOs) (EPA, 2002). The purpose of the model is to statistically estimate the relationship between measured nitrate concentrations in drinking water wells and various factors expected to influence these levels.

EPA's analysis used data from the USGS Retrospective Database, which provided a national sample of 2,928 wells tested for nitrate concentrations. The EPA model regressed measured nitrate levels on county-level characteristics (land use, soil type, septic system density, and nitrogen loadings) and well depth. EPA's analysis found that county-level nitrogen loadings had a positive and statistically significant effect on nitrate levels. The Agency used the results of this model to predict how reductions in loadings would reduce groundwater nitrate levels across the country. We developed a similar model at a more spatially refined level for the eastern North Carolina Coastal Plain to explain the observed variation in measured nitrate levels and to isolate the effect of nitrogen loadings related to swine farms in eastern North Carolina.

5.3.1 Dependent Variable: Groundwater Nitrate Concentrations

Measured groundwater nitrate concentrations in production wells constituted the key data component for the analysis. Unfortunately, no single, comprehensive database containing such information is available for eastern North Carolina. The USGS Retrospective Database used in the EPA study (EPA, 2002) only contained a few data points in the Albemarle drainage basin, which is not sufficient for a study focusing on eastern North Carolina. Although various sources of groundwater nitrate data were considered (including data from DENR regional offices) only one of these sources (the Groundwater Quality Database) contained sufficient information on well depths. Because of the probable importance of this variable, only these data were included in the regression analysis. The Groundwater Quality Database provides groundwater quality data for over 2,000 private wells in eastern North Carolina and was developed for a study sponsored by North Carolina’s Water Resources Research Institute (WRRI) (Devine, Baran, and Sewall, 2002). The original database contained data collected between March and July of 2000 and was obtained from the North Carolina Division of Environmental Health’s (DEH) State Public Health Laboratory. RTI obtained the database from the study researchers after they had processed the data and merged the data with a database of available well construction data.

As discussed in Section 5.2.1, nitrate contamination in the North Carolina Coastal Plain does not typically migrate through low-permeability confining layers underlying surficial aquifers. So it is unlikely that surface loadings of nitrogen in this region such as swine operations would lead to systematic impacts in deeper production wells. Therefore, the analysis focused on two subsamples of shallower wells:

- 516 wells with depths of less than or equal to 100 ft (referred to as the “100-ft sample”) and

- 281 wells with depths of less than or equal to 50 ft (referred to as the "50-ft sample").

5.3.2 Explanatory Variables

RTI compiled data from several sources and constructed a number of explanatory variables potentially related to groundwater nitrate concentrations, including

- swine farm inventory data and associated nitrogen loads;
- distance between groundwater-concentration measurements and swine farms within 3 miles;
- modeled atmospheric nitrogen deposition associated with swine farms;
- nonswine nitrogen source data, including fertilizer, septic systems, and nonswine atmospheric deposition; and
- related vulnerability factors, including a soil drainage characteristic and agricultural versus nonagricultural land use.

Table 5-1 describes the explanatory variables used in the regression analysis. Each of these variables is included based on scientifically grounded expectations about their potential influence on groundwater nitrate levels. In particular, nitrate levels should be positively influenced by whether local land is predominantly used for agriculture (AG), the level of fertilizer use (FERT), the density of septic systems in the area (SEPT), and the level of manure used from livestock other than swine. In addition, deeper wells (WELLDEPTH) and less permeable soil (such as clay) (DRAIN_C and DRAIN_D) are expected to have a negative effect on nitrate levels, all else equal. Controlling for these factors, the model can estimate the incremental impact of swine farm-related variables, including atmospheric nitrogen deposition (ATM_DEP), the distance between swine farms and nitrate measurement locations (within 3 miles), and distance-weighted sprayfield nitrogen loadings from swine farms within 3 miles (SPRAY_DIST) on groundwater nitrate levels. Note that the ATM_DEP term includes a component related to swine farms as well as a nonswine atmospheric deposition component. Each of the explanatory variables is described below along with the data sources.

Table 5-1. Descriptions of Regression Variables

Variable Name	Units	Description
ATM	$mg/m^2/yr$	Atmospheric nitrogen deposition, combining both swine and nonswine sources
SPRAY	mg/yr	Swine farm sprayfield nitrogen loadings
DIST	meters	Distance between the calibration wells and swine farms
AG	—	Agricultural vs. nonagricultural land use (dichotomous variable)
FERT	$mg/m^2/yr$	Fertilizer use in the region
SEPT	$mg/m^2/yr$	Nitrogen loadings from septic systems
LIVESTOCK	$mg/m^2/yr$	Nitrogen associated with waste from livestock other than swine (e.g., cows, chickens, turkeys)
DRAIN_B	—	Soil drainage—Type B (dichotomous variable)
DRAIN_C	—	Soil drainage—Type C (dichotomous variable)
DRAIN_D	—	Soil drainage—Type D (dichotomous variable)
WELLDEPTH	feet	Well depth

Atmospheric Nitrogen Deposition from Swine Farm Sources

The loadings from swine farm atmospheric nitrogen deposition correspond to the air modeling results documented in Chapter 2.

Nonswine Atmospheric Nitrogen Deposition Loading

Nonswine atmospheric nitrogen deposition was estimated from available wet and dry deposition data from ambient monitoring sites from 1996 through 2000 (see Chapter 4, Table 4-4). Monitoring stations that exhibited apparent influence from swine operations were excluded. Influence from swine operations was determined by proximity to intensive swine operations and comparison with atmospheric modeling results (see Chapter 2). Available data included 15 wet-deposition monitoring stations and three dry-deposition monitoring stations. Because of the relative consistency of dry deposition data and the sparsity of monitoring locations, we averaged dry deposition data and used a single-value estimate throughout the study area. Wet deposition data from 1996 through 2000 were averaged at each monitoring station and then interpolated through the study area using a simple inverse distance weighted method. The interpolated values were distributed on a 1 km grid throughout the study area.

Swine Farm Sprayfield Nitrogen Loading

The loadings to swine farm sprayfields used for the groundwater analysis followed the approach described in Chapter 2 for the air emissions modeling. This approach involved developing volatilization emission factors for animal housing, lagoon, and sprayfield operations associated with different farm types (e.g., wean to feed, feed to finish). The nitrogen remaining following these volatilization emissions comprised the sprayfield loadings for the groundwater modeling, which are listed in Table 5-2 for each farm type. We calculated the per-farm sprayfield loadings by multiplying the appropriate emission factor by the steady state live weight (SSLW) for each farm as provided in DENR's swine farm inventory.

Table 5-2. Calculated Swine Farm Sprayfield Loading Rates By Farm Type

Farm Type	Sprayfield Loading Rate (kg N/yr/lb)
Farrow to wean	0.0119
Wean to feed	0.0441
Farrow to feed	0.0172
Farrow to finish	0.0198
Feed to finish	0.0233

Distance to Swine Farms

We determined swine farm locations using the DENR swine farm inventory data. Distances to these farm locations were determined through GIS proximity analysis. Only farms within 3 miles of wells were included, because impacts to groundwater outside of this distance are considered highly unlikely.

Agricultural vs. Nonagricultural Land Use

The agricultural versus nonagricultural land use data are from the USGS National Land Cover Data Set (NLCD) (USGS, 1992). The NLCD was compiled from satellite imagery (circa 1992) with a spatial resolution of 30 meters. For our analysis only agricultural versus nonagricultural land use classes were distinguished. The

NLCD categories Herbaceous Planted and Cultivated were assigned as agricultural, while the rest of the land use categories were nonagricultural.

Fertilizer Nitrogen Loading

The fertilizer nitrogen loading data are from a USGS spatial dataset containing estimates of nitrogen fertilizer sales in 1991 as reported by EPA (Battaglin and Goolsby, 1994). Nitrogen fertilizer sales estimates are reported for each county in tons of nutrient sold. The rate of nitrogen fertilizer use in tons per square mile per year is also included and was used for the regression analysis.

Septic System Nitrogen Loading

We estimated septic system nitrogen loadings using 1990 Census data. A density of septic systems was derived by dividing the number of septic systems per Census block group by the area of each block group. We used an estimate of the nitrogen load per septic system of 22.5 lb/year from Horsely, Santos, and Busby (1996) to calculate the nitrogen loading from the number of septic systems per block group area.

Nonswine Livestock Nitrogen Loading

Nitrogen loadings from nonswine animal manure were based on county-level estimates of the nitrogen content of animal waste produced in 1992. These estimates are based on animal populations for those years from the 1992 Census of Agriculture and methods for estimating the nutrient content of manure from the Soil Conservation Service. Estimates at the county level are available for the nitrogen content in manure associated with various livestock. The specific data source was a geospatial dataset provided by USGS (Puckett, Hitt, and Alexander, 1998).

Soil Drainage Characteristic

The soil drainage characteristic is the soil hydrologic group in the State Soil Geographic Data Base (STATSGO) (Wolock, 1997). The soil hydrologic group can assume the following values:

- Type A—Low runoff potential; high infiltration rates even if saturated; sands and gravels (0.3 to 0.45 in/hr);
- Type B—Moderate infiltration rates if thoroughly wetted (0.15 to 0.30 in/hr); course to moderately fine textures;

- Type C—Slow infiltration rates if thoroughly wetted; moderately fine to fine (0.05 to 0.15 in/hr);
- Type D—Very slow rates; clays with lots of swelling; high water tables (<0.05 in/hr);

The North Carolina Coastal Plain contains no soils with a type A soil hydrologic group.

5.3.3 Model Formulation

A logarithmic form for the dependent variable, NITRATE, is selected because the distribution of this variable is highly skewed, with a long "right side tail." The model thus is of the following semi-log form:

$$\begin{aligned}\ln(NITRATE) = {} & \alpha_1 ATM + \alpha_2\left(\sum_{i=1}^{\#\,farms}(SPRAY_i / DIST_i)\right) + \\ & \alpha_3 AG + \alpha_4 FERT + \alpha_5 SEPT \\ & + \alpha_6 LIVESTOCK + \alpha_7 DRAINC + \alpha_8 DRAIND \\ & + \alpha_9 WELLDEPTH + CONSTANT\end{aligned} \tag{5.1}$$

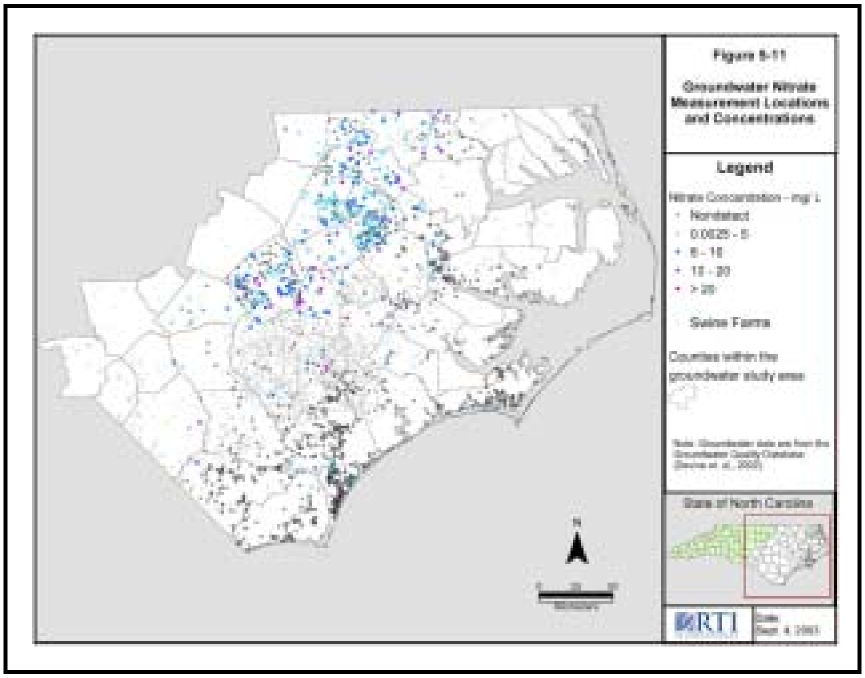
where the α terms are the regression coefficients and the explanatory variables are as described in Section 5.3.1. The second explanatory variable in Eq. (5.1), which we also refer to as SPRAY_DIST, is the sum of "inverse-distance-weighted" sprayfield loadings from swine farms within 3 miles of the sampled well. Loadings from more distant farms are expected to contribute less to groundwater nitrate levels; therefore, in this term, the sprayfield loadings estimates from each farm are divided by their distance from the well. Note that the variables AG, DRAINC, and DRAIND are dichotomous or "dummy" variables, which are equal to one or zero. For example, AG equals one when the land use is agricultural. This specification implies that the potential effects of nonagricultural land and DRAINB soils are captured within the CONSTANT term. In addition, interval regression was used rather than ordinary least squares (OLS) because of the large number of nitrate samples below detection limits. Interval regression is a maximum likelihood estimation approach that allows the dependent variable to be specified as either a point estimate or as an interval (defined by both a lower bound and an upper bound). For observations below the detection limit, the lower bound for NITRATE is set at zero and the upper bound is the corresponding detection limit.

5.4 RESULTS

5.4.1 Groundwater Nitrate Data Summaries

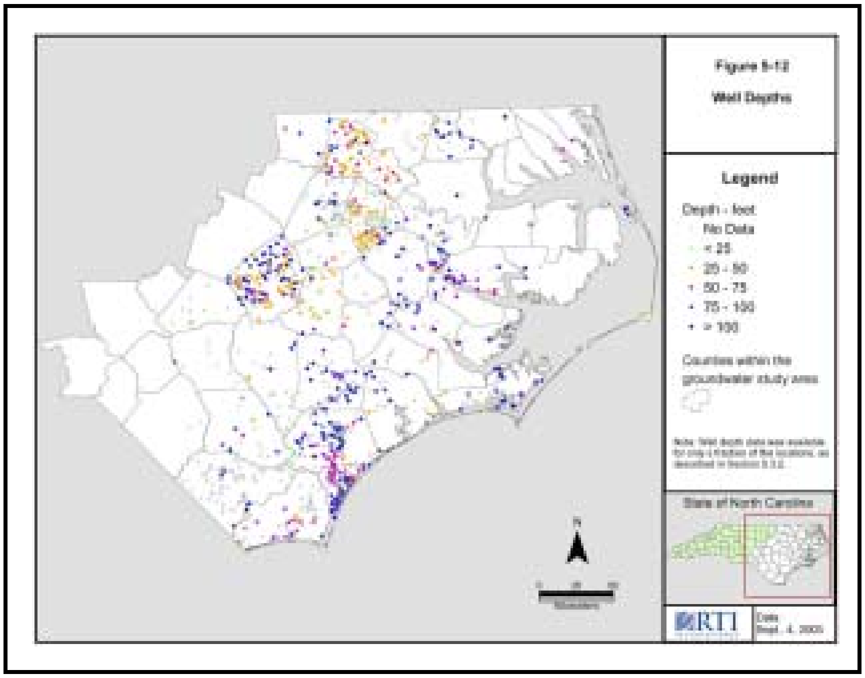
Figure 5-2 presents the nitrate concentrations at each of the groundwater measurement locations. Figure 5-3 shows the well depths for the groundwater measurement locations where well depth information was available.

Figure 5-2. Groundwater Nitrate Measurement Locations and Concentrations



The Groundwater Quality Database contains nitrate concentration data for 2,453 wells located in the groundwater study area. A large number of the sampled concentrations from these wells (78 percent) are below detection limits. As shown in Table 5-3, the average nitrate level for these wells is 1.3 mg/l (following standard practice, below-detection observations were set to one-half the detection limit for this calculation). The average number of swine farms within 3 miles of a well is 1.8 farms. Only 990 of the 2,453 wells have data on well depth. This subset of 990 wells has roughly the same distribution of nitrate levels and number of local swine farms as the full sample. The median well depth for this subsample is 100 ft and the 25th percentile is 50 ft.

Figure 5-3. Well Depths



Not surprisingly, as shown in Table 5-3, the wells with a depth of 50 ft or less (the 50-ft sample) have higher average nitrate levels than the remainder of the wells. The average nitrate concentrations for the 100- and 50-ft samples were 1.8 mg/L and 2.6 mg/L, respectively, compared to 1.3 mg/L for the entire sample. On average, these two samples of wells are also located near somewhat fewer swine farms (1.3 and 1.2 farms, respectively, versus 1.8 for the entire sample).

Table 5-4 compares groundwater nitrate levels with swine farm parameters for both the 50- and 100-ft samples. The following three cases were analyzed: distance to the nearest swine farm, number of swine farms within 3 miles, and the cumulative sprayfield loading for all farms within 3 miles. In each case and for both the 50- and 100-ft samples, there is a moderately increasing trend in concentration as the influence from swine farms increases. Using the 50-ft sample as an example, the average nitrate concentration for wells with no swine farms within 3 miles is 1.6 mg/L, while the average nitrate concentration is 4.3 mg/L for wells with at least one swine farm within 1 mile.

Table 5-3. Summary Statistics: Depth and Nitrate Levels for Sampled Private Wells in the Study Area

Variable	Number of Observations	Mean	Std. Dev.	Min.	Median	Max.
All Sampled Wells						
Nitrate[a] (mg/L)	2,453	1.32	2.18	0.05	0.5	21.73
Number of hog farms within 3 mi.	2,453	1.8	3.9	0	0	36
All Sampled Wells With Depth Data						
Nitrate[a] (mg/L)	990	1.28	2.08	0.05	0.5	20.59
Well depth (ft)	990	162.7	1198.5	7	100	37,017
Number of hog farms within 3 mi.	990	1.6	3.5	0	0	35
100-ft Sample (All Sampled Wells With Depth ≤100 ft)						
Nitrate[a] (mg/L)	516	1.84	2.69	0.05	0.5	20.59
Well depth (ft)	516	53.9	24.4	7	50	100
Number of hog farms within 3 mi.	516	1.3	2.8	0	0	35
50-ft Sample (All Sampled Wells With Depth ≤50 ft)						
Nitrate (mg/L)[a]	281	2.60	3.05	0.05	1.13	20.59
Well depth (ft)	281	34.8	9.7	7	35	50
Number of hog farms within 3 mi.	281	1.2	2.0	0	1	15

[a]Values below detection limit are set equal to one-half of detection limit.

Source: Devine, H.A., P.K. Baran, and L.C. Sewall. 2002. An Environmental Water Quality Data Visualization System for Private Ground Water Supplies, Draft Completion Report, submitted for review. North Carolina Water Resources Research Institute (WRRI) Project No. 50301.

5.4.2 Explanatory Variable Data Summaries

Table 5-5 provides summary statistics for the explanatory variables both for the 50-ft and 100-ft samples. Figures 5-4 through 5-10 show the distribution of nitrogen loadings throughout the eastern North Carolina groundwater study area. The units (mg N/m^2/yr) and color scales for each of these figures are the same to facilitate comparison of the loadings. Table 5-6 provides a quantitative comparison of the loadings per county throughout the region.

Table 5-4. Descriptive Statistics Comparing Nitrate Levels with Swine Farm Parameters

Category	Count	Percent	Nitrate Concentrations (mg/L)			
			Average	Standard Deviation	Min	Max
Statistics for the 50-ft Sample						
Distance to Nearest Swine Farm (miles)						
>3	133	47.3	1.6	1.9	0.5	9.3
2 – 3	78	21.0	2.7	2.7	0.5	12.0
1 – 2	59	19.9	3.8	4.1	0.05	20.6
<1 mile	33	11.7	4.3	3.8	0.5	15.1
Number of Farms within 3 Miles						
No farms	133	47.3	1.6	1.9	0.5	9.3
1 farm	70	24.9	3.4	3.3	0.05	13.2
2 farms	38	13.5	3.0	3.0	0.5	12.0
3 – 4 farms	25	8.9	3.4	3.3	0.5	10.3
5 or more farms	15	5.3	4.9	5.0	0.5	20.6
Cumulative Sprayfield Loading (mg $N/m^2/yr$)						
0	133	47.3	1.6	1.9	0.5	9.3
0.77 – 6,798	42	15.0	3.7	3.6	0.5	13.2
6,798 – 13,318	43	15.3	3.2	2.6	0.5	12.0
13,318 – 27,862	37	13.2	2.3	2.8	0.5	10.3
>27,862	26	9.3	5.2	5.1	0.1	20.6
Statistics for the 100-ft Sample						
Distance to Nearest Swine Farm (miles)						
>3	292	56.6	1.1	1.5	0.5	9.3
2 – 3	78	15.1	2.6	3.4	0.5	19.9
1 – 2	91	17.6	2.7	3.6	0.05	20.6
<1 mile	55	10.7	3.1	3.5	0.5	15.1
Number of Farms within 3 Miles						
No farms	292	56.6	1.1	1.5	0.5	9.3
1 farm	97	18.8	3.0	3.6	0.05	19.9
2 farms	50	9.7	2.6	2.9	0.5	12.0
3 – 4 farms	39	7.6	2.7	3.0	0.5	10.3
5 or more farms	38	7.4	2.5	4.3	0.5	20.6
Cumulative Sprayfield Loading (mg $N/m^2/yr$)						
0	292	56.6	1.1	1.5	0.5	9.3
0.77 – 6,798	60	11.6	3.1	4.0	0.5	19.9
6,798 – 13,318	54	10.5	3.0	2.8	0.5	12.0
13,318 – 27,862	54	10.5	2.0	2.6	0.5	10.3
>27,862	56	10.8	3.0	4.1	0.05	20.6

Table 5-5. Summary Statistics for Explanatory Variables

Variable	Unit	Number of Observations	Mean	Std Dev	Min	Median	Max.
All Sampled Wells With Depth ≤100 ft							
ATM	10^3 mg/m^2/yr	516	1.487	0.783	1.012	1.273	9.676
SPRAY	10^9 mg/yr	516	14.134	35.275	0	0	435.889
SPRAY_DIST[a]	10^6 mg/yr/m	516	5.426	14.026	0	0	176.951
AG		516	0.395	0.489	0	0	1
FERT	10^3 mg/m^2/yr	516	1.807	1.293	0	2.150	4.616
SEPT	10^3 mg/m^2/yr	516	0.211	0.335	0.008	0.103	2.877
LIVESTOCK	10^3 mg/m^2/yr	516	0.380	0.539	0	0.201	2.331
DRAIN_C		516	0.343	0.475	0	0	1
DRAIN_D		516	0.316	0.465	0	0	1
WELLDEPTH	feet	516	53.872	24.405	7	50	100
All Sampled Wells With Depth ≤50 ft							
ATM	10^3 mg/m^2/yr	281	1.499	0.827	1.022	1.279	9.676
SPRAY	10^9 mg/yr	281	12.732	23.785	0	1.923	181.970
SPRAY_DIST[a]	10^6 mg/yr/m	281	5.205	11.408	0	0.610	85.206
AG		281	0.470	0.500	0	0	1
FERT	10^3 mg/m^2/yr	281	2.152	1.264	0	2.247	4.616
SEPT	10^3 mg/m^2/yr	281	0.156	0.205	0.008	0.086	1.806
LIVESTOCK	10^3 mg/m^2/yr	281	0.468	0.510	0	0.464	2.046
DRAIN_C		281	0.374	0.485	0	0	1
DRAIN_D		281	0.228	0.420	0	0	1
WELLDEPTH	feet	281	34.829	9.698	7	35	50

[a]"Inverse-distance-weighted" nitrogen loadings from hog farms within a 3-mile radius of the well.

Figure 5-4. Estimated Fertilizer Nitrogen Loading

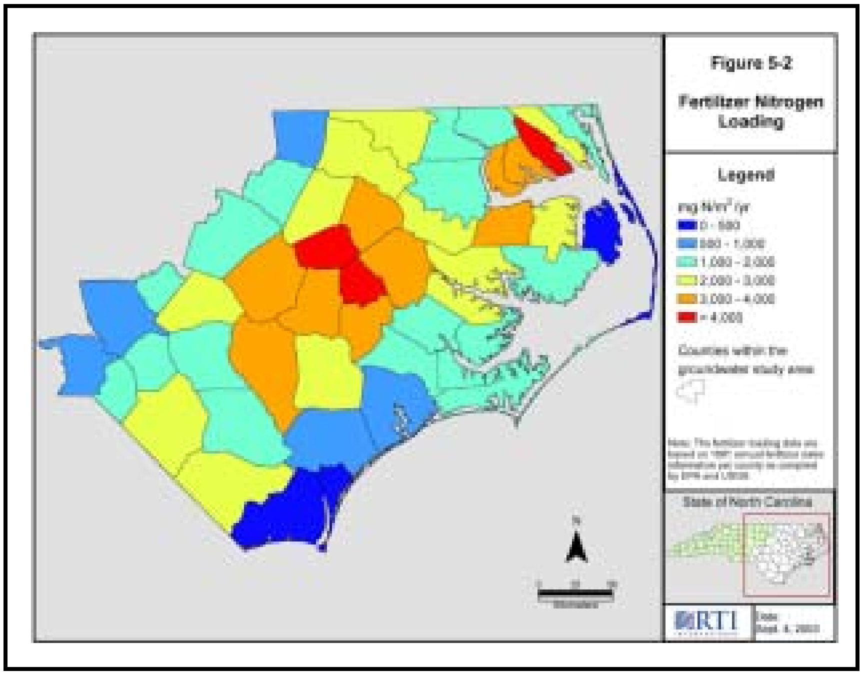


Figure 5-5. Estimated Nonswine Atmospheric Nitrogen Deposition Loading

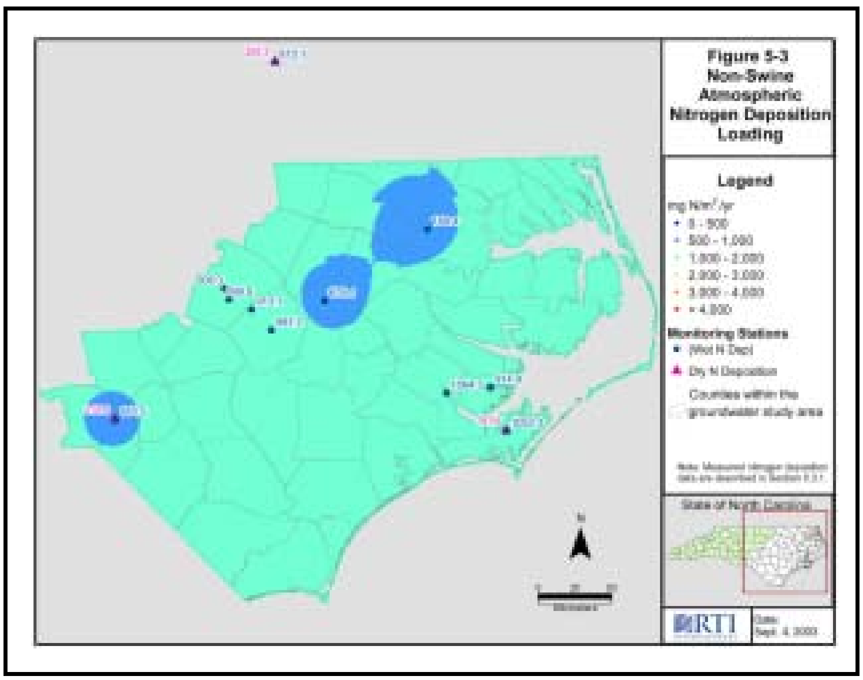


Figure 5-6. Estimated Nonswine Livestock Nitrogen Loading

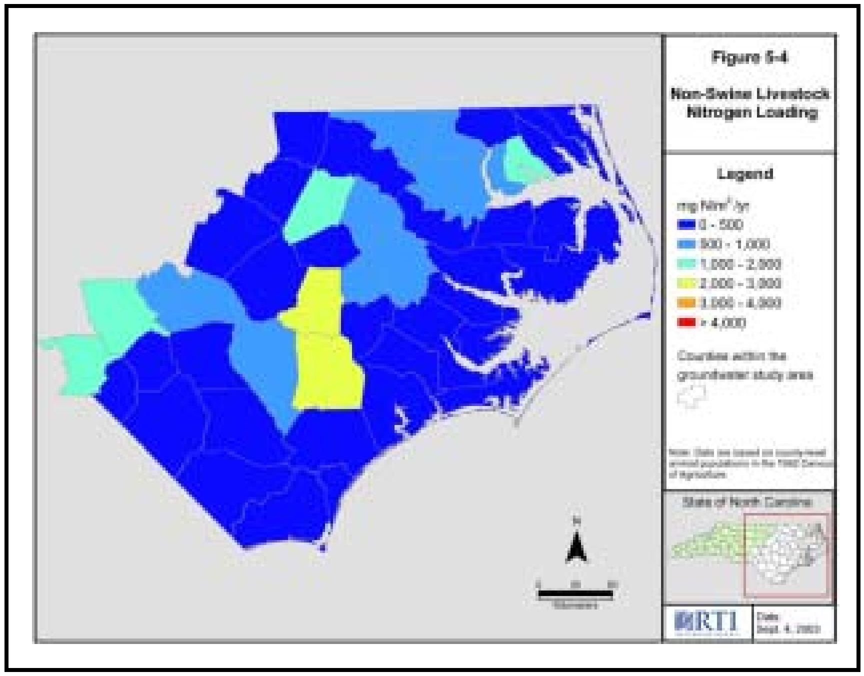


Figure 5-7. Estimated Septic System Nitrogen Loading

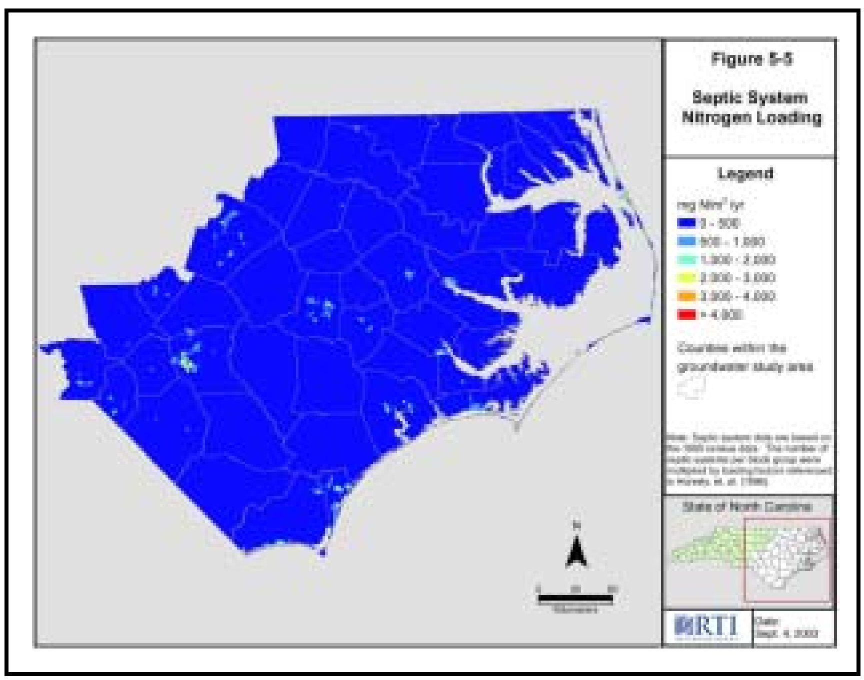


Figure 5-8. Estimated Swine Farm Sprayfield Nitrogen Loading (per county)

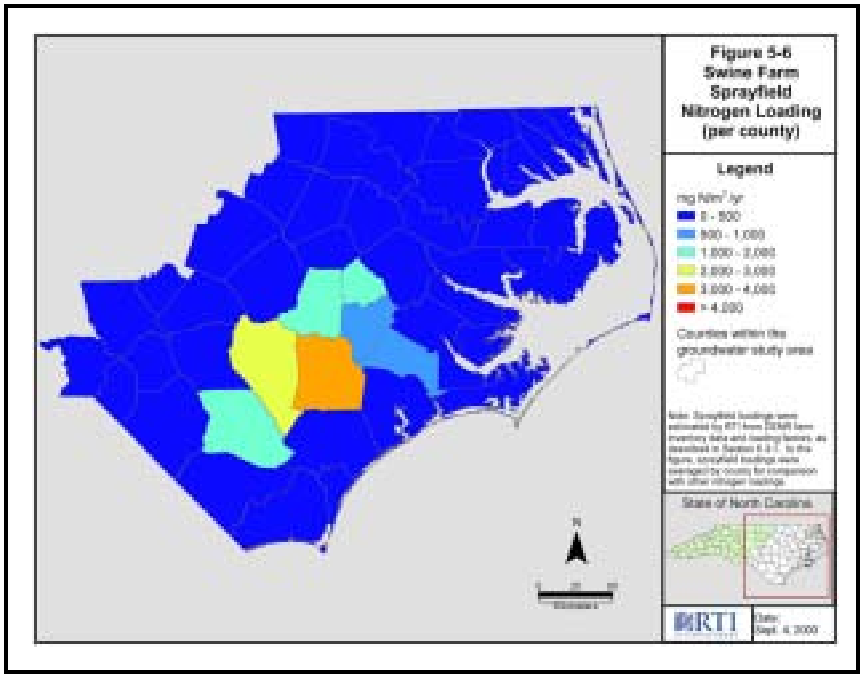


Figure 5-9. Estimated Swine Farm Sprayfield Nitrogen Loading (per farm)

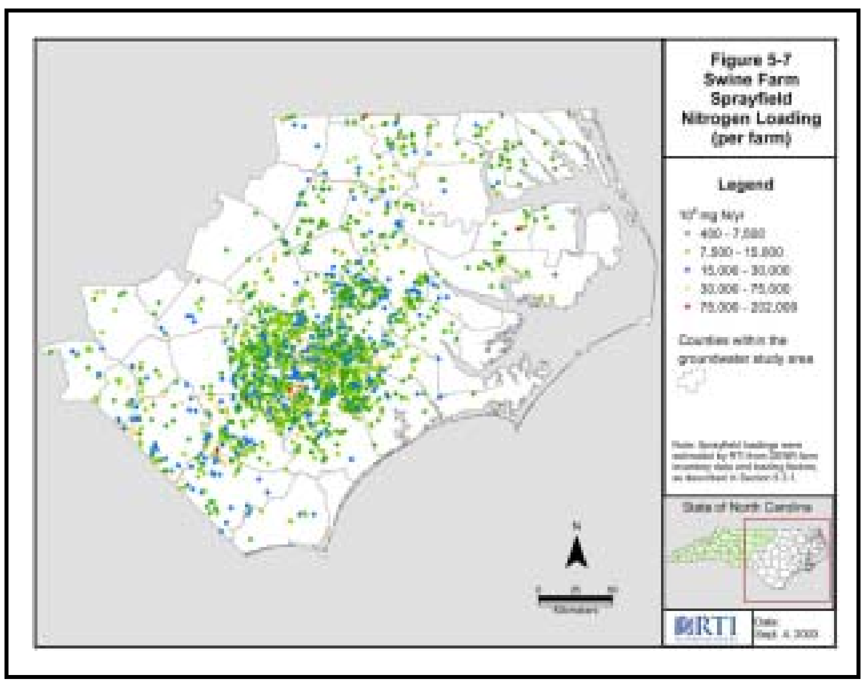
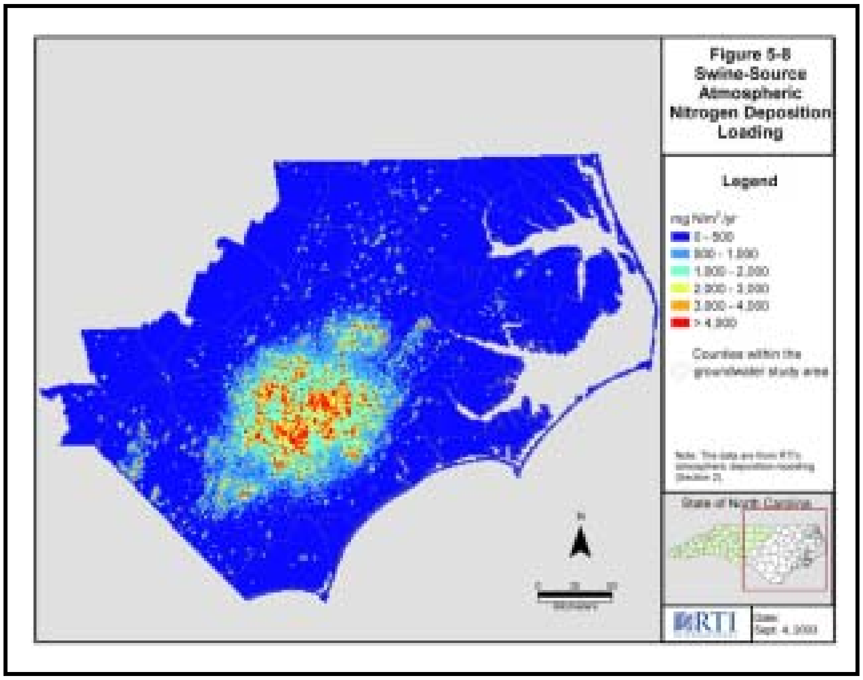


Figure 5-10. Estimated Swine-Source Atmospheric Nitrogen Deposition Loading



Comparison of the loadings indicates that fertilizer application accounts for 43 percent, which is the greatest percentage of the regional nitrogen loading. Nonswine atmospheric deposition accounts for 24 percent, and the total swine farm loadings account for 20 percent regionally. Exceptions to these regional average patterns include Bladen, Duplin, and Sampson counties, where nearly 50 percent of the nitrogen loading comes from swine farm sources.

Figure 5-11 shows the distribution of the agricultural versus nonagricultural land use. Figure 5-12 shows the distribution of the soil drainage characteristic.

5.4.3 Regression Analysis Results

The description statistics give a rough indication of groundwater differences across the landscape, but more rigorous procedures are needed to test whether different influences are statistically significant. This is accomplished by the regression analysis.

Regression results for the 100-ft and 50-ft samples are shown in Tables 5-7 and 5-8, respectively. In both cases, results for three model specifications are shown. The default specification includes

Table 5-6. Comparison of Estimated County-Level Nitrogen Loadings

	Loading Per County (1,000 kg N/m^2/year)								
	Nonswine Loadings					Swine Loadings			
County	Fertilizer	Atmospheric Deposition	Nonswine Livestock	Septic	Total	Sprayfield	Atmospheric Deposition	Total	Swine and Nonswine Total
Beaufort	4,396	2,543	18	136	7,093	325	406	730	7,823
Bertie	3,281	1,837	1,635	60	6,814	132	171	303	7,117
Bladen	2,583	2,544	271	82	5,480	2,331	2,640	4,971	10,451
Brunswick	908	2,513	36	321	3,778	210	262	472	4,250
Camden	1,592	664	9	28	2,294	10	3	13	2,306
Carteret	1,595	1,641	3	325	3,564	8	18	26	3,590
Chowan	1,588	474	239	35	2,336	58	40	97	2,434
Columbus	5,157	2,748	253	146	8,305	772	961	1,734	10,038
Craven	2,322	2,280	33	142	4,776	245	417	662	5,438
Cumberland	2,248	1,862	181	381	4,672	297	705	1,002	5,674
Currituck	1,184	730	9	66	1,989	2	1	3	1,992
Dare	0	1,096	0	246	1,342	0	3	3	1,345
Duplin	5,946	2,411	4,945	111	13,413	6,534	6,320	12,855	26,268
Edgecombe	4,029	1,306	744	75	6,155	442	545	987	7,142
Franklin	1,821	1,398	367	117	3,702	64	92	156	3,858
Gates	1,638	917	388	33	2,977	69	14	83	3,060
Greene	3,290	714	650	44	4,697	1,239	1,179	2,418	7,115
Halifax	4,256	1,914	879	98	7,147	256	390	646	7,792
Harnett	3,339	1,736	1,190	170	6,436	191	326	517	6,953

(continued)

Table 5-6. Comparison of Estimated County-Level Nitrogen Loadings (continued)

	Loading Per County (1,000 kg N/m²/year)								
	Nonswine Loadings					Swine Loadings			
County	Fertilizer	Atmospheric Deposition	Nonswine Livestock	Septic	Total	Sprayfield	Atmospheric Deposition	Total	Swine and Nonswine Total
Hertford	1,865	925	709	49	3,548	132	133	265	3,813
Hoke	1,300	1,041	35	64	2,440	211	302	513	2,952
Hyde	2,452	2,059	9	33	4,553	35	71	107	4,660
Johnston	6,589	2,349	881	213	10,031	588	1,044	1,632	11,663
Jones	1,642	1,475	75	31	3,222	831	977	1,808	5,030
Lee	661	737	559	90	2,047	6	19	25	2,072
Lenoir	4,071	1,169	512	108	5,859	875	1,360	2,235	8,094
Martin	3,550	1,247	340	57	5,193	25	145	170	5,364
Moore	1,044	1,905	3,597	168	6,715	105	177	283	6,997
Nash	3,701	1,439	2,476	123	7,739	254	325	579	8,318
New Hanover	107	609	4	170	890	0	34	34	924
Northampton	3,169	1,422	831	55	5,476	418	390	807	6,284
Onslow	1,724	2,349	352	229	4,654	597	871	1,468	6,122
Pamlico	1,470	1,072	8	53	2,603	10	30	40	2,643
Pasquotank	3,174	638	16	57	3,885	11	7	19	3,904
Pender	1,393	2,591	251	142	4,377	874	1,415	2,289	6,667
Perquimans	1,995	689	695	38	3,416	37	18	55	3,472
Pitt	5,603	1,825	1,220	157	8,805	728	950	1,678	10,483
Richmond	677	1,251	1,717	97	3,742	156	193	349	4,091

(continued)

Table 5-6. Comparison of Estimated County-Level Nitrogen Loadings (continued)

	Loading Per County (1,000 kg N/m^2/year)								
	Nonswine Loadings					Swine Loadings			
County	**Fertilizer**	**Atmospheric Deposition**	**Nonswine Livestock**	**Septic**	**Total**	**Sprayfield**	**Atmospheric Deposition**	**Total**	**Swine and Nonswine Total**
Robeson	6,939	2,596	707	233	10,474	1,105	1,273	2,378	12,853
Sampson	7,459	2,775	1,966	145	12,345	5,662	6,325	11,987	24,332
Scotland	1,044	819	347	73	2,283	330	378	708	2,992
Tyrrell	2,035	1,161	3	13	3,212	83	88	171	3,384
Wake	2,821	2,478	442	462	6,203	12	82	94	6,298
Warren	997	1,226	532	67	2,822	113	120	233	3,055
Washington	3,280	1,066	472	30	4,849	249	204	453	5,302
Wayne	5,010	1,570	2,950	220	9,751	1,696	2,168	3,864	13,615
Wilson	4,412	910	216	83	5,621	112	306	418	6,039
Total	**131,359**	**72,721**	**33,774**	**5,872**	**243,725**	**28,440**	**33,900**	**62,340**	**306,066**
Percentage	**43%**	**24%**	**11%**	**2%**	**80%**	**9%**	**11%**	**20%**	**100%**

Figure 5-11. Agricultural and Nonagricultural Land Use

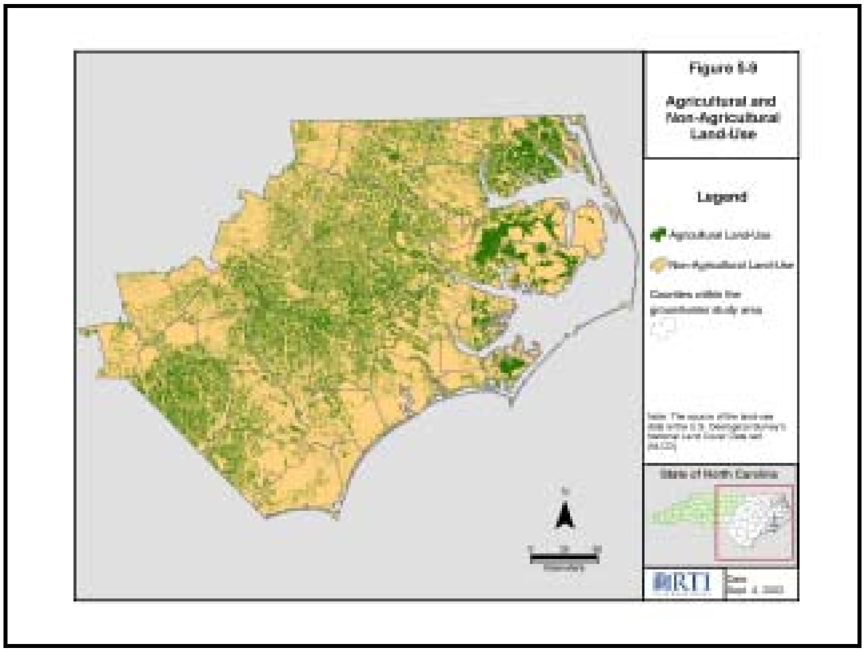


Figure 5-12. Soil Drainage Characteristic

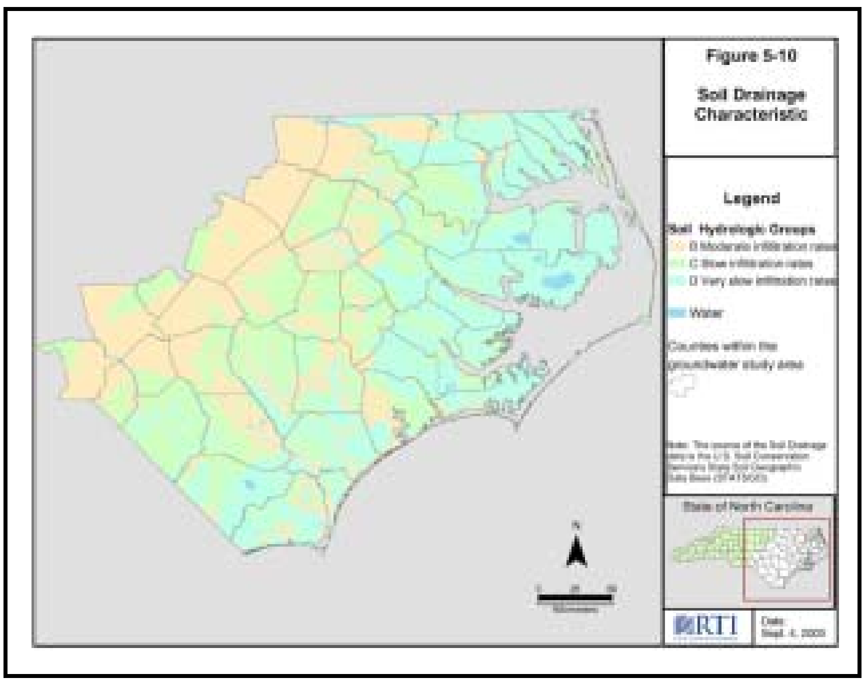


Table 5-7. Regression Results for Wells with Depth ≤100 ft (N = 516)

Explanatory Variable	Default Specification		ATM Specification		SPRAY_DIST Specification	
	Coefficient	z-stat	Coefficient	z-stat	Coefficient	z-stat
ATM	**0.274**	1.97	0.126	1.20	—	—
SPRAY_DIST	–0.013	–1.24	—	—	0.00043	0.06
AG	**0.819**	3.87	**0.801**	3.79	**0.791**	3.72
FERT	**0.474**	3.98	**0.484**	4.10	**0.492**	4.15
SEPT	–0.761	–1.34	–0.694	–1.26	–0.727	–1.29
LIVESTOCK	0.142	0.56	0.085	0.34	0.127	0.49
DRAIN_C	–0.168	–0.72	–0.155	–0.66	–0.146	–0.62
DRAIN_D	**–0.919**	–3.30	**–0.903**	–3.26	**–0.923**	–3.30
WELLDEPTH	**–0.036**	–7.44	**–0.037**	–7.58	**–0.036**	–7.47
CONSTANT	–0.680	–1.54	–0.519	–1.20	–0.366	–0.88
Sigma	1.815	0.09	1.818	0.09	1.825	0.09
LR χ^2	227.87	—	221.58	—	221.20	—
Prob > χ^2	0.0000	—	0.0000	—	0.0000	—

Notes: Interval regressions with Ln(NITRATE) bounds as dependent variables. Uncensored observations = 177; Left-censored observations = 339. Z-stat values are based on robust standard error estimates using a Huber-White correction. Values for sigma are standard error estimates. Coefficients in bold font are significant at a p = 0.1 level.

Table 5-8. Regression Results for Wells with Depth ≤50 ft (N = 218)

Explanatory Variable	Default Specification		ATM Specification		SPRAY_DIST Specification	
	Coefficient	z-stat	Coefficient	z-stat	Coefficient	z-stat
ATM	**0.261**	1.87	**0.275**	3.35	—	—
SPRAY_DIST	0.001	0.10	—	—	**0.01674**	2.03
AG	**0.947**	4.13	**0.949**	4.13	**0.920**	4.01
FERT	**0.415**	2.96	**0.414**	2.97	**0.441**	3.21
SEPT	–0.516	–0.73	–0.524	–0.74	–0.467	–0.66
LIVESTOCK	0.037	0.12	0.044	0.14	–0.023	–0.07
DRAIN_C	0.030	0.12	0.029	0.12	0.068	0.28
DRAIN_D	–0.385	–1.28	–0.385	–1.29	–0.390	–1.28
WELLDEPTH	0.012	1.01	0.012	1.01	0.013	1.04
CONSTANT	**–2.322**	–3.72	**–2.334**	–3.74	**–2.068**	–3.34
Sigma	1.597	0.09	1.597	0.09	1.606	0.09
LR χ^2	87.19	—	86.13	—	79.54	—
Prob > χ^2	0.0000	—	0.0000	—	0.0000	—

Notes: Interval regressions with Ln(NITRATE) bounds as dependent variables. Uncensored observations = 150; Left-censored observations = 131. Z-stat values are based on robust standard error estimates using a Huber-White correction. Values for sigma are standard error estimates. Coefficients in bold font are significant at a p = 0.1 level.

both ATM_DEP and SPRAY_DIST as explanatory variables with estimated coefficients for each variable. These two variables are relatively highly correlated, with correlation coefficients ranging from 0.75 to 0.78. The high degree of multicollinearity prevents strong inferences from being made when both variables are included in the regression. Therefore, two additional regression model specifications were included with either ATM_DEP (ATM_DEP Specification) or SPRAY_DIST (SPRAY_DIST Specification), but not both.

Nonswine Farm-Related Explanatory Variables

As shown in Tables 5-7 and 5-8, the coefficients for some of the explanatory variables are statistically significant and consistent with scientific expectations. For all three model specifications and for each sample, the coefficients for AG and FERT are consistently positive and significant. This result indicates that, controlling for other factors, wells within predominantly agricultural land and in areas with greater fertilizer application have statistically significant larger nitrate concentrations. In all model specifications for the 100-ft sample, the coefficients for WELLDEPTH and DRAIN_D are negative and significant, indicating that nitrate concentrations are generally lower in deeper wells and in wells located below less-permeable shallow soil. These two coefficients are not statistically significant for the 50-ft sample, possibly because of the smaller sample size. In addition, wells with a depth less than 50 ft are more likely screened within the surficial aquifer and thus relatively equally susceptible to contamination (as compared to the situation where wells are screened in deeper aquifers below confining units). The coefficients for LIVESTOCK, DRAIN_C, and SEPTIC do not have statistically significant impacts on nitrate levels for any of the specifications or samples.

Swine Farm-Related Explanatory Variables

Controlling for the factors discussed above, the regression analysis allows evaluation of the impacts of ATM and SPRAY_DIST on nitrate levels. When ATM (and not SPRAY_DIST) is included in the model estimation, its coefficient is positive for both the 50-ft and 100-ft samples. However, the coefficient is statistically significant (at a 0.10 level) only for the 50-ft sample. The results are similar for the SPRAY_DIST variable when it (and not ATM) is included in the regression model. These results indicate that nitrogen loadings from local swine farms have a statistically significant positive impact on nitrate levels in wells less than 50 ft deep.

Predictive Simulations

To evaluate the potential magnitude of impacts associated with nitrogen loadings from swine farms, we developed predictive simulations based on the regression results. The simulations each used the 50-ft sample (218 wells) and the regression results reported in Table 5-8.

The first simulation was based on the ATM model specification (Table 5-8) and evaluated the change in nitrate concentrations from eliminating the estimated swine farm contribution to atmospheric nitrogen deposition. Note that in this simulation, the atmospheric deposition component from nonswine sources was not changed. The average predicted decrease in nitrate concentrations for the 218 wells in the 50-ft sample was 0.34 mg/L when swine farm atmospheric nitrogen deposition was reduced to zero.

The second simulation was based on the SPRAY_DIST specification (Table 5-8) and evaluated the change in nitrate concentrations from eliminating swine farm sprayfield loadings. The average predicted decrease in nitrate concentrations for the 218 wells in the 50-ft sample was 0.26 mg/L when swine farm sprayfield loadings were reduced to zero.

Alternative MINDIST Regression Formulation

There was some concern that the sprayfield loadings may inaccurately weight farms with larger swine populations. Larger operations typically apply waste to larger sprayfields; thus, the loading per area may not be represented accurately by a single per-farm loading. To further evaluate the data without this potential weighting bias, we developed an additional regression formulation that captured the swine farm influence by the distance (in miles) to the nearest swine farm (MINDIST). The results in Table 5-9 show that, for the wells in the 50-ft sample with at least one farm within 3 miles (N = 148), MINDIST has a negative and statistically significant effect on nitrate concentrations. This result indicates that nitrate concentrations generally increase as the distance to swine farms decreases.

Table 5-9. MINDIST Regression Results for Wells with Depth ≤50 ft (N = 148)

	Coefficient	z-stat
MINDIST	**–0.422**	–2.350
AG	**0.994**	3.26
FERT	**0.719**	2.99
SEPT	1.450	0.86
LIVESTOCK	–0.354	–1.00
DRAIN_C	0.286	0.92
DRAIN_D	0.128	0.28
WELLDEPTH	–0.000572	–0.03
CONSTANT	–1.719	–1.70
Sigma	1.591	—
LR χ^2	30.96	—
Prob > χ^2	0.0001	—

Notes: Interval regressions with Ln(NITRATE) bounds as dependent variables. Uncensored observations = 93. Left-censored observations = 55. Z-stat values are based on robust standard error estimates using a Huber-White correction. Values for sigma are standard error estimates. Coefficients in bold font are significant at a p = 0.1 level.

Based on these results, we also conducted a predictive simulation to examine the magnitude of these estimated impacts. If MINDIST was set equal to 3 miles for each well, such that no farms were closer than 3 miles to a well, the results yield an average predicted nitrate reduction of 1.66 mg/L for the 148 wells.

5.5 DISCUSSION

The groundwater regression modeling indicates that the nitrogen loadings from swine farms have a statistically significant effect on groundwater nitrate concentrations in shallower wells (less than 50 feet), but these effects are relatively small in magnitude. In Chapter 6 we further explore these results by approximating the economic benefits to private well users of reducing swine farm contributions to nitrate levels.

The apparent limited system-wide influence of swine farms on nitrate in groundwater production wells in the North Carolina Coastal Plain can be understood based on the regional hydrogeology. In eastern North Carolina, the majority of groundwater production wells recover groundwater from relatively deep, confined aquifers (e.g., Cape Fear Aquifer, Black Creek

Aquifer). A confined aquifer is a relatively permeable material (e.g., sandstone, limestone) below a relatively impermeable confining material (e.g., clay). These aquifers are generally preferred water sources, because the water quality is typically better than the shallow groundwater near the surface. If releases from swine farms reach groundwater, the impacts likely will be to the shallow groundwater as opposed to the deeper, confined aquifers. Because most groundwater production wells are screened within the deeper, confined aquifers, the potential for impacts from swine farms on groundwater production wells is generally limited. The Coastal Plain hydrogeology is significantly different than the situation in many other areas of the country. For example, many Midwestern areas with high concentrations of CAFOs are underlain by aquifers where surface loadings can more readily migrate to groundwater used as a primary drinking water source. This hydrogeologic difference likely explains the increased impacts demonstrated by the national EPA CAFO analysis versus the present analysis of eastern North Carolina.

The results suggest that the overall swine farm contribution to nitrate levels in drinking water is relatively small when considering the entire population of groundwater wells in the region and taking other nutrient sources into consideration. To analyze more local-scale conditions, a site-specific, deterministic modeling approach would likely be more appropriate.

The results of RTI's groundwater analysis should not be interpreted as evidence that swine farms in North Carolina cannot or do not, in certain circumstances, impair groundwater quality. In fact, impacts to surficial groundwater associated swine farms are well documented (see Section 5.2.3). Given enabling conditions (e.g., a shallow well located downgradient and near a sprayfield), nitrate contamination of drinking water wells is possible; however, nitrate contamination has only been attributed conclusively to swine farm sources in isolated cases. Rather than investigating such isolated, local-scale situations, the regression analysis described in this chapter is designed to investigate general and systematic impacts within the region. The results suggest that the overall swine farm contribution to nitrate levels in drinking water is relatively small when considering the entire population of groundwater wells in the region and taking other nutrient sources into consideration. To analyze more local-scale conditions, a site-specific, deterministic modeling approach would likely be more appropriate.

Certain further qualifications on the analysis should be considered. First, the regression analysis is wholly dependent on the available groundwater nitrate data, which as previously described, are quite limited for this region. Second, no attempt was made to estimate the potential impacts to surficial groundwater not associated with drinking water wells. Third, it should be noted that the shallow groundwater contamination associated with swine farms can readily lead to surface water contamination as the groundwater flows into streams, ditches, and other discharge areas. These potential impacts were not investigated in the groundwater analysis; however, surface-water impacts were analyzed separately as discussed in Chapter 4. Finally, groundwater travel time can be relatively slow. Therefore, some impacts associated with swine facilities constructed relatively recently may not yet have reached groundwater production wells. The impact from this groundwater travel time is not believed to be significant for the general results; however, it may be significant in isolated instances.

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6 Monetized Benefits of Changes in Environmental Quality

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This chapter describes the methods, data, and results of the economic benefits assessment stage of this research study. The purpose of this assessment is to estimate, to the extent possible, the dollar value of benefits associated with reducing environmental releases from hog farms. As described in previous chapters, releases from swine operations can negatively affect environmental quality through various media (air, land, groundwater, and surface water). In the process, they can impair the functioning of natural ecosystems and reduce the level of "services" that these systems provide to humans. Therefore, the main objective of this task can be thought of as assessing how reductions in environmental releases (i.e., through improved waste management practices) will enhance the value of these services.

Using results from the OPEN team's analyses of alternative technologies and the previously described environmental models, the benefits assessment can translate changes in environmental releases and impacts into measures of human well-being, expressed in monetary terms. Figure 6-1 displays the four main analytical components of the benefits assessment (in the shaded boxes) and

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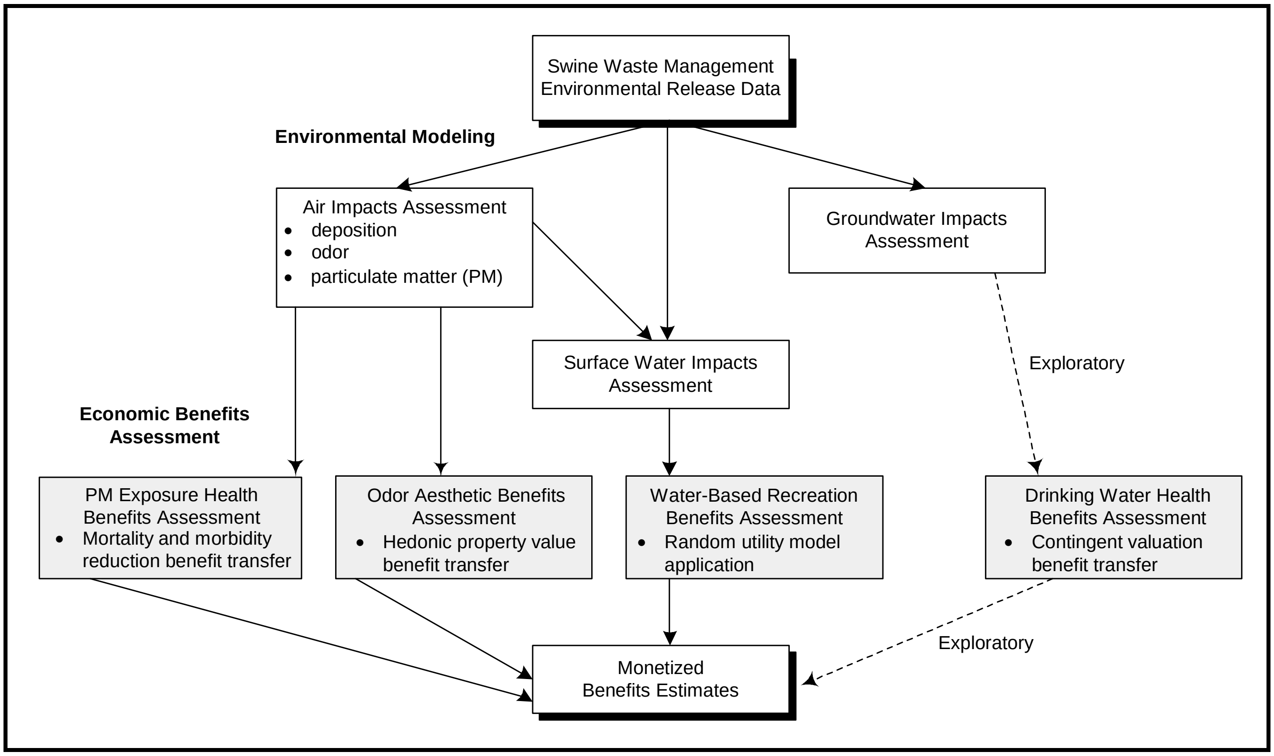
Figure 6-1. Overview of Benefits Assessment Approach and Linkages to Environmental Modeling



how each component relates to the environmental analyses. These four components can be briefly described as assessments of

- ➤ odor reduction benefits,
- ➤ recreation benefits associated with improved surface water quality, and
- ➤ health benefits from reductions in ambient levels of particulate matter,
- ➤ health benefits associated with reductions in groundwater nitrate concentrations.

It must be emphasized that these four components do not address all of the potential environmental benefits associated with alternative waste management technologies. Rather, because of limitations in resource and data availability and the state of the science, they address the benefits that are expected to be the largest and/or can be estimated most reliably within the scope of this study.

Throughout the analysis, we apply the concept of willingness to pay (WTP) to estimate the monetary value of benefits associated with improvements in environmental quality. WTP is generally accepted by economists as the appropriate measure for valuing changes in individuals' well-being.[2] Environmental improvements do not necessarily result in directly observable monetary gains, but this does not mean they have no value. To estimate individuals' WTP for these nonpecuniary improvements, we use "nonmarket" valuation techniques, which have been specifically developed for such purposes.

Given the resource constraints and scope of this project, this assessment has not involved developing an entirely new, primary nonmarket valuation study. Instead, we have selected several existing applications of these valuation methods, and we have adapted them to address the environmental changes expected to result from the alternative waste management technologies. In most cases, this modeling approach can be described as "benefit transfer." Benefit transfer refers to the practice of taking benefit values estimated in one context (i.e., from existing nonmarket valuation studies) and adapting them to value policy-related changes in a separate, but similar, context.

Below we describe in detail the data, methods, results, and uncertainties associated with each of the four components of the benefits assessment. We conclude with a discussion of potential benefits that were not quantified or monetized in this analysis.

6.1 BENEFIT ESTIMATION FOR ODOR-RELATED CHANGES IN AIR QUALITY

Odor emissions from hog farms are a continuing concern in North Carolina, particularly for residents living in close proximity to farms. Corresponding with the rapid increase in the number of hog farms, odor-related complaints increased significantly in the 1990s (Swine Odor Task Force, 1995). However, significant uncertainty exists regarding the characteristics and practices at hog farms that contribute most directly to odors; the chemical constituents that are most directly associated with odors; and the health, psychological, and lifestyle impacts of odors on local residents (Thu, 1997; Shusterman, 1992; Schiffman, 1998; Schiffman, Bennett, and Raymer, 2001).

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[2]As appropriate, we will also use willingness-to-accept (WTA) measures, which are conceptually similar to WTP.

To test for and approximate the size of the disamenity effects (i.e., welfare loss) associated with proximity to hog farms, economists have primarily used hedonic property value models. These empirical models characteristically regress housing price data on various property attributes, including measures of size, distance, and/or direction to hog farms. They are used to estimate the implicit price effects associated with each separate attribute.

The data, methods, and results of existing hedonic studies applied to measuring hog farm disamenities vary widely; however, most find significant negative price effects. One exception is a Minnesota study by Taff, Tiffany, and Weisberg (1996) that found a positive effect of the number of feedlots within 3 miles. Most other studies, including Abeles-Allison and Conner (1990) in Michigan, Mubarak, Johnson, and Miller (1999) in Missouri, Ready and Abdalla (2003) in Pennsylvania, and Herriges, Secchi, and Babcock (2003) in Iowa, as well as four studies conducted in North Carolina—Palmquist, Roka, and Vukina (1997) and three studies summarized in Thomas et al. (2003)—have found evidence of statistically significant negative impacts associated with proximity to hog farms.

Although the collective evidence from these studies suggests that these negative residential property price effects associated with hog farms do exist, each study has limitations for specifically measuring odor-related effects for hog farms in North Carolina. First, and most importantly, only one of these studies—Palmquist, Roka, and Vukina (1997)—has thus far been published in the peer-reviewed literature.[3] Second, many studies have been conducted outside of North Carolina, which makes their results somewhat more questionable for measuring disamenities in this state. Third, although most of these studies find significant negative price effects associated with distance and/or size of local hog operations, none of them can specifically attribute this effect to odor. Herriges et al. (2003) come closest to isolating odor disamenities by including a predominant wind direction in their calculations. They find some evidence that negative effects are largest for properties that are near and downwind of farms.

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[3]Many of the other cited hedonic studies are still in the process of being peer reviewed.

We apply the results of the Palmquist, Roka, and Vukina (1997) study for our analysis. The main advantages of this study is that it has withstood peer review for a major journal in the field of environmental economics, and it was applied in North Carolina.

Given this collective evidence and the comparative advantages and disadvantages of the various studies, we apply the results of the Palmquist, Roka, and Vukina (1997) study for our analysis. The main advantages of this study is that it has withstood peer review for a major journal in the field of environmental economics, and it was applied in North Carolina. We acknowledge that limitations and uncertainties are associated with applying this study to measure odor-related disamenities, which we discuss in more detail in Section 6.1.3.

6.1.1 Methodology and Data

To assess benefits in monetary terms of reductions in odors from hog farms, we use a benefit transfer approach. That is, we use the results from the study conducted by researchers at NCSU (Palmquist, Roka, and Vukina, 1997) and adapt them to assess odor-related benefits for our study area. Palmquist, Roka, and Vukina applied a hedonic property value method, using data on housing prices in nine counties in southeastern North Carolina. Controlling for other housing attributes, they found that proximity to hog farms had a significantly negative impact on housing values and that these effects varied by the size of the operation. These price differentials are assumed to reflect individuals' WTP to avoid hog farms and, in particular, their odors.

We use the results from Palmquist, Roka, and Vukina to develop a benefit transfer (or "valuation") function that translates reductions in odor levels (at various distances) into dollar values. The hedonic property valuation method draws on the economic principle that housing prices are systematically related to a vector of housing characteristics. This systematic relationship can be characterized and measured through a hedonic price function. Using data from 237 home sales between January 1992 and July 1993, the U.S. Census, and the State Veterinarians Office, Palmquist, Roka, and Vukina estimated a hedonic function of the following form:

$$\ln V_i = \alpha_1 + \alpha_2 A_i + \alpha_3 \left[\ln(M1_i + \gamma_2 M2_i + \gamma_3 M3_i)\right] + \varepsilon_i \qquad (6.1)$$

where

- $\ln V_i$ = the logged sales value of property i
- A_i = physical, sociodemographic, and neighborhood characteristics of property i

- $M1_i$ = estimated annual hog manure production within 0 to 0.5 mile of property i
- $M2_i$ = estimated annual hog manure production within 0.5 to 1 mile of property i
- $M3_i$ = estimated annual hog manure production within 1 to 2 miles of property i

The dependent variable in the hedonic function is (the natural log of) the sales value of the 237 home sales in the nine counties. Only parcels that were (1) rural or within a jurisdiction of fewer than 2,500 people and (2) 10 acres or fewer were selected for the analysis. Sales values were obtained from regional Farm Credit Services and real estate listings. In addition to the sales values, data on selected characteristics (A) of the property were also collected from these two sources.[4]

To estimate annual hog manure production in the vicinity of each property (M1, M2, M3), Palmquist, Roka, and Vukina used data from the North Carolina State Veterinarian's Office, which consisted of the number of herds and head capacity of breeding, finishing, and nursery hogs within the three distance categories from each property.[5] To convert animal head capacity estimates (for each of the three distance categories) to estimates of annual manure production, Palmquist, Roka, and Vukina used a linear conversion factor that assumed a constant manure output per hog per year (1.5 tons per year of manure per animal head-capacity). The structure of Eq. (6.1) allowed the researchers to measure and test for whether the marginal effect of manure production on property values declined with distance from the property.

Palmquist, Roka, and Vukina found that hog manure production within 2 miles of a property had a negative and statistically significant effect on its price. In addition, they found that this negative effect declined (in absolute value) with distance from the property.

We use these results to specify a valuation function with respect to changes in manure output. Using the superscript 0 to denote

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[4]Variables used when estimating the hedonic price function included but were not limited to heated area, lot size, bathrooms, age of home, commute time, and family income.

[5]These data did not include the exact location or direction of the hog operations with respect to each property.

baseline conditions and the superscript 1 to denote conditions after a change (i.e., with different hog waste management technology), a change in property value can be expressed as $\Delta V = V^1 - V^0$. Holding all other attributes (A) of the property constant in Eq. (6.1), the change in value associated with changes in local hog manure equivalents can be expressed as

$$\Delta V = V^0 \frac{(M1^1 + \hat{\gamma}_2 M2^1 + \hat{\gamma}_3 M3^1)^{\hat{\alpha}_3} - (M1^0 + \hat{\gamma}_2 M2^0 + \hat{\gamma}_3 M3^0)^{\hat{\alpha}_3}}{(M1^0 + \hat{\gamma}_2 M2^0 + \hat{\gamma}_3 M3^0)^{\hat{\alpha}_3}}. \tag{6.2}$$

Therefore, with information about a parcel's location (with respect to swine operations) and its baseline value (V^0) and local hog manure output, we can use the Palmquist, Roka, and Vukina parameter estimates ($\hat{\alpha}_3$, $\hat{\gamma}_2$, $\hat{\gamma}_3$) in Eq. (6.2) to estimate a household's gain from reductions in manure equivalents (within a 2-mile radius). In other words, we use Eq. (6.2) as a valuation function for each potentially affected household.

To use this valuation function for estimating the benefits of *odor reductions*, we must express odor reductions in terms of *equivalent* reductions in the number of hogs or, specifically, reductions in manure generation (i.e., into equivalent changes in M1, M2, and M3 for each parcel). Absent more detailed empirical evidence on the precise relationship between the size of and odor emissions from hog farms, we assume that reductions in the equivalent number of hogs at a given farm would result in a proportionate reduction in odor. In other words, we assume that an X percent reduction in odor from a farm would have the same effect and can be represented by an equivalent X percent reduction in manure production at the farm.

To apply the valuation function specified in Eq. (6.2) in eastern North Carolina requires that we identify the location and selected characteristics of rural residential properties that are within 2 miles of at least one hog farm. In particular, estimating the price effect for a specific residential property requires estimates of the

- baseline sales value of the home (V^0),

- baseline number of hogs within each distance of the property ($N1^0$, $N2^0$, and $N3^0$), [6] and
- percent reduction the number of hogs and manure output (which is assumed to be equivalent to percent reduction in odor) within each distance.

To maintain consistency with the original study, our analysis initially focused on properties located in the nine counties that were included in the Palmquist, Roka, and Vukina study (see Table 2). We acquired parcel data from Duplin, Johnston, Lenoir, Onslow, Pender, Pitt, Sampson, and Wayne counties. Because of limitations in data availability (i.e., GIS coverages in electronic format), two of the counties in the Palmquist study—Bladen and Green—could not be included in the analysis. Parcel data and corresponding GIS coverages were not available in electronic format for Sampson County, resulting in incomplete coverage. The southeast corner of the county, mostly covering the town of Clinton, was not available from the county tax offices. This missing data may have accounted for 15 to 20 percent of the county area; therefore, our analysis covers the majority but not the entire set of properties potentially affected in Sampson County. However, we added Onslow County to the analysis because of the large number of hog farms located there and the availability of GIS data for the county. As shown in Table 6-1, based on data for North Carolina's inventory of swine operations (NCDENR, 2002), 63 percent of North Carolina hog farms are located in the eight counties included in our analysis.

For the eight selected counties, we then used the available data to identify rural residential properties. Table 6-2 reports the total number of rural residential properties (fewer than 10 acres) located in these counties. Based on GIS coordinates for these properties and for North Carolina hog farms, we identified the subset of these properties that are located within 2 miles of at least one hog farm. Table 6-2 also reports numbers and percentages for these specific properties. A total of 119,716 rural properties in these counties were identified as being located within 2 miles of a hog farm. According to the Palmquist, Roka, and Vukina findings, these are the households most likely to benefit from reductions in farm sizes and/or the associated disamenities such as odor. We then used GIS

[6] Using the same manure-animal head conversion assumption as Palmquist et al. (1.5 tons per animal per year), we can directly translate these numbers to the annual manure equivalents (M1, M2, and M3) needed for using Eq. (6.2).

Table 6-1. County-Level Characteristics and Data Availability

County	Percentage of North Carolina Hog Farms	Included in Palmquist, Roka, and Vukina Study	Included in RTI Analysis	GIS Coverages Available	Property Sales Value Data Available	Property Tax Value Data Available
Bladen	6%	●				
Duplin	22%	●	●	●	●	●
Greene	5%	●				
Johnston	3%	●	●	●		●
Lenoir	3%	●	●	●	●	●
Onslow	3%		●	●	●	●
Pender	3%	●	●	●	●	●
Pitt	3%	●	●	●		●
Sampson	20%	●	●[a]	●[a]		●[a]
Wayne	6%	●	●	●	●	●

[a]Parcel data and corresponding GIS coverages partially missing.

Table 6-2. Number of Geo-Referenced Rural Residential Properties in the Eight-County Study Area

County	Total	With Hog Operations within 2 Miles	Percentage with Hog Operations within 2 Miles
Duplin	19,839	19,307	97%
Johnston	51,583	18,743	36%
Lenoir	15,705	8,880	57%
Onslow	44,436	13,813	31%
Pender	30,675	10,613	35%
Pitt	26,855	8,414	31%
Sampson	15,864	15,258	96%
Wayne	32,202	24,688	77%
8-County Total	237,159	119,716	50%

to spatially link these residential properties to each hog farm within three distances—0 to 0.5, 0.5 to 1, and 1 to 2 miles—and we used the NCDENR inventory of hog operations to determine the hog-head capacity at each of these farms. For these properties, the average number of hogs within 2 miles varies from 5,425 in Johnston County to 26,793 in Duplin County.

One of the important limitations of the available property data from many counties is the lack of information on V^0, the baseline sales values for the homes, which is critical for applying the Eq. (6.2) benefit transfer function. Sales values are not available at all for three counties (including Sampson) and only partially available for four counties. However, with the exception of Sampson County, tax-assessed values are available for virtually all properties in these counties. The lack of either sales ***or*** tax-assessed values for properties in Sampson County is of particular concern because of the large number of farms and properties in this county.

To fill in the gaps in sales value data, where possible we used information on the tax-assessed value to predict sales values. Prediction equations relating these two values were estimated using regression analysis. We first selected 2002 as the year of the analysis and, to the extent possible, converted available sales value data to 2002 dollars using the Office of Federal Housing Enterprise Oversight (OFHEO) House Price Index (HPI).[7] For each of the five counties with sales value data (Duplin, Lenoir, Onslow, Pender, and Wayne), we regressed sales values (natural logarithm, in 2002 dollars) on corresponding tax-assessed values (also natural log). The results are reported in Table 6-3.

As these six equations show, the relationship between sales and tax values is relatively stable across counties. All equations show both a high level of statistical significance (t-statistics in parentheses) and reasonable explanatory power (R-squared values between 20 and 40 percent).

The equation results reported in Table 6-3 were used to estimate sales values for parcels that only had available data for their tax value. We used regressions (1) through (5) respectively for parcels in Duplin, Lenoir, Onslow, Pender, and Wayne counties. For

[7]The OFHEO HPI used was a North Carolina-specific value with historic records available as far back as 1975.

Table 6-3. Regression Results Relating Sales Values and Tax Values for Rural Residential Properties

	Dependent Variable: ln(SALES VALUE)					
	[1]	[2]	[3]	[4]	[5]	[6]
Explanatory Variables	**Duplin N = 6,464**	**Lenoir N = 6,687**	**Onslow N = 26,133**	**Pender N = 10,215**	**Wayne N = 14,476**	**8-County Total N = 63,975**
ln(TAX VALUE)	0.5286 (48.3)	0.4511 (43.57)	0.5156 (90.56)	0.6634 (81.53)	0.5834 (85.73)	0.5744 (166.74)
Constant	4.354 (39.16)	5.519 (50.81)	5.234 (85.18)	3.494 (41.45)	4.497 (63.79)	4.458 (122.85)
R-squared	0.2652	0.221	0.2388	0.3942	0.3367	0.3029

Note: t-statistics in parentheses.

parcels in Johnston and Pitt, we used the pooled regression to predict sales values.

For Sampson County an alternative approach was required to estimate sales values, because neither sales nor tax value data were available. In this case, we assigned to each parcel the median self-assessed property value for its corresponding Census 2000 block group (also converted to 2002 dollars using the HPI).

Table 6-4 provides summary statistics by county of the resulting baseline sales value estimates for rural residential properties in 2002.

Table 6-4. Summary Statistics for Estimated 2002 Sales Values (Rural Residential Properties within 2 Miles of at Least One Hog Operation)

County	N	Mean	Median	S.D.
Duplin	19,307	$34,162	$46,141	$24,803
Johnston	18,743	81,912	52,565	80,631
Lenoir	8,880	42,190	39,047	34,067
Onslow	13,813	67,866	74,011	55,667
Pender	10,613	47,258	71,618	24,558
Pitt	8,414	77,687	52,932	75,200
Sampson	15,258	66,704	54,628	66,704
Wayne	24,688	69,804	48,228	66,780

6.1.2 Results

The estimated aggregate increase in value associated with 10 and 50 percent odor reductions was $5.1 million and $33.8 million respectively for these properties.

Based on these baseline property value (V^0) estimates, we applied the benefit transfer function in Eq. (6.2) to simulate implicit price effects on these homes (almost 120,000 properties). In particular, we estimated the price effect of reducing odor emissions by 10 percent and 50 percent from all hog farms.[8] These odor reductions were simulated by reducing equivalent herd sizes on these farms by 10 and 50 percent, respectively.[9] Note that the 10 and 50 percent reductions were chosen for illustrative purposes only. They are not intended as estimates of the actual reductions expected when alternative technologies are implemented. Information on actual expected reductions will need to be provided by OPEN team researchers when available (see Chapter 1).

On an annualized basis, these values translate to roughly $330,000 and $2.2 million per year, respectively for the properties included in this study.

The estimated benefits associated with these hypothetical odor reductions are summarized in Table 6-5. The estimated aggregate increase in value associated with 10 and 50 percent odor reductions was $5.1 million and $33.8 million respectively for these properties.

Even for the 50 percent reduction scenario, this represents an average of less than $300 per property. On an annualized basis (assuming a 30-year time frame with a 5 percent rate of interest), these values translate to roughly $330,000 and $2.2 million per year, respectively for the properties included in this study.

6.1.3 Uncertainties and Limitations

Because it was originally estimated and applied in our study area and because it has undergone formal peer review, Palmquist, Roka, and Vukina (1997) provides the most defensible and applicable study for this benefit transfer. Nevertheless, the results described above must be interpreted with caution. A number of uncertainties and limitations with this benefit transfer approach deserve further discussion and consideration.

[8] According to information in the NCDENR inventory, 1,501 hog farms are located within at least 2 miles of one of these properties. By design, the model only attributes odor reduction benefits to these farms.

[9] In other words, $M1^1$, $M2^1$, and $M3^1$ were set at 10 and 50 percent below their baseline levels for each farm.

Table 6-5. Estimated Benefits of Selected Hypothetical Odor Reductions for Rural Residents (in thousands of 2002 dollars)[a]

	10% Reduction		50% Reduction	
County	**Property Value Increase (thousands$)**	**Annualized Gain (thousands$)[b]**	**Property Value Increase (thousands$)**	**Annualized Gain (thousands$)[b]**
Duplin	445 (153–737)	29 (10–48)	2,936 (1,003–4,869)	191 (65–317)
Johnston	807 (264–1,349)	52 (17–88)	5,323 (1,731–8,914)	346 (113– 580)
Lenoir	284 (94–472)	18 (6–31)	1,870 (617–3,123)	122 (40– 203)
Onslow	700 (233–1,166)	46 (15–76)	4,617 (1,531–7,703)	300 (100–501)
Pender	378 (125–630)	25 (8–41)	2,492 (819–4,164)	162 (53–271)
Pitt	343 (114–573)	22 (7–37)	2,266 (747–3,786)	147 (49–246)
Sampson	112 (381–185)	73 (25–12)	7,363 (2,495–12,232)	479 (162–796)
Wayne	106 (353–1,762)	69 (23–115)	6,978 (2,315–11,642)	454 (151–757)
Total	**5,129 (172–8,541)**	**334 (112–556)**	**33,846 (11,258–56,433)**	**2,202 (732–3,671)**

[a]95 percent confidence intervals in parentheses.
[b]Assuming a 30-year time frame and a 5 percent interest rate.

First, a potentially important source of uncertainty in the results is captured by the 95 percent confidence intervals provided in Table 6-5. These intervals are based on the statistical error (estimated variance-covariance) associated with the empirically estimated parameters $\hat{\alpha}_3$, $\hat{\gamma}_2$, and $\hat{\gamma}_3$ in Palmquist, Roka, and Vukina, and they indicate how the parameter uncertainty affects the potential range of model predictions. Second, in our analysis, we used the size (number of animal units) of a hog farm as a direct proxy for odor. In particular, we assumed that holding management technology constant at baseline conditions, if farm size were to be reduced by a fixed percentage, would have a directly proportionate effect on odor emissions. Although the size and corresponding level of manure production on a farm is certainly an important contributor to odor, no empirical studies are available to quantify the specific relationship between odor and size (under average lagoon and sprayfield conditions).[10] Although there is no a priori reason to believe that the proportionality assumption biases our results, it does increase the level of uncertainty associated with them.

Third, it is possible that the price effects measured in Palmquist, Roka, and Vukina and used in this study capture effects other than odor that are potentially associated with hog farms. Because Palmquist, Roka, and Vukina did not include a direct measure of odor or include information on the direction between hog farms and properties, it is possible that the parameter estimates implicitly include price effects for other hog farm disamenities, such as other air and water quality impacts or other nuisance (e.g., traffic) effects. It is possible that hog farms are located in areas that are farther from local public goods like schools and parks, in which case the effects measured in Palmquist, Roka, and Vukina partially capture these negative distance effects. In contrast, it is also possible that the parameter estimates capture some countervailing positive amenities (e.g., positive economic impacts) associated with proximity to hog farms. However, taken as a whole, these uncertainties tend to indicate that the measured effects and results summarized in Table 6-5 might overestimate the benefits associated specifically with odor reductions.

Fourth, the analysis only estimates property value effects in eight counties in North Carolina.[11] As shown in Table 6-1, these counties account for a majority (63 percent) of hog farms in the state, but roughly one-third are excluded. The effect of this exclusion is to underestimate odor-related benefits, most likely by 30 to 40 percent if farms in the other counties have similar odor-related impacts.

Finally, the parameters of the hedonic model are designed to estimate the price effects of marginal (i.e., small) changes in

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[10]For example, Sweeten (1998) measures odor levels for two different sized farms (8,400 vs. 200 sow operation) and, as expected, finds lower odor concentrations from the smaller farm; however, differences in waste management technologies used across the two farms makes it difficult to compare them directly.

[11]As previously indicated, data for part of Sampson County are also missing for the analysis.

property attributes. Model results associated with large (nonmarginal) changes in odor are therefore subject to more uncertainty. The larger and more widespread the changes in odor are (and hence more likely to alter the housing market as a whole) the more likely it is that the model results will overestimate the benefits of odor reductions. This suggests, for example, that the benefits estimates associated with 50 percent reductions in Table 6-6 are best interpreted as upper-bound values.

6.2 BENEFIT ESTIMATION FOR IMPROVEMENTS IN SURFACE WATER QUALITY

One of the most important ways in which individuals use and interact with surface waters in North Carolina is through recreational activities such as fishing, boating, and swimming. Therefore, to assess benefits from improvements in nutrient-related surface water quality resulting from alternative hog waste management practices, we focus on recreation-related benefits.

To assess recreation benefits associated with surface water quality improvements, we adapted and re-estimated an existing empirical recreation demand model for North Carolina (Phaneuf, 2002).

Several studies across the United States have documented how changes in water quality can affect recreational choices and, in the process, can reveal individuals' values for better water quality. By combining data on recreation behavior and site-specific water quality, these studies use recreation demand modeling methods to estimate individuals' implicit WTP for improved water quality. In this analysis we used a similar approach. We developed and estimated a recreation demand model that links directly with the results of the water quality model described in Chapter 4. We demonstrate how the model results can be used to estimate benefits for selected reductions in swine-related nutrient loads to surface waters in eastern and central North Carolina.

6.2.1 Methodology and Data

To assess recreation benefits associated with surface water quality improvements, we adapted and re-estimated an existing empirical recreation demand model for North Carolina (Phaneuf, 2002), which was developed with funding from North Carolina's Water Resources Research Institute. The methods and data used in the original model provided much of the structure that was needed for this analysis. However, to specifically address benefits estimation for alternative hog waste management technologies, we adapted the empirical model in several ways to make best use of the water quality modeling output described in Chapter 4.

The modeling approach used for this analysis is best described as a travel cost random utility maximization (RUM) model. RUM approaches are now widely accepted and applied in environmental, transportation, marketing, and several other areas of applied economics to model discrete choice behavior (see McFadden's [2001] Nobel acceptance paper for historical background on the RUM model, technical details, and an application to recreation demand). In many environmental applications, including this one, the behavior of interest is the observed choice of recreation site or location (see, for example, Hausman, Leonard, and McFadden, 1995).

The RUM model seeks to explain a person's recreation decision on a given choice occasion from a discrete set of alternatives. The choice made is based on the characteristics of the alternatives and the importance of these characteristics to the person. Characteristics that are typically of importance in recreation demand models include the implicit price of site access and environmental and amenity aspects of the site. The implicit access price consists of the direct travel cost of reaching the site and the indirect opportunity cost of travel time. For water recreation applications, water quality is typically an important attribute determinant of choice. To model this situation, a conditional indirect utility function, V_j, is specified for each of the J alternatives:

$$V_j = V_j(y, p_j, q_j, \gamma), \quad j = 1,\ldots J, \tag{6.3}$$

where y is income, p_j is the price of a visit to site j (constructed from the travel cost and time cost of the trip), q_j is a measure of the attributes (quality levels) at site j, and γ is a vector of parameters.

The conditional indirect utility function quantifies the benefit to the individual of visiting site j, but this benefit cannot be directly observed. Instead it must be inferred from what can be observed – i.e., the choice of a recreation site. Utility maximization implies that the alternative generating the highest benefit on a given choice occasion will be chosen. Mathematically, site j is assumed to be chosen if

$$V_j > V_i \ \forall \ i \neq j. \tag{6.4}$$

It is assumed that the conditional indirect utility functions are composed of a systematic component that is observable to the investigator and a component that is random to the investigator but known by the individual. The random component accounts for unobserved heterogeneity in consumer preferences and nonmodeled characteristics of the available recreation sites. Typically the systematic and random components are assumed to enter the functions linearly, as follows:

$$V_j\,(y, p_j, q_j) = \beta\,(y - p_j) + \delta q_j + \varepsilon_j, \quad j = 1,\ldots,J, \tag{6.5}$$

where the model parameters are γ=(β,δ). Under this specification, β is the effect on utility of changes in money (price or income), and δ is the marginal impact of changes in quality.

From the perspective of the analyst the problem is now probabilistic. The probability that an individual will visit a given site on a given choice occasion is the probability that the site has the highest associated utility. Thus, the probability of a visit to site j is

$$\begin{aligned} \text{prob (j)} &= \text{prob}\,(V_j > V_i) \\ &= \text{prob}\,(v_j + \varepsilon_j > v_i + \varepsilon_i) \\ &= \text{prob}\,(v_j - v_i > \varepsilon_i - \varepsilon_j) \quad \forall i \neq j. \end{aligned} \tag{6.6}$$

If it is assumed that the random terms are distributed type I extreme value, a multinomial logit model of site choice emerges (McFadden, 2001). This is a convenient assumption, because it provides for a closed form for the probability of visiting a site on a given choice occasion, given by

$$\ln\,[\text{prob (j)}] = v_j - \ln\left(\ln\sum_{k=1}^{J} e^{v_k}\right) \tag{6.7}$$

This (log) probability can be specified for each choice occasion observed in the sample and maximum likelihood used to recover estimates of the parameter vector.

Estimation of the parameter vector provides a characterization of consumer preferences for the attributes of the sites, up to the unobserved random term, via the conditional indirect utility functions. This characterization can be used to measure the monetary benefits or damages from changes in site characteristics.

Compensating variation (CV, also referred to as a change in consumer surplus) is a theoretically consistent measure of these benefits. By definition, compensating variation is the amount of money that would need to be taken away from an individual following an improvement in the resource such that he/she would be exactly as "well off" as he/she was before the change. For random utility models that are linear in price, expected compensating variation is given by

$$E(CV) = -\frac{1}{\beta}\left[\ln\sum_{j=1}^{J} e^{\hat{v}_j^0} \quad \ln\sum_{j=1}^{J} e^{\hat{v}_j^1}\right] \tag{6.8}$$

where β is the estimated coefficient on price, $\hat{v}_j^0$ is the predicted deterministic component of utility with the original value of the characteristics, and $\hat{v}_j^1$ is the function with the new values of the characteristics, the change for which we are measuring the welfare effect. For a recreation site quality improvement, compensating variation is a measure of WTP for the quality improvement and can be interpreted as the amount visitors would pay to purchase the water quality improvement. The equation is calculated for each person in the sample and the mean used to provide an estimate of the WTP in the population. Note that, because the model is set up to analyze a choice occasion, the units on the welfare effect are dollars per choice occasion.

To calculate individual annual and aggregate annual benefits from RUM models of site choice, additional assumptions and caveats are needed. If visitors do not change the number of trips they make in response to a quality improvement, the annual value of the improvement is the product of the number of trips taken and the per-trip welfare measure. If, as would be expected, people take more trips when quality improves, this annual measure is a lower bound on total WTP. Similarly annual aggregate benefits are calculated by estimating the total number of trips made by the population of interest to the area under study and scaling this by the per-trip WTP for the improvement.

To support benefits assessment for this application, the RUM needed to be designed with the same spatial dimension as the water quality model described in Chapter 4. This spatial link between the models was required so that predictions from the water quality model could be used as explanatory variables in the economic model and changes in these predictions could be used to assess the monetary value of the improvements. As described in Chapter 4, the water quality model was designed to predict surface water quality at the level of the 14-digit hydrological unit (14-digit HUC) for a defined area of eastern North Carolina. We therefore defined the choice set for the RUM model as the set of 14-digit HUCs that fall within this eastern North Carolina study area. In other words, we treated each 14-digit HUC in the study area as a discrete option j for water-based recreation.

Estimation of the RUM required data on recreation trips to the study area, including most importantly information about the origin and destination of each trip. These data for constructing the RUM model came from two main sources: the 1994 National Survey of Recreation and the Environment (1994 NSRE) and, to a lesser extent, the 2000 NSRE. The 1994 NSRE was a collaborative effort between several federal agencies and consists of four individual components. EPA administered the National Demand for Water Based Recreation Survey as a component of the overall 1994 NSRE. This survey used a random digit dialing population-based sample, stratified to ensure adequate representation of each state, to assess the recreation use of water resources in the country. Approximately 16,000 responses are available nationwide. Individuals were asked to report information on boating, swimming, fishing, and viewing or near shore recreation activities. Detailed information on the most recent trip taken for each of these four activities was solicited. This includes the name and location of the water body, the type of water body, whether the trip was for a single day, and activity-specific information. Individuals also provided information on the number of trips they made to this site, for this activity, during the year. Finally, demographic information is available, including household income and home location. For the original study (Phaneuf, 2001; 2002), this information was used to construct a database of activity-specific trip-taking outcomes and the frequency of visitation for sites throughout North Carolina. A subset of this information corresponding to trips taken in southeastern North Carolina was selected for this study.

Beginning in 1999 several federal agencies began work on a new version of the NSRE, known as the 2000 NSRE. This survey followed a similar format to the 1994 version but had several improvements. As in 1994, it was stratified to provide good representation of residents in each state. In particular, information of some type was obtained from 677 North Carolina residents. The survey was divided into several modules soliciting information on specific types of recreation use. We focused on the freshwater recreation module. This module provides trip-specific information useful for our RUM model and information needed for the aggregate benefits estimates.

The survey solicited detailed information on the most recent freshwater trip, followed by questions on a second trip. The information obtained for both first and second trips is the same. Thus, each person can provide information on two unique visits. In addition, the survey provides information on the number of times a visit to this site occurred during the year. The variables of interest include the state in which the visit occurred, the name of the water body, the city nearest the water body, the frequency of this visit, and information about the respondent.

The specific steps involved in constructing and using the model employing the NSRE and the modeled water quality data are the following:

- Select observations on trips from the two NSRE data sets that occurred in the water quality model zone.
- Identify the 14-digit watershed (also referred to as a 14-digit "HUC") in which the trip occurred.
- Construct a matrix of prices (travel costs) for visits to each of the zones for each individual trip-taker represented in the data.
- Obtain predictions from the water quality model of baseline water quality conditions at each of the 14-digit HUCs.
- Estimate the RUM model of site choice as a function of travel costs and baseline quality levels.
- Obtain predictions from the water quality model for new conditions and use the RUM model to predict WTP for the improvements.

Both the 1994 and 2000 NSRE provide nationwide data on water-based recreation trips; therefore, we began by selecting trips from these datasets that occurred in the eastern North Carolina study area. The study area was originally defined to include 635 14-digit HUCs, so we treated these HUCs as the relevant choice set. Based on this specification, we identified data for 175 water-based recreation trips in the study area, including 155 trips from the 1994 NSRE and 20 trips from the 2000 NSRE. We used GIS software and the trip location information provided by the NSRE data to identify the 14-digit HUC corresponding to each of the 175 trips. Because the 1994 survey provides detailed information on each of four water-based activities and the 2000 survey provides detailed information on the two most recent trips, in some cases the 175 observations include multiple trips for a single respondent. When combined with NSRE data on trip frequency (respondents' reports of the number of trips made to the identified site during the year), the combined data provide information on a total of 695 trip-choice outcomes.

Estimation of the RUM requires information on travel costs for each respondent for each destination in the choice set. In other words, we needed to calculate the vector of "prices" for each of the trip-taking agents for the 635 alternatives available to them. As is commonly done, we assumed that travel cost is a function of the distance between a respondent's home and the location of the trip destination. The home location was approximated by the zip code centroid for each NSRE respondent, and the destination location for each 14-digit HUC was also approximated by its centroid. We used the software package PCMiler to compute the distance in miles and road travel time between each respondent's zip code and the 635 destinations. This provided us with a 175 X 635 matrix of one-way distances and a 175 X 635 matrix of one-way travel times.

Based on estimates of travel distance and travel time, travel costs per trip can be calculated in a variety of ways. For our analysis, we used the following specification for round-trip travel costs ($PRICE_{ij}$) for respondent i to destination (14-digit HUC) j:

$$PRICE_{ij} = [(\$0.21) * (DISTANCE_{ij})] + [(0.33) * (INCOME_{ij} / 2000) * HOURS_{ij}] \quad i=1,..,175,\ j=1,...,635 \tag{6.9}$$

In this equation the first term is the out-of-pocket travel cost and the second term is the opportunity cost of travel time. DISTANCE is the round-trip travel distance in miles; HOURS is the round-trip travel time in hours; and INCOME is the annual income for the respondent (in 1994 dollars), which is divided by the average number of working hours in a year to arrive at the average wage rate. This specification assumes a per-mile out-of-pocket cost of

$0.21 and an opportunity cost of travel time equal to one-third the average wage rate, a common assumption. There is a large literature in recreation demand on how travel costs should be calculated, particularly with respect to the opportunity cost of time (see McConnell and Strand [1981] and Larson [1993] for different perspectives on using a fraction and the full wage rage, respectively, and Shaw [1992] for an early overview of the topic). Our assumptions tend to be conservative in that the American Automobile Association currently uses $0.33/mile as the cost of road transportation, and a fraction (as opposed to the full) wage rate for the opportunity cost of time implies smaller implicit trip costs. These decisions suggest the welfare measures from our RUM model will be smaller than under alternative assumptions and can be thought of as more of a lower bound.

As development of the RUM and water quality models progressed, it became clear that it would be difficult and perhaps inappropriate to estimate the model with the full choice set of 635 HUCs. The water quality model is predominantly an inland model for freshwater rivers and streams; however, several of the most downstream HUCs in the study largely comprise estuarine or coastal waters. As a result, we could not ultimately include 72 of the original 635 14-digit HUCs identified as the study area in the water quality model.

Effectively linking the RUM and water quality model therefore required one of the following steps:

- Option 1—Fully Restricted Choice Set: Drop from the RUM choice set all 72 of the HUCs that do not contain a direct prediction from the water quality model;
- Option 2—Unrestricted Choice Set: Use the results from the 565 HUCs included in the water quality model to construct (indirect) predictions of water quality for all 72 of the excluded HUCs; or
- Option 3—Partially Restricted Choice Set: Drop the HUCs closest to the coast and construct (indirect) predictions for the remaining missing HUCs.

Inspection of the trip-taking data suggested that Option 1 would preclude estimation of the RUM because dropping all 72 HUCs absent in the water quality model would leave insufficient observations to identify the parameters of the RUM model. Option 2 would provide the most data for model estimation, but the RUM estimates would be most uncertain because predictions for the coastal HUCs (constructed from the included inland HUCs) would likely be highly inaccurate.

Option 3, which included a combination of dropping HUCs and constructing predictions for others, was defined as a compromise approach for the final model. Using best professional judgment, we identified 34 of the missing 72 HUCs that would best be dropped from the model and 38 HUCs for which water quality could reasonably be approximated using information from the directly modeled HUCs.

Although Option 3 was the preferred approach, Option 2 was retained for comparative purposes. To implement both options, we generated water quality predictions for all 72 missing HUCs using variations of the same general approach. For a small number, we identified a single adjacent HUC that would serve as a suitable proxy for the water quality level in the missing HUC. For the majority of the missing HUCs, we substituted the average modeled predictions from all of the other 14-digit HUCs that were modeled and were located in the same larger eight-digit HUC. When possible, we used the same strategy at a higher resolution using 11-digit HUCs.

The results of the RUM estimation using both Options 2 and 3 are described below.

6.2.2 Results

Using a multinomial logit (MNL) specification, we estimated the following general RUM specification for both the unrestricted and partially restricted choice sets (Options 2 and 3):

$$v_j = \beta \, price_j + \gamma TN_j + \delta TP_j + \varepsilon_j, \quad j = 1,\ldots,635, \qquad (6.10)$$

where $price_j$ is the travel cost of reaching site j, and TN_j and TP_j are the predictions at site (HUC) j for total nitrogen and total phosphorus concentrations, respectively. Because water quality impairments are assumed to decrease a site's attractiveness, we expect negative coefficient estimates for total nitrogen and total phosphorus. Model results for both options are reported in Table 6-6.

Table 6-6. Regression Results for Trip Choice RUM (Multinomial Logit Model)[a]

Model Specification	Option 2 (Unrestricted Choice Set) (1)	(2)	(3)	Option 3 (Partially Restricted Choice Set) (4)	(5)	(6)
Travel cost (PRICE)	–0.0313 (–23.81)	–0.0331 (–25.57)	–0.0371 (–24.48)	–0.0465 (–15.43)	–0.0477 (–16.11)	–0.0454 (–15.151)
Total nitrogen (TN)	0.178 (3.679)	–0.2657 (–7.633)		–0.24 (–2.249)	–0.4312 (–5.872)	
Total phosphorus (TP)	–13.207 (–10.689)		–9.861 (–12.475)	–5.031 (–2.269)		–9.215 (–6.385)
Size of choice set (HUCs)	635	635	635	601	601	601
Number of choice outcomes	695	695	695	207	207	207

[a]t statistics in parentheses below coefficient estimates.

Regression Results Using the Unrestricted Choice Set (Option 2)

Using water quality approximations and corresponding travel cost estimates for the full choice set when both TN and TP are included (first specification in Table 6-6), the coefficients on price and TP are negative and significant as expected. The coefficient on TN, however, is positive and significant, suggesting increases in nitrogen increase the attractiveness of a recreation site. This counter-intuitive result is almost certainly due to co-linearity between the two pollutant measures, which is exacerbated by the large number of missing HUCs that needed to be predicted. To confirm this suspicion, we also ran models that included each pollution prediction alone. When only TN is included (second specification in Table 6-6), we find that the coefficient on TN is negative and significant as expected. When only TP is included (third specification in Table 6-6), we again find TP to be negative and significant. Taken as a whole, these results suggest that sites with better water quality are more attractive to recreators than those with worse water quality, all else equal.

Regression Results Using the Partially Restricted Choice Set (Option 3)

The reduced choice set model consists of 601 sites for which only 54 individual trip observations are available, containing information on 207 choice outcomes. Using this partially restricted choice set, we estimated the same three model specifications. In all three specifications, the travel cost coefficient is negative as expected. When we include both TN and TP, the estimated coefficients are both negative and significant as *a priori* expected. These results suggest that co-linearity is less of a problem for this model. Co-linearity is likely smaller for this choice set because fewer missing HUCs are included; hence, a smaller number need to be predicted. The results are similar in the last two specifications, where TN and TP are included separately. Taken as a whole, these estimates support the hypothesis that water quality matters in the choice of recreation site, and despite the large reduction in data the parameters are still identified with an acceptable level of precision. Furthermore, and in contrast to the full choice set model above, quantitatively similar estimates are found across the three specifications.

Estimated Per-Trip Benefits from Reductions in Swine-Related TN and TP Loads to Surface Water

To assess per-trip benefits of selected improvements in water quality, we applied the model results summarized in Table 6-6. For each model option and specification, Table 6-7 provides both the estimated average baseline consumer surplus per trip and the estimated average gain in per-trip consumer surplus associated with two scenarios. The two scenarios are defined respectively as hypothetical 10 and 50 percent reductions in swine-related surface water loadings of TN and TP in the study area.

Although the magnitudes of the welfare estimates are plausible for all specifications, there are several reasons to discount the estimates from the unrestricted choice set models (Option 2, specifications [1], [2], and [3]). First, the positive sign on TN in the complete specification makes it difficult to interpret the effects of improvements in water quality, because decreases in total nitrogen will lead to reduce utility levels at recreation sites. This in turn makes it difficult to use the full specification for welfare analysis. As noted above, the sign of this estimate is likely due to co-linearity, the degree of which is increased beyond the normal correlation in nitrogen and phosphorus concentrations by the large number of predictions needed to fill in gaps in the water quality model. Each

Table 6-7. Per-Trip Benefit Estimates for Selected Water Quality Scenarios (in 2002 dollars)[a]

	Option 2 (Unrestricted Choice Set)			**Option 3 (Partially Restricted Choice Set)**		
Model specification	(1)	(2)	(3)	(4)	(5)	(6)
Baseline per-trip consumer surplus	$19.41 (8.43)	$36.31 (8.29)	$17.19 (8.37)	$36.65 (8.86)	$39.58 (8.64)	$37.74 (9.72)
Gain in per-trip consumer surplus						
10 percent loadings reduction	$0.15 (0.73)	$0.45 (0.04)	$0.34 (0.02)	$0.42 (0.06)	$0.48 (0.04)	$0.28 (0.03)
50 percent loadings reduction	$0.75 (0.33)	$2.50 (0.22)	$1.88 (0.11)	$2.42 (0.40)	$2.80 (0.24)	$1.52 (0.06)

[a]Estimated standard errors in parentheses.

of these suggests the partially restricted choice set model is preferred.

The per-trip gains for a 10 percent reduction in loadings from hog farms are estimated to be between $0.28 and $0.48 per trip (about 1 percent of the value of a trip). For a 50 percent reduction in loadings, the per-trip gains increase to between $1.52 and $2.80.

Using the partially restricted choice set model, the baseline consumer surplus estimates shown in Table 6-7 are about $37 per trip.[12] The per-trip gains for a 10 percent reduction in loadings from hog farms are estimated to be between $0.28 and $0.48 per trip (about 1 percent of the value of a trip). For a 50 percent reduction in loadings, the per-trip gains increase to between $1.52 and $2.80.

Estimated Aggregate Benefits

RUM models are designed to provide estimates of the per-trip increase in consumer surplus from a quality change accruing to an individual. Additional information and the assumptions described above are needed to translate these per-trip estimates into estimates of the total annual benefits for North Carolina residents. From an operational perspective, estimating aggregate benefits requires estimates for

- ➤ the total number of trips taken in North Carolina each year,
- ➤ the proportion of these trips that occur in the model zone, and
- ➤ the relevant trip-taking population in the state.

[12]All values are updated to 2002 dollars using the consumer price index (CPI).

The general strategy we used to estimate aggregate benefits was to estimate the total number of annual freshwater recreation trips that occur in the model zone and multiply this number by the per-trip consumer surplus estimate. To first calculate the total annual number of trips in North Carolina, we used information from the 156 North Carolina residents who answered questions in the freshwater module of the 2000 NSRE. Of these respondents, 54 percent indicated they took single-day trips to freshwater sites in the state during the previous 12 months.[13] Among these participants, the median number of trips was 5 and the average number was 12. We focus on the median as a conservative estimate of total annual trips, because it discounts the influence of individual outliers who took large numbers of trips. According to U.S. Census Bureau data, the 2002 adult (over 18 years old) population of the state is 6.15 million. Based on the NSRE 54 percent participation rate, a rough estimate of annual trip takers in North Carolina is 3.26 million people. Using the NSRE trip frequency estimates, the estimated total number of trips in North Carolina is therefore 16.3 million, using the NSRE's median number of trips among participants (39.1 million using the NSRE average).

To next estimate the portion of these trips that occurred in the study area, we again used information from the NSRE. In geo-coding the individual trips from the 2000 NSRE survey for inclusion in the RUM model, we determined that approximately 22 percent of the statewide trips occurred in the water quality model zone. Applying this fraction to our approximation of total annual trips, we estimated between 3.6 million freshwater-based recreation trips per year to the study area.

Finally, to estimate aggregate benefits for the two loadings reduction scenarios, we multiplied the estimated per-trip consumer surplus gains (from Table 6-7) by the estimated total number of annual trips (3.6 million). Using the preferred model specification (column [4] in Table 6-6 and 6-7, which includes both TN and TP), the total recreation benefits in the study area for the two scenarios were estimated to be:

- 10 percent loadings reduction: $1.5 million (with a 95 percent confidence interval of $1.1 to $1.9 million).

[13] NSRE respondents are selected at random from the population, including individuals who do not participate in water-based recreation.

- 50 percent loadings reduction: $8.7 million ($5.9 to $11.5 million).[14]

6.2.3 Limitations and Uncertainties

A number of assumptions inherent in our modeling effort should be kept in mind when interpreting results. First, recreation demand models of the type presented here focus on the benefits of water quality changes that accrue to current recreation users only. The model does not provide information on benefits associated with nonrecreation aspects of water quality, nor does it account for the benefits that may accrue to individuals who currently are nontrip takers but may later take trips in response to higher quality. Second, our annual benefits estimates are based on an estimate of the number of trips taken under current conditions, suggesting our estimates do not account for the likely increase in annual trips in response to improved water quality. Each of these two factors, combined with our use of conservative figures in constructing travel cost estimates for the sample and total trip estimates for the study area, suggests our benefit estimates can be interpreted as a lower bound on the WTP for surface water quality improvements.[15]

Several sources of uncertainty also need to be acknowledged and considered when interpreting the benefits estimates from the RUM model. First, as with any statistical modeling effort there is parameter estimate uncertainty and model/specification uncertainty. The confidence intervals on the aggregate benefit measures address the first type of uncertainty and provide a sense of the precision at which the model is estimated. The effect of specification uncertainty can be seen in part by comparing the results across the columns in Tables 6-6 and 6-7. In addition, the task of overlapping the RUM and water quality models required compromises on the number of observations available to estimate the RUM model. These compromises contribute to model uncertainty (as can be seen by comparing results from Option 2 and Option 3), and they limited the complexity of the specifications we were able to explore. Nonetheless, the parameter estimates and welfare calculations are consistent with the range of values seen in the literature.

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[14]The 95 percent confidence intervals were constructed based on the standard error estimates for the per-trip welfare gains reported in Table 6-7.

[15]For example, if we used the average number of annual trips from our sample (12 per year) rather than the median (5) to estimate total trips to the study area, the benefit estimates would increase by a factor of 2.4.

Finally, several sources of uncertainty accumulate in the steps used to estimate aggregate annual benefits. These sources include errors in the estimation of annual trips, population-wide participation, the proportion of trips occurring in the model zone, and the total adult trip-taking population. The cumulative size and direction of these effects on the final results are unfortunately difficult to quantify.

6.3 BENEFIT ESTIMATION FOR REDUCTIONS IN AMBIENT PM_{FINE} LEVELS

In addition to reducing deposition and runoff of nutrients to surface waters, reductions in ammonia emissions from hog farms are also expected to reduce ambient levels of fine particulate matter (PM_{Fine}) in eastern and central North Carolina. Human exposures to elevated levels of PM_{Fine} have been associated with a wide variety of adverse health effects. These effects range from relatively minor acute conditions respiratory conditions to increased mortality risk (particularly in older populations).

To assess the benefits of reducing PM_{Fine} levels through alternative hog waste management technologies, we applied a two-part modeling framework. With this framework, we first estimated the reduced incidence of selected health effects resulting from reduced PM_{Fine} exposures. We then applied a benefit transfer model to estimate total WTP to avoid these effects. The modeling approach and results are described below.

6.3.1 Data and Methodology

As described in Chapter 3, RTI developed an environmental model to estimate reductions in county-level ambient PM_{Fine} levels in the study area associated with reductions in ammonia emissions from hog farms. To estimate benefits for these PM_{Fine} reductions, we used a model previously developed by EPA. EPA has most recently used this model in its benefits analysis of the proposed Nonroad Landbased Diesel Engine Rule. A detailed description of the model is provided in EPA's documentation of their analysis (EPA, 2003). In our analysis, we adapted and applied the model to measure and value the health-related benefits that are expected to result from the modeled reductions in PM_{Fine}.

EPA's PM health benefits model has two main components. The first component uses evidence from the epidemiology literature to estimate reductions in the annual incidence of specific health outcomes associated with reductions in ambient PM_{Fine} levels. Based on the findings in this literature, EPA specified concentration-response (CR) functions to estimate the reductions in incidences of health outcomes such as premature mortality, chronic bronchitis, hospitalization for asthma, and acute illnesses such as lower respiratory symptoms. Table 6-8 provides a complete list of the modeled health outcomes. It also lists for each outcome the epidemiological studies and CR functions used and the population affected. As shown in the table, the age group of the study population differed across health outcomes.

Table 6-8. Epidemiological Studies, Targeted Population Groups, and CR Functions for Health Outcomes Affected by PM_{Fine}

Health Outcomes	Epidemiological Study	Study Population	CR Function[a]
Mortality	Krewski et al. (2000)	30 and over	$(e^{\beta.\Delta PM} - 1) \times (INC)_{county} \times POP_{30+}$
Chronic bronchitis	Abbey et al. (1995)	27 and over	$(e^{\beta.\Delta PM} - 1) \times INC \times POP_{27+} \times 0.9465$
Asthma hospitalization	Sheppard et al. (1999)	Under 65	$(e^{\beta.\Delta PM} - 1) \times INC \times POP_{65-}$
Acute bronchitis	Dockery et al. (1996)	8 to 12	$\left(\frac{INC}{(1 - NC)e^{\beta \Delta PM} + INC} \quad INC\right) \times POP_{8\text{-}12}$
Lower respiratory symptoms	Schwartz et al. (1994)	7 to 14	$\left(\frac{INC}{(1 - INC)e^{\beta \Delta PM} + INC} \quad INC\right) \times POP_{7\text{-}14}$
Work loss days	Ostro (1987)	18 to 65	$(e^{\beta.\Delta PM} - 1) \times INC \times POP_{18\text{-}65}$
MRAD	Ostro and Rothschild (1989)	18 to 65	$(e^{\beta.\Delta PM} - 1) \times INC \times POP_{18\text{-}65}$

[a]CR functions estimate Δy (change in annual incidence of the health outcome) using β (estimated coefficient from the epidemiological study) and data on POP (size of the affected population, as specified in subscript), and INC (baseline annual incidence rate per person of the health outcome).

The second component uses evidence from the health valuation literature to estimate the dollar value of the avoided health outcomes. The estimates are expressed as unit (i.e., per case avoided) values. Where possible, EPA's model uses available estimates of individuals' average WTP to reduce the probability or certainty of experiencing the specific health outcomes. For example, to value reductions in premature mortality, the model uses estimates of individuals' WTP to reduce risks of death, which are translated into "value of statistical life" terms (i.e., value per avoided premature death). A similar approach is used for valuing reductions in the incidence of chronic and acute bronchitis, lower respiratory symptoms, and minor restricted activity days (MRAD). As a second best alternative, when WTP estimates are not available in the literature for health effects of interest, EPA's model uses cost-of-illness (COI) estimates. COI estimates are used mainly for hospitalizations for asthma and for work loss days. COI estimates only capture avoided health-related expenditures and/or lost earnings due to illness but not the value of avoided pain and suffering; therefore, they are interpreted as lower-bound estimates of individuals' total WTP to avoid illness. The unit value estimates used in EPA's PM health benefits model (converted to 2002 dollars using the consumer price index) are summarized in Table 6-9.

Table 6-9. Unit Values Estimates for Modeled Health Outcomes

Health Outcomes	**Value Estimates (thousands 2002 $ per avoided incidence)**
Mortality	6,093[a]
Chronic bronchitis	356
Asthma hospitalization	7.4
Acute bronchitis	0.062
Lower respiratory symptoms	0.017
Work loss days	0.114
MRAD	0.052

[a]Assumes a mean lag of 5 years between exposure and death; average of estimates using 3 percent and 7 percent discount rates.

To operationalize EPA's model for our analysis and apply the CR functions for each health outcome, we needed county-level estimates of incidence rates for the health outcome, population size for the demographic group specified in the CR functions, and the change in average ambient PM_{Fine} level. National averages for incidence rates were used for all counties in the study area for all health outcomes except mortality. National incidence rates and coefficients (β) used in the CR function are summarized in Table 6-10. To estimate county-specific incidence rate for mortality, the number of nonaccidental deaths in the 30-or-more (30 plus) year age group was divided by the total 30-plus population in the county. Population size for selected age groups within each county was estimated using population estimates for 2002, which are based on U.S. Census 2000. If a county was partially included in the study area, then the population was adjusted in proportion to the geographical area of the county that falls within the study area. The environmental model described in Chapter 3 provided the final set of required inputs for running the PM benefit model—estimated changes in average county-level ambient PM levels. It is important to emphasize that, based on the specification of the CR functions, the PM health benefits model does not require information on *baseline* PM_{Fine} levels. In other words, health benefits are assumed to depend only on the change in PM_{Fine} levels and not on baseline levels.

Table 6-10. Coefficient and Incidence Rates Used in CR Functions

Health Outcomes	Coefficient (β)	Annual Incidence Rate (INC) (additional cases per person)
Mortality	0.0046257	County specific
Chronic bronchitis	0.0132	0.00378
Asthma hospitalization	0.0027	0.00000395
Acute bronchitis	0.0272	0.043
Lower respiratory symptoms	0.01823	0.0012
Work loss days	0.0046	0.00595
MRAD	0.0022	0.02137

6.3.2 Results

To assess PM-related benefits using the model described above, we evaluated two scenarios in particular: 10 and 50 percent reductions in *modeled* PM_{Fine} levels across the study area. Because the modeled PM_{Fine} levels only account for the contribution of ammonia emissions from hog farms in the study area, these two scenarios also correspond to 10 and 50 percent reductions in ammonia emissions.

The total central estimate of health-related benefits sums to $38 million and $189 million per year respectively for the 10 and 50 percent ammonia emission and PM_{Fine} reduction scenarios.

The results are summarized in Table 6-11, which presents the central estimates of avoided incidence and corresponding benefits in monetary terms. The largest estimated incidence reductions are for acute conditions, in particular lower respiratory symptoms and MRADs; however, the total value of these avoided cases represents a relatively small portion of total benefits of PM_{Fine} reduction. Estimated reductions in premature deaths—six per year for the 10 percent reduction scenario and 32 per year for the 50 percent reduction scenario—account for a large majority of the total benefits. The total central estimate of health-related benefits sums to $38 million and $189 million per year respectively for the 10 and 50 percent ammonia emission and PM_{Fine} reduction scenarios.

6.3.3 Uncertainties and Limitations

The key components of EPA's modeling framework described above for estimating PM-related benefits (i.e., the CR functions and the valuation estimates) have been extensively reviewed by the Agency. Nevertheless, the model contains limitations and uncertainties, which are important to consider when interpreting the model results. Consequently, the results presented in Table 6-11 are best interpreted as midpoint estimates within a range of uncertainty.

In Table 6-11, roughly 95 percent of the benefits are attributed to avoided morality; therefore, the results are particularly sensitive to this value.

Table 6-12 summarizes many of the primary sources of uncertainty that EPA has identified. Most sources relate to the selection and application of CR functions and valuation methods. In this application, one of the key sources of uncertainty is the value assigned to avoided mortality. In Table 6-11, roughly 95 percent of the benefits are attributed to avoided morality; therefore, the results are particularly sensitive to this value. Based on an extensive review and analysis of the mortality valuation literature, EPA's framework uses a midpoint of between $6 million and $7 million per premature death avoided; however, it is acknowledged the most defensible value estimates from the literature tend to range between $4 million and $9 million (Viscusi, 1993). This uncertainty alone implies that the total benefit estimates fall within a range of $25 to $55 million for the 10 percent reduction scenario, and $127 to $276 million for the 50 percent reduction scenario.

Table 6-11. Central Estimates for Benefits of Reductions in Ambient PM_{Fine} Levels

	10% Reduction		**50% Reduction**	
Reduction in PM_{Fine} Concentration ($\mu g/m^3$)[a]	0.0373		0.187	
Avoided Health Outcomes	**Number of Cases Avoided (cases/yr)**	**Value of Cases Avoided ($millions/yr)[b]**	**Number of Cases Avoided (cases/yr)**	**Value of Cases Avoided ($millions/yr)[b]**
Mortality	5.94	36.22	29.68	180.81
Chronic bronchitis	4.41	1.46	20.49	7.29
Hospitalization—asthma	0.50	0.0034	2.5	0.02
Acute bronchitis	11.92	0.0007	59.14	0.0037
Lower respiratory symptoms	134	0.0022	667	0.01
Work loss days	918	0.13	4582	0.67
MRAD	1577	0.08	7892	0.41
Total monetized health benefits ($millions/yr)[b]		37.90		189.21

[a]County-level population-weighted average.
[b]In 2002 dollars.

Another key source of uncertainty has to do with which constituents of fine particulate matter contribute most to observed health effects observed (McCubbin et al., 2002). Fine particles originating strictly from swine sources may have different effects than those measured in the epidemiological studies listed in Table 6-8. For example, they may be less harmful than fine particulates from combustion sources; however, evidence regarding these differences is still limited.

As indicated in Table 6-12, another potentially important source of uncertainty is the set of PM-related health effects for which defensible CR functions are not available. It is possible that by failing to quantify the full range of health effects, the modeling framework underestimates the benefits of reducing PM_{Fine} levels.

6.4 BENEFIT ESTIMATION FOR REDUCTIONS IN GROUNDWATER NITRATE LEVELS

As discussed in Chapter 5, surface loadings of nitrogen are potentially important contributors to elevated nitrate levels in

Table 6-12. Primary Sources of Uncertainty in the PM_{Fine} Benefits Analysis[a]

1. *Uncertainties Associated with Concentration-Response Functions*
    - The value of the β coefficient in each CR function.
    - Application of a single CR function to pollutant changes and populations in all locations.
    - Similarity of future year CR relationships to current CR relationships.
    - Correct functional form of each CR relationship.
    - Extrapolation of CR relationships beyond the range of PM concentrations observed in the study.
    - Application of CR relationships only to those subpopulations matching the original study population.
2. *Uncertainties Associated with PM Mortality Risk*
    - No scientific literature supporting a direct biological mechanism for observed epidemiological evidence.
    - Direct causal agents within the complex mixture of PM have not been identified.
    - The extent to which adverse health effects are associated with low level exposures that occur many times in the year versus peak exposures.
    - The extent to which effects reported in the long-term exposure studies are associated with historically higher levels of PM rather than the levels occurring during the period of study.
3. *Uncertainties Associated with Possible Lagged Effects*
    - The portion of the PM-related long-term exposure mortality effects associated with changes in annual PM levels that would occur in a single year is uncertain as well as the portion that might occur in subsequent years.
4. *Uncertainties Associated with Baseline Incidence Rates*
    - Some baseline incidence rate estimates may not accurately represent the actual location-specific rates.
5. *Uncertainties Associated with Economic Valuation*
    - Unit dollar values associated with health welfare endpoints are only estimates of mean WTP and therefore have uncertainty surrounding them.
    - Mean WTP (in constant dollars) for each type of risk reduction may differ from current estimates because of differences in income or other factors.
6. *Uncertainties Associated with Aggregation of Monetized Benefits*
    - Health benefits estimates are limited to the available C-R functions. Thus, unquantified or nonmonetized benefits are not included.

[a]Adapted from EPA (2003).

groundwater. These elevated levels can in turn pose health risks to humans. Nitrate concentrations above 10mg/l are of particular concern because exposures at these levels have been found to cause methemoglobinemia (“blue baby syndrome”) in bottle-fed infants. Even at levels below 10mg/l, nitrate exposures may contribute to health risks; however, evidence of these effects is very limited.

In Chapter 5 we described the results of a regression model, which examined nitrate levels in private drinking water wells in eastern North Carolina and evaluated their potential association with swine farm sources of nitrogen. Based on a limited sample of relatively shallow groundwater wells (a majority of which had concentrations below 10 mg/l) we found evidence of a small but statistically significant effect related to swine farms. We also estimated that, even by eliminating the nitrogen contribution from local farms, the average predicted change in nitrate levels for our sample of 218 wells was less than 1 mg/l.

The results of our empirical model suggest that the measurable groundwater benefits associated with alternative hog waste management technologies are likely to be relatively small. Therefore, rather than attempting the very data intensive task of identifying all of the potentially affected wells in eastern North Carolina and spatially linking them to the inventory of hog farms, we have conducted a more limited exploratory analysis of potential benefits. The methods and results of this exploratory analysis are described below.

6.4.1 Data and Methodology

The results of the predictive simulations described in Chapter 5 imply that a reduction in swine farm loadings would reduce nitrate levels in the sample of 218 drinking water wells by an average of no more than 1 mg/l. To assess the monetary value of these reductions, we adopted a benefit transfer approach that corresponds to the methodology used by EPA (2003) in its analysis of concentrated animal feeding operation (CAFO) regulations. According to this approach, the average household is willing to pay roughly $2 per year (in 2002 dollars) for each 1 mg/L reduction in nitrate levels below the 10mg/l threshold. EPA arrived at this estimate based on an extensive review of the literature and the selection of results from two survey-based nonmarket valuation (contingent valuation) studies (Crutchfield, Cooper, and Hellerstein, 1997; De Zoysa, 1995), which were conducted in various parts of the United States.

To extrapolate these per-household values across our study area and roughly estimate the total potential groundwater benefits in eastern North Carolina, we assume that our sample of wells is reasonably representative of the study area.

6.4.2 Results

Using the methodology outlined above and the predictive simulation results from Chapter 5, the average annual benefit of eliminating swine farm loadings from sprayfields and air deposition would be $0.52 and $0.68, respectively, per affected household in our sample.

The aggregate annual benefit of completely removing swine farm loadings in the eastern North Carolina region from either sprayfields or air deposition would be between $116,000 and $152,000.

Table 6-13 documents the calculations used to extrapolate these results and approximate the aggregate benefits of removing these swine farm loadings. According to water usage estimates received from DENR for 2000 (NCDENR, 2003), approximately 2 million people in the groundwater study area use groundwater as their drinking water source. Assuming an average household size of 2.5 (the North Carolina average in the 2000 census), this translates to approximately 800,000 households using groundwater. Further assuming that the distribution of well depths in the WRRI database is representative of the entire population, 28 percent of the households would have wells with depths less than 50 ft. Based on the regression results (Tables 5-7 and 5-8), it is assumed that only wells less than 50 ft are significantly influenced by swine farm loadings. Therefore, the aggregate annual benefit of completely removing swine farm loadings in the eastern North Carolina region from either sprayfields or air deposition would be between $116,000 and $152,000. As expected, these results suggest that the aggregate measurable benefits based on our regression model are quite small.

6.4.3 Limitations and Uncertainties

The benefit transfer and aggregation approach described above clearly requires a number of strong assumptions. However, the purpose of this analysis was not to provide an exact measure of aggregate benefits or a modeling framework to evaluate a wide range of loadings reduction scenarios for groundwater protection. Rather, the purpose was to evaluate the potential range of benefits that might be generated by applying the results of our regression model.

Given the relatively small estimated benefits associated with a complete removal of swine farm loadings, we conclude that a more intensive analysis of the incremental benefits associated with

Table 6-13. Calculation of the Estimated Annual Groundwater Protection Benefits of Eliminating Swine Farm Loadings

Population and Households using Groundwater	
(A) Population using groundwater within study area (DENR)	2,000,000
(B) Average people per household (North Carolina 2000 census average)	2.5
(C) = (A)/(B) Households using groundwater within study area	800,000
Fraction of Wells/Households Affected	
(D) Number of wells with depth data	990
(E) Number of wells with depth <50 ft	281
(F) = (E)/(D) Percentage of wells with depth <50 ft	28%
Benefit Calculation	
(G) = (C)*(F) Households assumed to use wells with depth <50 ft	224,000
(H) Average annual per household benefit of eliminating swine farm loadings (SPRAY_DIST model specification)	$0.52
(I) Average annual per household benefit of eliminating swine farm loadings (ATM model specification)	$0.68
(J) = (G)*(H) Total annual benefits of eliminating swine farm spray field loadings	**$116,000**
(K) = (G)*(I) Total annual benefits of eliminating swine farm atmospheric deposition loadings	**$152,000**

alternative swine waste management technologies is not warranted. Consequently, we did not perform further groundwater benefits analyses, and we did not integrate the groundwater impacts and benefits into the integrated benefits assessment tool described in Chapter 7.

6.5 NONMONETIZED BENEFITS

In addition to the benefits estimated and summarized above, North Carolinians will potentially benefit from alternative hog waste management technologies in a number of other ways. Because a quantitative assessment of these other benefits is not feasible within the scope of this project, we instead describe several of them in more qualitative terms below.

Benefits from Reductions in Pathogens. First, as described in the third year report by the Project OPEN Science Team (Project OPEN Science Team, 2003), there are a number of potential pathways through which humans can be exposed to pathogens originating from swine waste. Technologies that reduce releases of disease-transmitting vectors and airborne pathogens can therefore provide health benefits to exposed populations. A quantitative assessment of these benefits would require an environmental and economic modeling approach similar to the one we have described for assessing health related benefits from reductions ammonia releases and resulting ambient PM_{Fine} levels. Unfortunately, the current state of the science is not sufficiently advanced to support an integrated assessment of changes in environmental fate and transport of pathogens and resulting reductions in the incidence of pathogen related illness. Nevertheless, these potentially avoided illnesses should not be overlooked in assessing the benefits of alternative waste management technologies.

Benefits from Improved Estuarine and Coastal Water Quality. Our analysis of water quality related benefits focuses on impacts in inland waters; however, this omits a potentially large portion of total water quality benefits. As described in Chapter 4, nutrient loadings from rivers and streams and from atmospheric deposition contribute to eutrophication and other water quality impairments in coastal and estuarine waters. According to our modeling efforts, swine waste accounts for 30 percent of the nitrogen and 11 percent of the phosphorus delivery to coastal waters from inland, free-flowing streams and rivers in the study area. Reducing nutrient loads from swine operations can therefore improve water quality in coastal and estuarine waters. The resulting benefits are most likely larger than those estimates for inland waters because coastal and estuarine waters are more actively used for recreation and for commercial fishing. However, quantification of these benefits was not feasible, primarily because it was beyond the scope of this project to model the more complex estuarine environment.

Benefits from Avoided Spills. The water quality impacts that we have modeled and described above can be thought of as primarily affecting the long-term steady state conditions of surface waters. However, as evidenced by events in 1995 (Onslow County spill) and in 1999 (Hurricane Floyd), hog waste lagoons are also subject to discrete, large-scale releases (i.e., spills). The benefits of avoiding these episodic events cannot be meaningfully captured or estimated through our surface water model; however, there is evidence that households in North Carolina would be willing to pay for waste management programs that include reductions in the probability of future waste lagoon spills (Mansfield and Smith, 2002). If certain alternative waste management technologies reduce the probability of spills, then these benefits should also not be overlooked.

"Nonuse" Benefits. A majority of the estimated environmental benefits described in this chapter can be thought of as "use" related benefits. That is, they accrue to individuals or households through direct use or contact with environmental conditions, such as through recreational activities or residential location. From a societal perspective, however, even individuals who do not directly experience environmental improvements may nonetheless value (and thus benefit from) knowing that these improvements occur (see, for example, Freeman [2003]). Although the magnitude of these "nonuse" benefits is very difficult to accurately measure, they might also be considered qualitatively in an assessment of benefits.

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7 Environmental Benefits Assessment Model: Overview and Scenario Analysis

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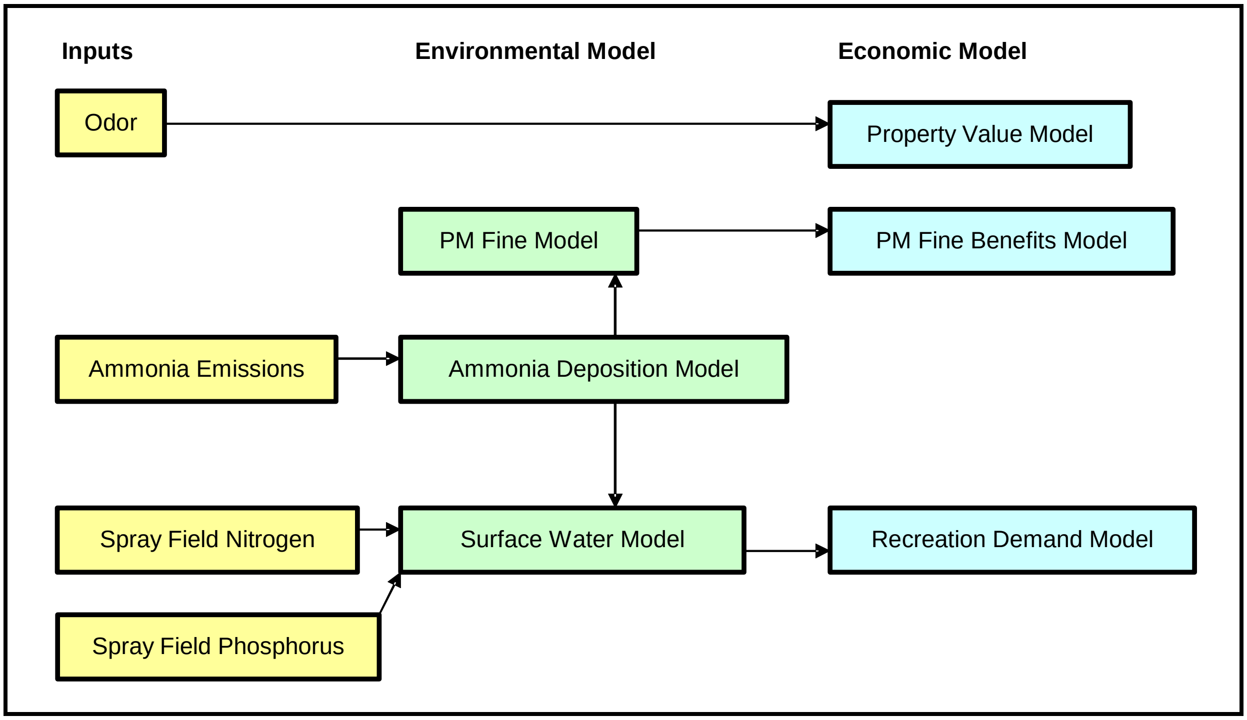
As described in previous chapters, RTI developed several environmental and economic models to assess the potential benefits of applying environmentally superior waste management technologies at hog farms in North Carolina. In addition to estimating how current waste management practices at hog farms affect environmental conditions, these models were designed to estimate how reducing pollution from hog farms could translate into environmental improvements and economic benefits.

To integrate all of the environmental and economic modeling components into a common framework, we developed a software tool, which we refer to as the Environmental Benefits Assessment Model (EBAM).

To integrate all of the environmental and economic modeling components into a common framework, we developed a software tool, which we refer to as the Environmental Benefits Assessment Model (EBAM). This chapter describes the main features of this integrated modeling system and demonstrates the model by summarizing results for selected pollutant reduction scenarios.

Figure 7-1 provides an overview of the integrated modeling system. As input, it requires specified reductions for four categories of pollutant releases from hog farms—odor, ammonia emissions, and sprayfield loadings of both nitrogen and phosphorus. Based on these specifications, the system first runs the ammonia deposition

Figure 7-1. Relationship of Load Inputs, Environmental Models, and Economic Models



model (described in Chapter 2). The results of this model feed into the PM_{Fine} and surface water quality models (Chapters 3 and 4, respectively), whose results then feed into the PM_{Fine} benefits and recreation benefits models, respectively (Chapter 6). The odor reduction benefits are estimated separately through the property value model.[1] Each of the three economic models estimates a separate component of benefits (in dollar terms) for North Carolina. These estimates can therefore be directly compared or added together.

One of the key features of this system is that it was designed to serve as a "decision tool" for evaluating alternative waste management technologies. The design of the system allows the user to first specify "technology options," which are defined strictly in terms of load reductions for each of the four pollutant release categories at hog farms. All load reductions enter into the model as a percentage load reduction compared to the baseline loads defined in the

[1]Because of the limits of the groundwater quality and valuation models and the finding of relatively small potential impacts associated with swine farm loadings (Chapter 6), these modeling components were not included in the integrated system.

previous chapters. The user is then allowed to assign technology options to different farm types and to include all North Carolina hog farms in the simulations or to restrict the simulation to a subset of operations. Based on these specifications, the model assesses environmental changes and benefits for different combinations of technology adoption. This model run mode is hereafter referred to as the "technology option" mode. Once load reduction estimates become available for the 18 candidate technologies, this system can be used to construct and evaluate benefits for alternative technology adoption scenarios.

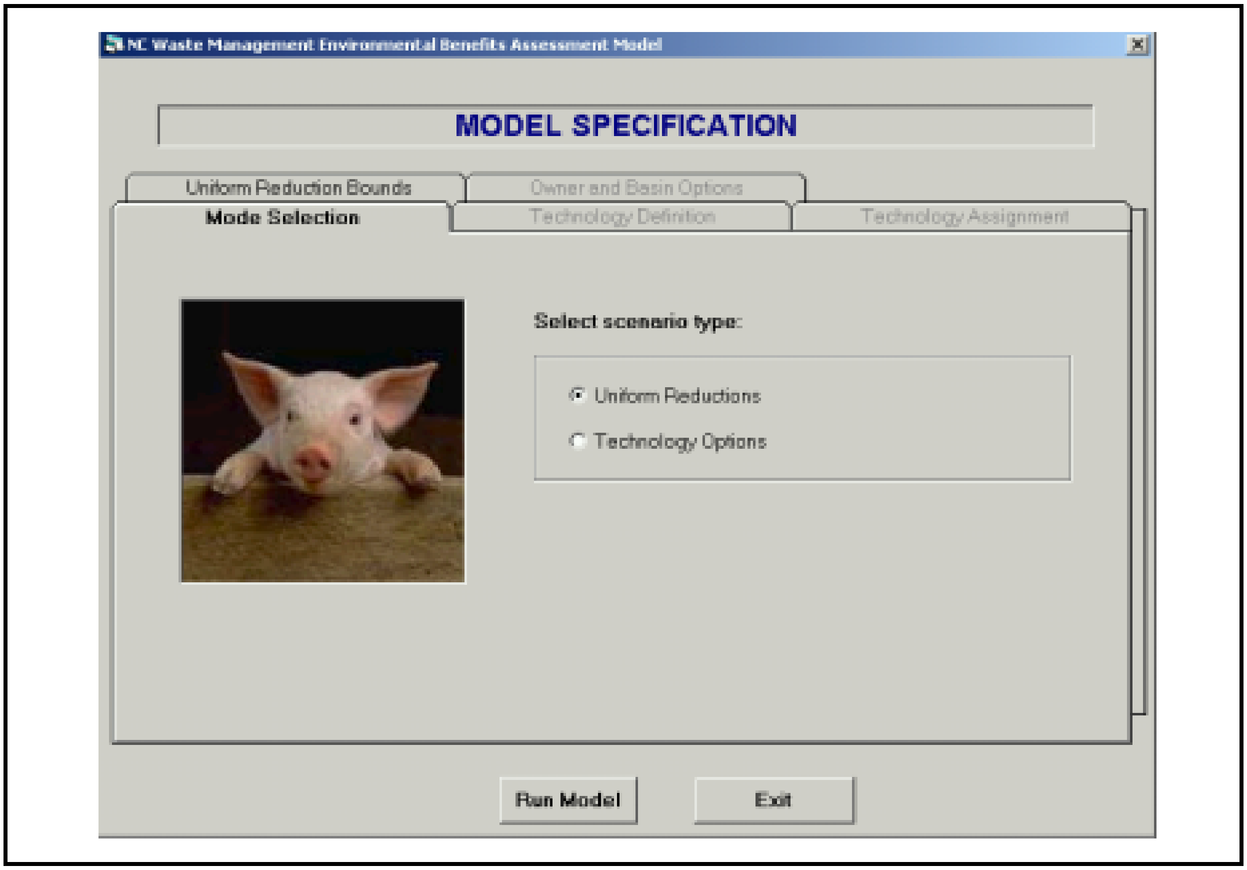
In the absence of sufficient monitoring data to specify load reductions for each of the control technologies, we designed a second run mode. This approach allows the user to conduct a large number of model runs by uniformly reducing one pollutant load input variable at a time. The user dictates the number of model runs by specifying a minimum, maximum, and incremental load reduction factor for each of the four input variables. The most significant aspect of this mode is that each load variable is reduced uniformly across all farms. This run mode is hereafter referred to as the "uniform reduction" mode. A primary advantage of this approach is that it allows one to systematically compare the effects and benefits of different combinations of reductions across the four pollutant release categories. This application of the modeling system is illustrated below.

The EBAM was developed to allow multiple users and multiple runs at one time. We created a central ORACLE database to store all supporting data tables. Individual users may specify run options, run the models, and access results through a menu-driven Visual Basic graphical user interface (GUI) that resides on the user's personal computer.

7.1 MODE SELECTION

After logging in, the user is presented with the "Model Specification" screen shown in Figure 7-2. In particular, it shows the Mode Selection tab, which allows the user to select the technology options mode or the uniform reductions mode.

Figure 7-2. Mode Selection Tab



If the user selects the technology options mode, the Uniform Reduction Bounds tab becomes disabled, and the other tabs become enabled. In this mode, the user must complete the Technology Definitions, Technology Assignment, and Owner and Basin tabs in that order.

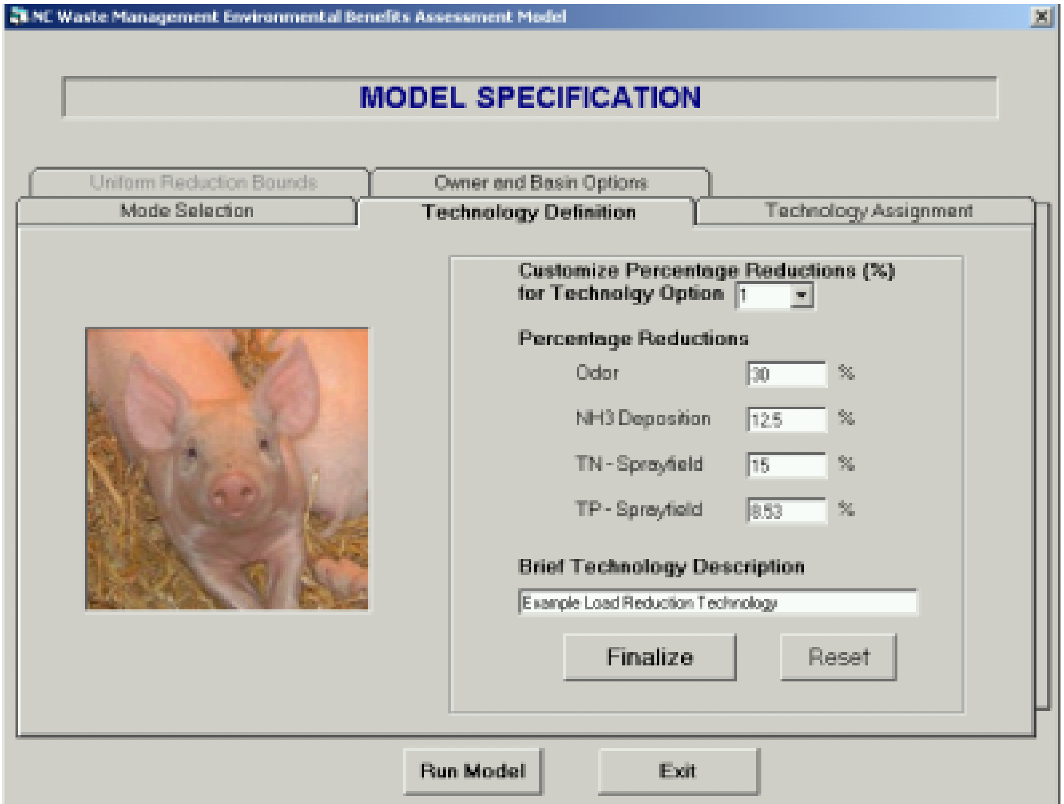
If the user selects the uniform reductions mode, the Uniform Reduction Bounds tab becomes enabled, and the other tabs become disabled because they are not relevant for this mode.

7.2 TECHNOLOGY OPTIONS MODE

In the technology options mode, the user defines a complete technology adoption scenario. The scenario is created by first defining technology types and then specifying where (i.e., at which farms) these technologies are to be applied.

The user must begin by defining customized control technology options using the Technology Definition tab (Figure 7-3). Each

Figure 7-3. Technology Definition Tab

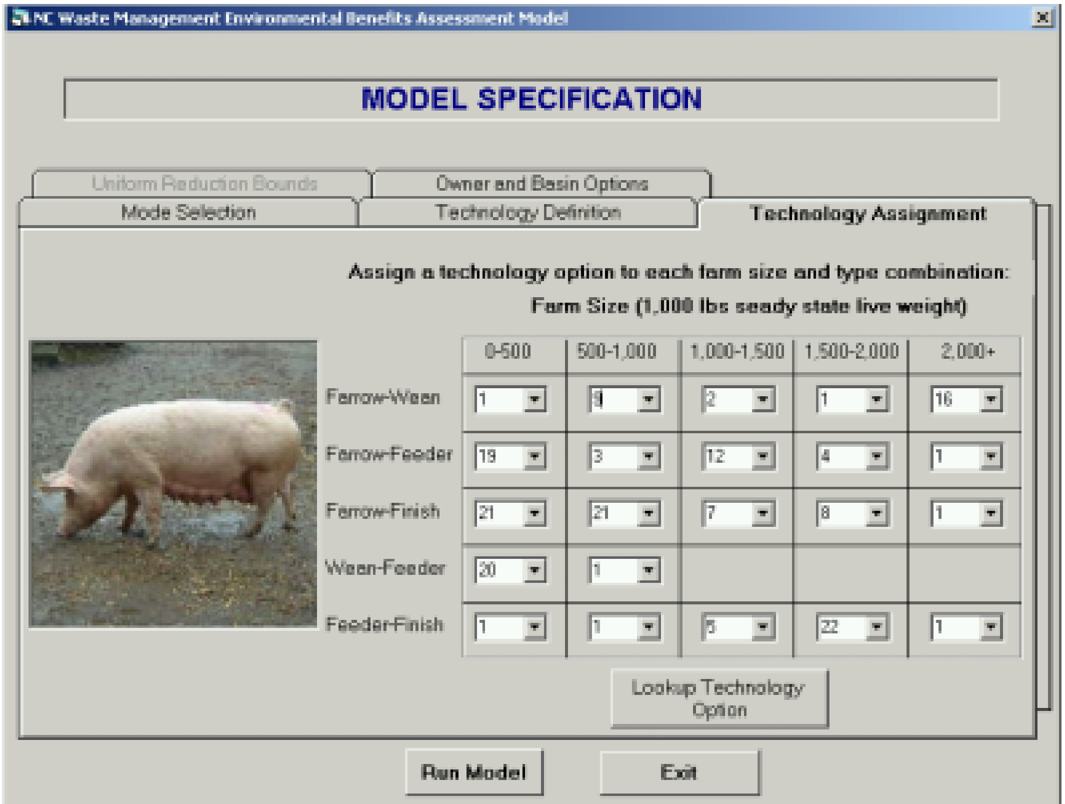


technology option is first assigned an identification number by the user (and a technology name if desired). Then, for each technology option, the user specifies reduction factors for each of the four pollutant variables. Example input for this tab is shown in Figure 7-3.

Once the technology options are defined, the user must assign the technology options to different farm categories. The Technology Assignment tab (Figure 7-4) defines 22 growth stage/farm size categories. It allows a different technology option to be assigned to each category.

To further customize the technology adoption scenario, two additional suboptions are provided in the technology options mode. The Owner and Basin tab allows the user to only apply technology options to a subset of hog farms, based on either ownership type or river basin location of the farm (Figure 7-5). The user may choose

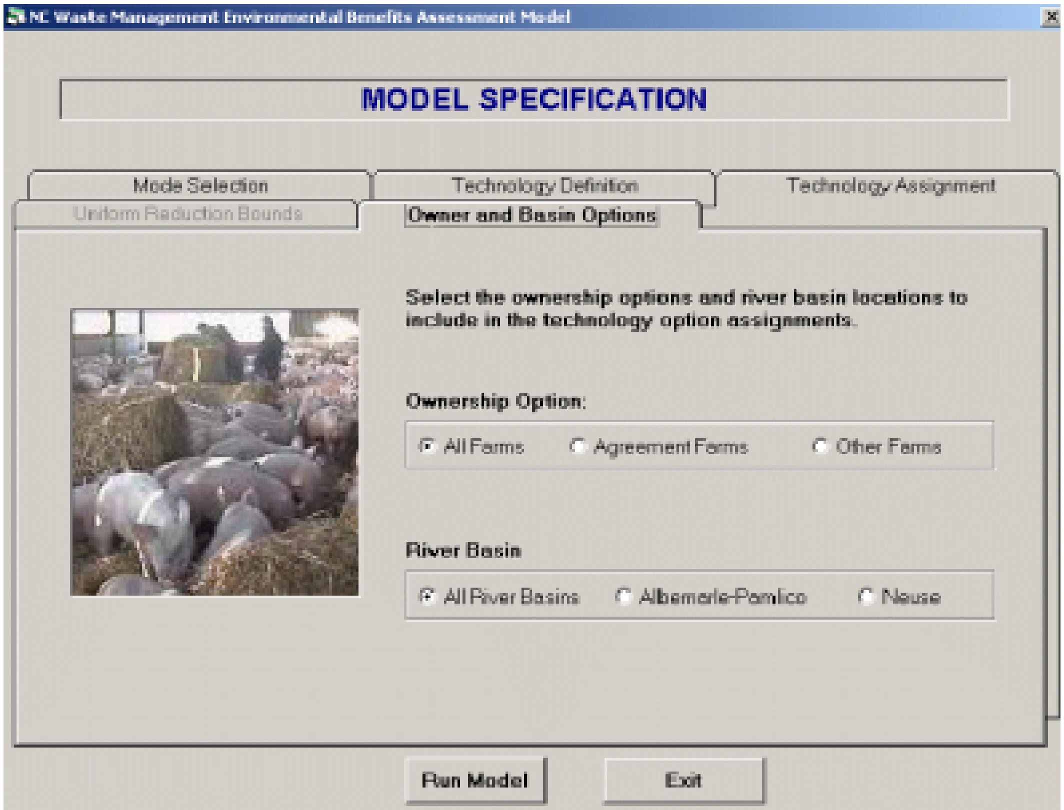
Figure 7-4. Technology Assignment Tab



to apply the technology options to all farms in the state,[2] only to the farms that are owned by companies specifically covered by the Agreement (e.g., Smithfield Foods, Premium Standard Farms), or only to farms that are owned by entities not covered by the Agreement. Similarly, the user may apply the technology options to all farms, to those located in the Neuse River Basin only, or to those located in the Albemarle-Pamlico River Basin only. If the user does select a subset of farms based on ownership or river basin, the other farms will be assigned their baseline loads, thereby reflecting the assumption of no change in waste management practices.

[2] By “all farms in the state” we refer to all 2,295 farms tracked in the North Carolina Division of Water Quality database, referred to in Chapter 1.

Figure 7-5. Owner and Basin Options Tab



Once the technology adoption scenario has been defined through these three tabs, the user can run the model and generate benefit estimates by selecting the “Run Model” button on the Technology Definition tab.

7.3 UNIFORM REDUCTIONS MODE

In the uniform reduction mode, the user defines percent reductions for each of the four pollutant release categories and applies these reductions to all hog farms. In other words, percent reductions can vary across the four pollutant categories, but they are applied uniformly to all farms. In effect, this is like applying a single technology option to all farms.

In EBAM, this mode is also set up to run multiple uniform reduction scenarios, using different combinations of reductions for the four pollutant categories. The Uniform Reduction Bounds tab allows the user to enter the minimum, maximum, and stepwise percent reductions for each pollutant load input variable, as shown in Figure 7-6. These variables determine which load variables are reduced from their baseline values and how many integrated model runs are conducted.

Figure 7-6. Uniform Reduction Bounds Tab

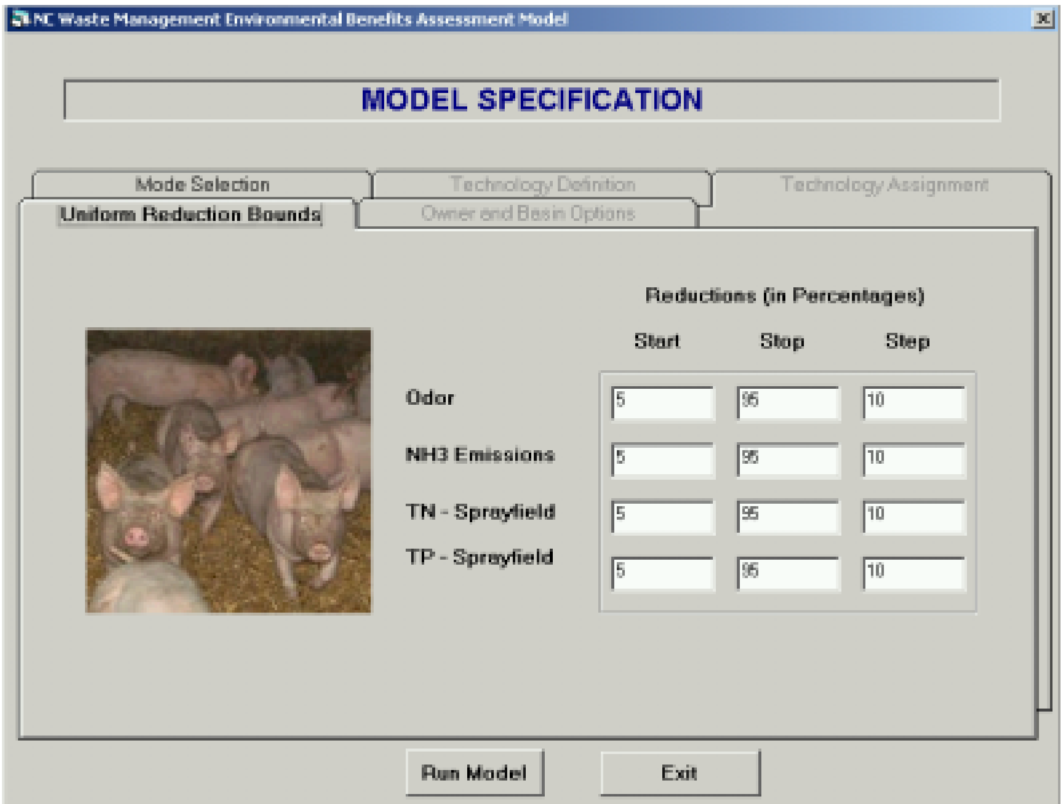


Figure 7-6 shows example inputs for this screen. In this case, each pollutant variable is reduced from its baseline value by between 5 percent and 95 percent, using 10 percent increments. For each of the four variables, this implies 10 reduction values (5, 15, 25,…95). This example results in 10,000 (10 X 10 X 10 X 10) combinations of pollutant reductions, each of which represents a separate model run.

Once this screen is finalized, the user can hit the “Run Model” command button. This will commence the automated uniform reduction model runs. Each model run takes approximately 45 minutes.

The reason for examining multiple combinations of pollutant reductions is that the estimated benefits of a percent reduction in one pollutant are not necessarily independent of the assumed percent reductions in the other pollutants. In other words, nonlinearities in the underlying models may imply, for example, that a 10 percent reduction in ammonia emissions from all farms will generate different estimates of water recreations benefits if sprayfield nitrogen loads are reduced by 10 percent than if sprayfield nitrogen loads are reduced by 90 percent. The nonlinearities and interdependencies in the modeling system (in particular through the water quality and recreation benefits model) can most easily be evaluated by running multiple combinations in the uniform reduction mode.[3]

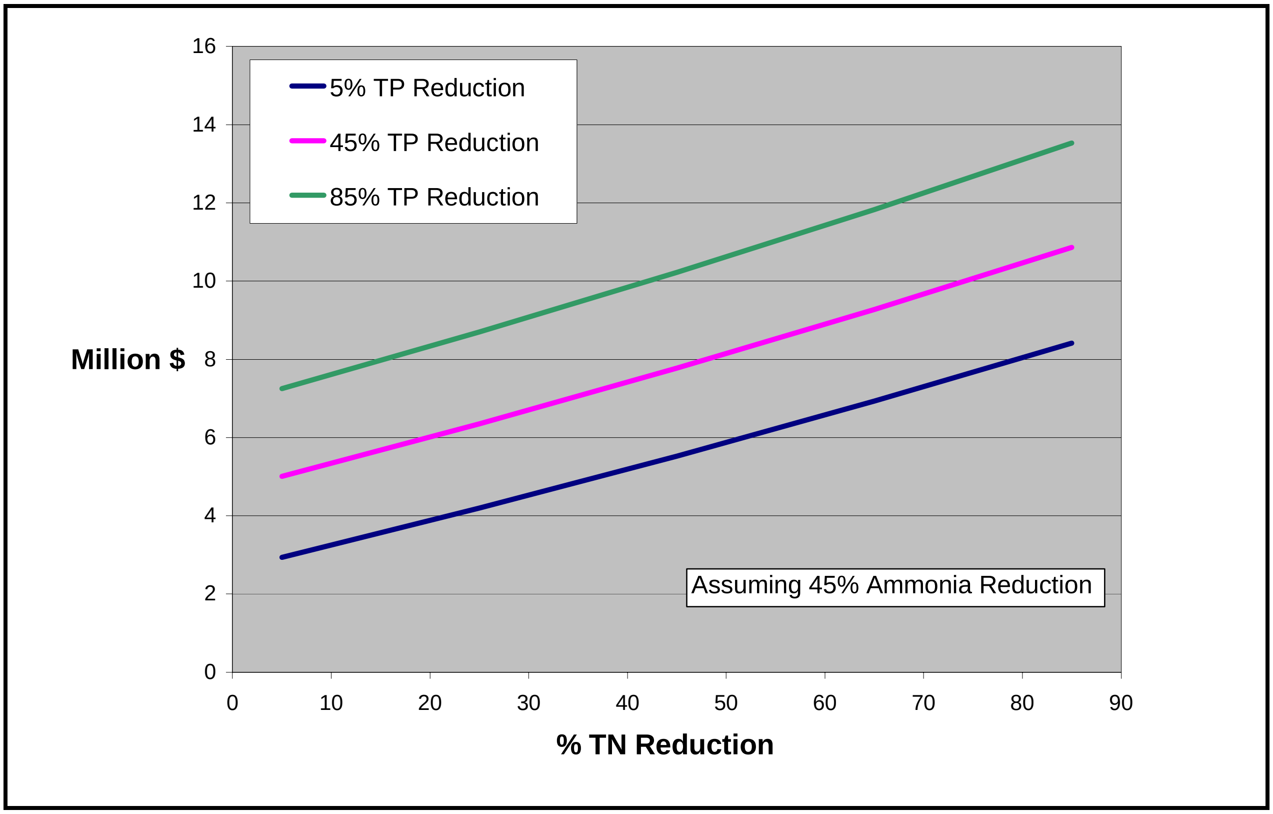
7.4 RESULTS WITH THE INTEGRATED MODELS

To illustrate how EBAM can be used and to evaluate the relative magnitude (and interdependencies) of benefit estimates for different pollutant reductions, we ran multiple scenarios through the uniform reductions mode. In particular, we varied reductions in each pollutant between 5 and 85 percent, with “steps” of 20 percent (i.e., by 5, 25, 45, 65, and 85 percent). This procedure resulted in 125 (= 5 X 5 X 5) model runs for different combinations of ammonia and sprayfield (TN and TP) loadings reductions. The five levels of odor reductions were run separately. The results are summarized below.

Figures 7-7, 7-8, and 7-9 show how different combinations of reductions in ammonia emissions and sprayfield loadings affect the aggregate estimates of recreation benefits from modeled surface water quality improvements. One clear implication of these results is that the benefit estimates increase almost linearly with respect to percent reductions in each of the pollutant categories (i.e., there is little to no curvature in the plotted relationships). A second implication is that the interdependencies between reductions in three pollutants are relatively small.

[3] As implied by Figure 7-1, the estimated odor reduction benefits are independent of the benefits from reductions in ammonia emissions and sprayfield loadings. Therefore, the odor reduction benefits can be run separately in the uniform reduction mode, thereby reducing the number of combinations to be examined.

Figure 7-7. Total Estimated Fresh Water Recreation Benefits for Different Combinations of TN, and TP Reductions (in 2002 dollars)



For ammonia emissions, each percent reduction generates an estimated $62,000 in additional benefits to freshwater recreators. For TN and TP sprayfield loadings, the incremental benefits are $73,000 and $59,000, respectively.

The graphical results displayed in Figures 7-7 through 7-9 are also summarized numerically in Table 7-1. This table reports regression results, where the dependent variable is the total water recreation benefit estimate and the independent variables are the percent reductions in the three pollutant categories. The first column of results does not include any interactions (multiplication) between the independent variables. The coefficients on the pollutant reduction variables (each of which is positive and statistically significant) measure the increment in total estimated benefits associated with each percent reduction in the pollutant. For ammonia emissions, each percent reduction generates an estimated $62,000 in additional benefits to freshwater recreators. For TN and TP sprayfield loadings, the incremental benefits are $73,000 and $59,000, respectively. The second column of results includes interactions between the three pollutant reduction variables. The model fit improves slightly, and the coefficients for

Figure 7-8. Total Estimated Freshwater Recreation Benefits for Different Combinations of TP and Ammonia Reductions (in 2002 dollars)

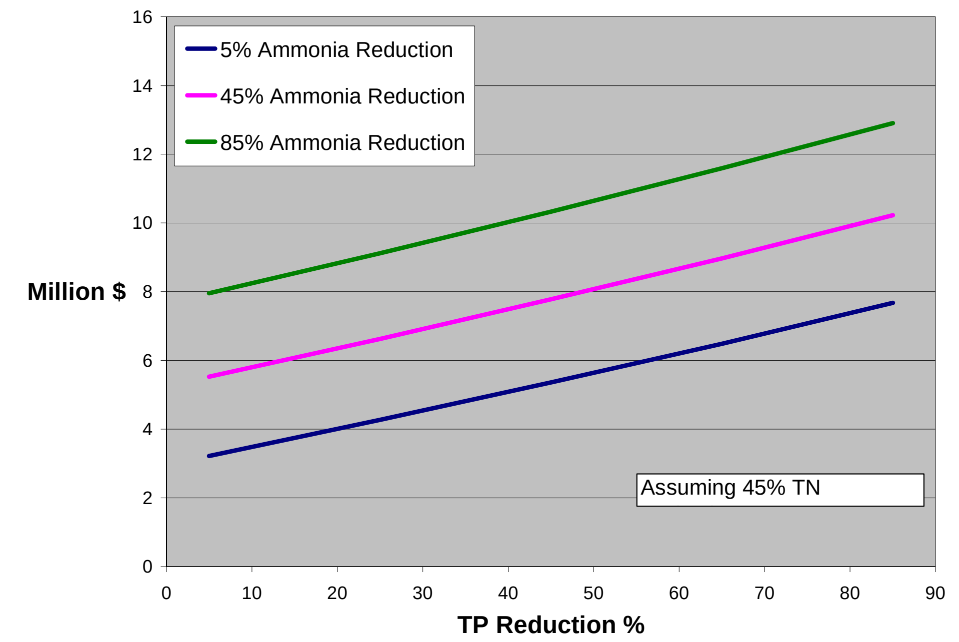


Figure 7-9. Total Estimated Freshwater Recreation Benefits for Different Combinations of Ammonia and TN Reductions (in 2002 dollars)

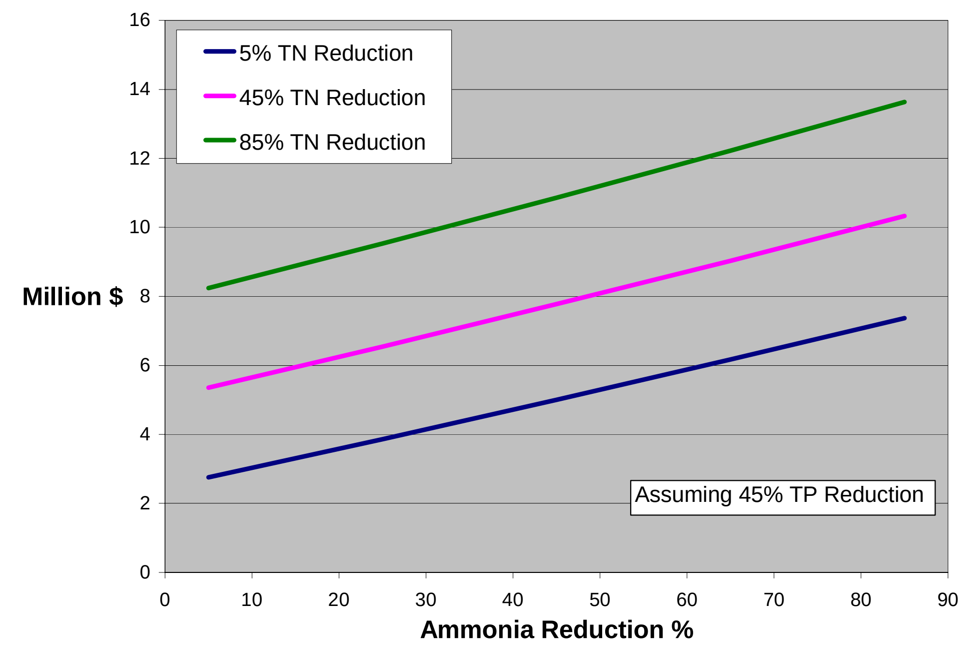


Table 7-1. Response Surface Regression for Estimated Water Recreation Benefits

Dependent Variable = Total Estimated Water Recreation Benefits		
Explanatory Variables	**Coef (t-Stat)**	**Coef (t-Stat)**
% TN reduction	73,401 (131.14)	62,331 (93.12)
% TP reduction	58,896 (105.22)	49,685 (74.23)
% ammonia reduction	62,424 (111.53)	53,474 (79.89)
% TN reduction X % TP reduction		126 (13.1)
% TN reduction X % ammonia reduction		120 (12.5)
% TP reduction X % ammonia reduction		79 (8.2)
Constant	–826,521 (–17.81)	–168,808 (–4.16)
R-squared	0.997	0.9993
N	125	125

these interaction terms are also positive and significant, but their magnitude is relatively small. These results suggest that the incremental benefits of reducing one pollutant are higher when the reductions in the other pollutants are higher, but only by a small amount. This can also be seen by the slight “fanning” out of the lines in Figures 7-7 through 7-9 as one moves to the right.

The estimated health benefits associated with reductions in PM_{Fine} levels are strictly associated with reductions in ammonia emissions. The relationship between these two variables is shown in Figure 7-10. Again, the relationship between reductions in ammonia emissions and total estimated benefits (in this case health benefits) is very close to linear. Each percent reduction in ammonia emissions is estimated to generate approximately $3.78 million in health benefits.

The estimated odor-related benefits are represented in Figure 7-11. In this case, the estimated benefits are nonlinear with respect to percent reductions odor. At lower levels (between 5 and 25 percent reductions) each percent reduction in odor generates roughly

Figure 7-10. Total Estimated PM-Related Health Benefits from Reductions in Ammonia Emissions (in 2002 dollars)

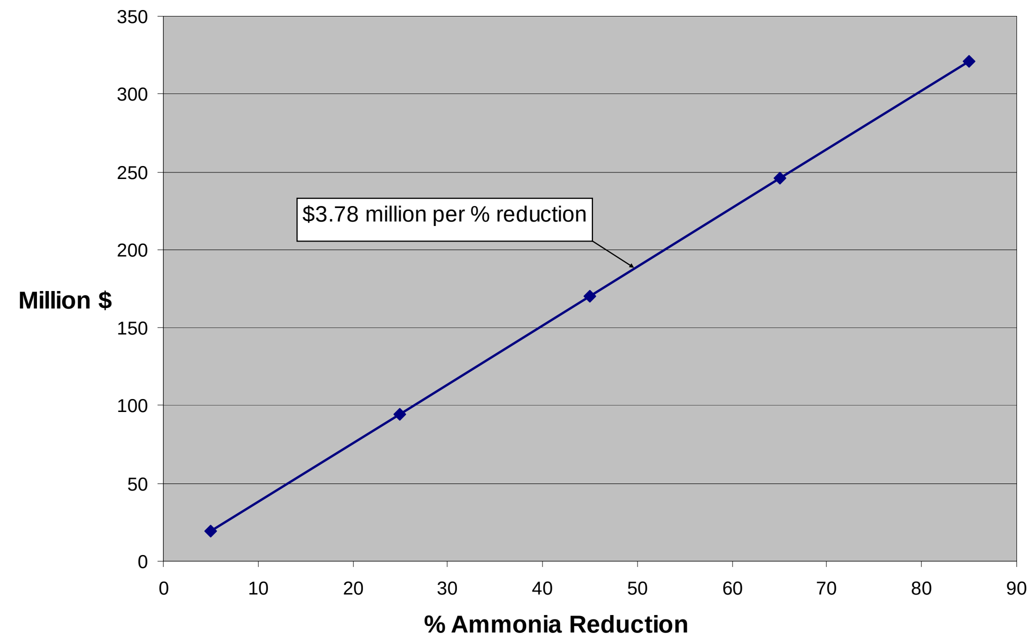


Figure 7-11. Total Estimated Property Value Benefits of Odor Reductions (in 2002 dollars)

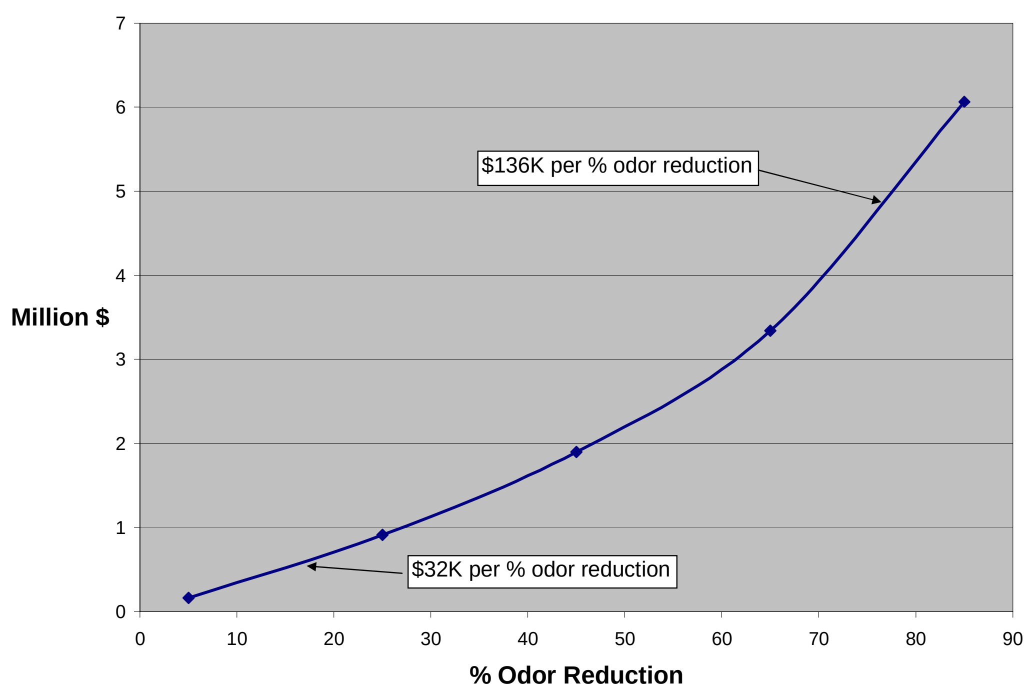


Table 7-2. Estimated Incremental Benefits per Percent Reduction in Pollutant Releases

Benefit Category	Controlled Emissions/Loadings	Additional Benefits (millions of 2002 dollars per year) per Percent Reduction
Freshwater recreation	Ammonia Total nitrogen Total phosphorus	0.06 0.07 0.06
Avoided PM-related health effects	Ammonia	3.78
Odor reduction for local residents	Odor	0.03 – 0.14

One result that stands out rather starkly from the summary estimates in Table 7-2 is that the estimated health benefits associated with ammonia reductions are distinctly (nearly two orders of magnitude) larger than the other benefit estimate categories.

$32,000 in estimated benefits to local residents. At higher levels, this estimate increases to $136,000 per percent reduction.

The incremental estimated benefits associated with percent reductions in each of the four pollutant categories are summarized in Table 7-2. It must be emphasized these are modeled estimates and are best interpreted as midpoints within a range of uncertainty.

The inherent limitations and uncertainties associated with each of these estimates are discussed in detail in previous chapters. One result that stands out rather starkly from the summary estimates in Table 7-2 is that the estimated health benefits associated with ammonia reductions are distinctly (nearly two orders of magnitude) larger than the other benefit estimate categories. This result in particular has potentially important implications when comparing the benefits of alternative technologies with different pollutant reductions.

A Surface Water Model Technical Specification

A.1 SWINE SOURCE SPATIAL CONSIDERATIONS

The model development process included associating the point location of each facility available in the facility database with adjacent land application areas. The first step in this analysis involved estimating the area of land required using North Carolina State University (NCSU) Extension Service guidelines (1980) that include ranges for swine waste for different waste treatment systems/lagoon capacity, crop, and facility/animal type. From the applicable ranges, we assumed that the operative range was equal to the lower quartile within the range (0.5 X median). This approach was chosen to subjectively account for the assumed economic incentives in minimizing costs associated with waste spray/distribution systems. The algorithm for specifically associating facilities with land cover cells was the following:

1. For facilities assumed to require up to 1 sq km, with the facility latitude and longitude locating the operation on a land cover cell with the category of agriculture or agriculture/herbaceous or agriculture/woodland: associate that cell alone with the facility.
2. For all facilities assumed to require up to 1 sq km, with the facility latitude and longitude not locating the operation on a land cover cell with the category of agriculture or agriculture/herbaceous or agriculture/woodland: associate the nearest cell with the category of agriculture or agriculture/herbaceous or agriculture/woodland alone with the facility.
3. For all facilities assumed to require >1 sq km, a GIS algorithm was written that searched adjacent cells for the agriculture or agriculture/herbaceous or agriculture/woodland cells in closest proximity, and associated found cells until the facility's area requirement was met.

4. No cell was assigned more than one facility.
5. Agriculture/herbaceous or agriculture/woodland cells were assumed to be evenly divided (50/50) between these two categories, with a total of 0.5 sq km, therefore available for land application on any one cell in these categories.

In developing this algorithm, the primary assumption was that identified land cover cells receive swine waste, although the precise area within the cell or quantity applied is not specified or required information. This assumption may result in some very localized spatial bias but should generally be unbiased for reporting at a 14-digit watershed scale. The major determinant in the spatial accuracy of locating land application areas may be associated with the degree to which manure is transported off site for application. For the purposes of this study, it was therefore assumed that any transportation outside of the 14-digit watershed in which a facility was located was negligible.

Of the total population of facilities, 87 percent were assumed to require 1 km^2 (or less) of land, with 6 percent requiring 1 to 2 km^2, and 6 percent requiring 2 to 10 km^2. The largest facility was assumed to require 16 km^2.

Data from the North Carolina Division of Water Quality were used to locate facilities and determine facility type and size.

The database was enhanced by assigning geographic coordinates (latitude/longitude) to swine facilities in eastern North Carolina that were lacking georeferencing in the NCDWQ database. First, we prioritized the counties with missing coordinates for farms (Table A-1). The counties were prioritized in descending order by hog population (total number of head for the sites missing coordinates, by county) and by county ranking from the U.S. Department of Agriculture's (USDA's) 1997 ranking of the six largest counties (in pig population) in North Carolina.

This process identified 168 facilities to be georeferenced, with the highest priority counties being Sampson, Duplin, Bladen, Columbus, Jones, and Edgecomb. All 84 facilities in those six priority counties were geolocated, with the "level of certainty" for each facility also documented. In addition, following the prioritization guidelines described above, all facilities in Wayne, Greene, Nash, Pender, Johnston, Pitt, Washington, Lenoir, and

Table A-1. Swine Facility Georeferencing Overview

County	Number of Sites Originally without Latitude/Longitude	Design Capacity (pigs)	1997 USDA Inventory (Number of Pigs) for Six Largest North Carolina Counties	Percentage of Inventory without Latitude/Longitude
Sampson	39	164,027	1,776,000	9.24
Duplin	25	121,348	2,034,000	5.97
Wayne	12	48,226		
Bladen	7	42,904	759,000	5.65
Edgecomb	5	29,670	169,000	17.56
Greene	8	29,600		
Pender	10	27,932		
Columbus	6	27,446	258,000	11
Lenoir	5	26,404		
Jones	2	23,643	253,000	9.35
Hoke	2	22,944		
Nash	8	22,867		
Pitt	4	18,026		
Johnston	4	17,536		
Robeson	3	17,440		
Richmond	4	15,100		
Washington	4	11,100		
Harnett	2	6,450		
Hertford	3	5,664		
Brunswick	2	4,322		
Moore	3	4,256		
Cumberland	1	2,400		
Beaufort	1	2,245		
Franklin	1	2,205		
Chatham	1	2,000		
Martin	1	1,900		
Onslow	1	1,776		
Northampton	2	1,700		
Cabarrus	1	1,200		
Halifax	1	280		
Total	**168**	**702,611**	**More than 5,249,000**	**Less than 13.4%**

Robeson counties were carefully located (an additional 58 facilities), and, because of their importance to this study, three farms owned by Smithfield Carroll's Farms (in Hertford and Northampton counties) were also located and cross-checked. All farms in the remaining counties were located as described below (in Item No. 4).

RTI determined geographic coordinates (latitude and longitude in decimal degrees) for these facilities using the following methods:

1. First, using the table of farms, RTI attempted to use ArcGIS's "geocoding addresses" tool (in conjunction with ArcGIS's StreetMap USA local address geocoding service extension) to obtain geographic coordinates for these farms based on their mailing addresses. The parameters used for selection were spelling sensitivity of 80 percent, a minimum "candidate" score of 10 (out of 100), and a minimum match score of 60. RTI recognizes that the descriptions of the *locations* of the farms are more accurate than a geocoded mailing address, with respect to the actual location of the swine facility; however, in some cases (especially for the lower-priority counties), the geocoded addresses without a spatial cross-check were necessary.
2. After the addresses were geocoded, the descriptions given for the location of each farm in the six priority counties were all checked against 7.5 minute USGS Quad maps (available through ©LandNet Corp.'s *LandViewer*™, using the "interactive maps" option). North Carolina state road numbers were checked and often located using North Carolina county maps, while street names were located and checked using a combination of the *North Carolina Atlas & Gazetteer* (1993) and MapQuest.com's *MapQuest*© (1996-2002). MapQuest's site (http://www.mapquest.com) also gives the viewer the opportunity to cross-check locations with aerial photography (of indeterminate age) at 1 meter or better resolution for most of the study area. This aerial photography service is made possible by MapQuest's partnership with *GlobeXplorer*, Inc. © 2002.
3. The farms in Wayne, Greene, Nash, Pender, Johnston, Pitt, Washington, Lenoir, and Robeson counties were similarly located. In late June 2002, Gary Saunders of the North Carolina Department of Environment and Natural Resources (DENR's) Division of Air Quality provided RTI with an updated spreadsheet, in which he had provided latitudes and longitudes for many of the farms originally missing coordinates. For the counties mentioned above, Mr. Saunders' coordinates were checked against the on-line 7.5 minute quad maps (http://www.landnetusa.com), and adjusted if necessary. Adjustments were made to Mr. Saunders' coordinates if the coordinate recorded did not match the locational descriptions contained in the original

DWQ database of 209 facilities missing latitudes and longitudes. (RTI also reviewed data provided by Dr. Bailey Norwood of North Carolina State University's Agricultural and Resource Economics [ARE] unit but did not find additional information beyond the above sources to assist with locating missing farms.)

4. The facilities in the remaining counties (Harnett, Hertford, Brunswick, Moore, Cumberland, Beaufort, Franklin, Chatham, Martin, Onslow, Northampton, Cabarrus, and Halifax) were provided with coordinates of varying accuracy, none of which were checked closely against the Internet sources mentioned above. If Mr. Saunders' spreadsheet provided revised coordinates for farms in those counties, his coordinates were almost always used without a cross-check. If no coordinates were provided by Mr. Saunders' table, but ArcGIS had provided a tentative location based on mailing address, that set of geocoded coordinates was used. In all other instances, the Internet resources described above (primarily *LandViewer*™, because it provided latitude and longitude in decimal degrees, based on 7.5 minute USGS quadrangle maps) were used to derive the best estimate of geographic location, given the time and budget constraints of this project.

A.2 NITROGEN AND PHOSPHORUS EDGE-OF-FIELD DELIVERY FROM SWINE OPERATIONS

Nitrogen and phosphorus edge-of-field delivery was calculated using methods described in Schwabe (1996) and Norwood (2003a) and county- and crop-based data provided by Norwood (2003b) and the Conservation Technology Information Center (2002). Nitrogen delivery was estimated as

$$\text{Nitrogen EOFDC} = 1/(1 + \exp(2.33137 - P^{0.984645}) \qquad \text{(A.1)}$$

where P is a county-specific permeability index value.

The phosphorus delivery coefficient was calculated based on the universal soil loss equation (USLE):

$$\text{Phosphorus EOFDC} = 1/(1 + \exp(2.918179191 - \text{USLE}^{0.623066904})) \qquad \text{(A.2)}$$

For phosphorus delivery, a USLE value was calculated for each county and crop. The USLE value equals

$$\text{USLE} = R * LS * K * C \qquad \text{(A.3)}$$

where

R = rainfall factor,

LS = length/slope,

K = erodibility factor, and

C = cropping factor.

Values for C were based on a crop rotation of corn, soybeans, wheat, and weighting tillage data by the cropland in the county in each type of tillage. The following C values were assumed:

	Conventional Tillage	Conservation Tillage
Corn	0.25	0.15
Soybeans	0.30	0.05
Wheat	0.04	0.03
Bermuda grass	0.007	

In calculating the C factor, it was assumed that 60 percent of the swine waste is applied to row crops and 40 percent to Bermuda grass (Norwood, 2003a).

A.3 LAND USE/COVER

Land use/land cover (LU/LC) data served several purposes in the modeling process. First, the LU/LC data were used to identify/estimate lands on which swine waste is being applied (described below) and to link swine sources to the surface water network. For this use, Advanced Very High Resolution Radiometric (AVHRR) data were used as a 1 km^2 grid.[1] Second, the National Land Cover Dataset was used to estimate land use/cover by 14-digit watershed for nonswine modeling input.

A.4 SURFACE WATER HYDROGRAPHY

A hydrographic/hydrologic database was created based on the River Reach File Version 3 (RF3). With extensive and unique attributes and code embedded in the database, RF3 provided the routing and modeling "engine" for linking sources with surface waters and delivering pollutants through surface water networks. The RF3 database for the study area includes approximately 25,000

[1]This data source was chosen because of historical success in integrating these data with the Reach File.

networked reaches, terminating at the Tar-Pamlico, Neuse, White Oak, New, and Cape Fear estuaries. Important surface water attributes (e.g., channel characteristics, streamflow statistics) in addition to extensive study area-specific routing features have been incorporated in the RF3 database to support modeling efforts.

RF3 is a hydrologic database of the surface waters of the continental United States. The RF3 network is based on 1:100,000 scale digital line graph hydrographic data and has been designed expressly to establish hydrologic ordering, to perform hydrologic navigation for modeling applications. Recent projects sponsored by the U.S. Environmental Protection Agency (EPA) and the United States Geological Survey (USGS) have resulted in improved ability of the RF3 stream network data to support modeling efforts (RTI, 2001a,b).[2] The RF3 network used for the model includes almost 25,000 reaches accounting for almost 24,000 stream miles. All of the hydrologic, hydraulic, and water quality calculations completed for this study were done for each of these reaches. Model results were reported at the outlet of each 14-digit hydrologic unit (HU) (as defined by federal and state agencies) and also compiled for major river basins. The surface water model network in the study area included 565 HUs.

A.5 FLOW AND HYDRAULIC CHARACTERISTICS

Streamflow was modeled using the methodology described by RTI (2001a). Streamflow statistics integrated into the Reach File database were used to calculate yield (flow per unit area). A land cover grid was linked to RF3 reaches, allowing for drainage area summation by reach. The combination of reach-specific drainage area and yield allowed for the development of a flow routing routine. Annual average flows were used for the analysis described herein, although the analysis could be readily adapted for other flow conditions (e.g., summer low-flow).

The relationship between stream flow and channel width used was (Keup, 1985)

[2] The National Hydrography Database (NHD), released by the USGS and EPA in 2001, was built with RF3 architecture. Therefore, employing RF3 as the modeling stream network facilitates integration with NHD if warranted for future studies.

$$W = 5.27 * Q\ 0.459 \tag{A.4}$$

where

W = channel width (ft) and

Q = discharge (stream flow in cubic feet per second [cfs]).

Channel depths were calculated based on the classic Manning's N formulation for channel resistance analysis. Assuming a rectangular channel cross-section, the following formula was used to calculate stream depth:

$$y0 = 0.79(Q * n/(W * (S0)0.5\ )0.6 \tag{A.5}$$

y0 = channel depth (ft),

Q = discharge (stream flow in cfs),

n = Manning's N roughness coefficient,

W = channel width (ft) calculated above, and

S0 = channel slope (ft/ft)

Manning's N was calculated, based on sinuosity, as

$$\text{Manning's } N = 0.0016 + 0.0234 * S, \tag{A.6}$$

with a lower limit of Manning's N = 0.025 and an upper limit of Manning's N = 0.040 (Henderson, 1966).

Sinuosity estimates were calculated as

$$S = SEGL/DIST$$

S = sinuosity measure (unitless),

SL = segment length of the reach (mi), and

DIST = straight-line distance between upstream and downstream nodes of the reach (mi).

Stream velocity for each reach, therefore, was calculated as

$$V = Q/(W*y0) \tag{A.7}$$

V = velocity (ft/sec),

Q = discharge (streamflow in cubic feet per second, cu ft/sec),

y0 = channel depth (ft) calculated above, and

W = channel width (ft) calculated above.

Time-of-travel along a stream reach was calculated as

$$Tt = SL/(V) \qquad (A.8)$$

Tt = time-of-travel along stream reach.

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