Nitrogen Management Project Protocol: A Background Paper on Quantification of N_2O Emission Reductions

Steven De Gryze, PhD

John Kimble, PhD

Johan Six, PhD

Bill Salas, PhD

Please provide comments to steven.degryze@terraglobalcapital.com

December 2011

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Highlighted Sections are New from Draft Published July 2011

Acronyms

AAPFCO	Association of American Plant Food Control Officials
AMA	Agricultural Management Assistance
ARMS	Agricultural Resource Management Service
BMPs	Best Management Practices (practices used to achieve TMDLs)
CDFA	California Department of Food and Agriculture
CSP	Conservation Stewardship Program
CTA	Conservation Technical Assistance
CWA	Clean Water Act
CZMA	Coastal Zone Management Act
EPA	Environmental Protection Agency
EQIP	Environmental Quality Incentive Program
ERS	Economic Research Service
NASS	National Agricultural Statistics Service
NIFA	National Institute of Food and Agriculture
NRCS	Natural Resource Conservation Service
NSP	Nonpoint Source Pollution
PSNT	Pre-Sidedress Nitrogen Test
SDMA	Safe Drinking Water Act
TFI	The Fertilizer Institute
TMDL	Total Maximum Daily Load
USDA	United States Department of Agriculture
USGS	United States Geological Service

Definitions

Direct N_2O emissions	N_2O emitted from the soil at the project site, as measured using static flux chambers.
Global Warming Potential	A measure that expresses how much heat a greenhouse gas traps in the atmosphere relative to CO_2. The global warming potential takes into account the absorption of infrared radiation by a given greenhouse gas, the spectral location of its absorbing wavelengths and the atmospheric lifetime of the species.
Indirect N_2O emissions	N_2O emitted beyond the project site, derived from N that is lost from the project site through leaching and volatilization, and converted to N_2O elsewhere.
Mineral N	Soil N in the form of NH_4^+, NO_3^- or NO_2^-. These forms are readily available for plant uptake and drive multiple microbial N transformations including nitrification and denitrification.
Organic fertilizer N	N amendments in organic form, derived from biological products such as animal manure, wastewater sludge and compost.
Residual N	NO_3^- left in the soil profile after harvest. After discounting for leaching and gaseous N loss, this NO_3^- is can be used by the next crop.
Residue N	N contained in crop residues. In subsequent cropping seasons, crops can take up N derived from crop residues upon mineralization.
Synthetic fertilizer N	Synthetically produced N, including anhydrous ammonia, urea, ammonium nitrate as well as multi–nutrient fertilizers (e.g., N–P–K fertilizers) or 'enhanced–efficiency' N fertilizers (e.g., slow release, controlled release and stabilized N fertilizers).

1 Introduction and Practices

1.1 Background

Nitrogen (N) is an essential nutrient for plants. Healthy crops contain up to 5% of this nutrient. During much of history, N was supplied to crops primarily in organic form such as through manure application and N-fixing legumes. However, during the latter part of the 19th century, inorganic N replaced organic N as the main source of this nutrient. Today, inorganic N is essential to world food production. Applying the proper rate of nitrogen at the right time during the year are major management decisions agricultural producers have to make. Using too little N may results in lower yields, poorer crop quality, and, hence, reduced profits. When too much N is applied, yields and quality are generally not compromised, but profit may be reduced and negative environmental effects will occur including surface and groundwater contamination with nitrates due to N leaching and contribution to global environmental change due to nitrous oxide (N_2O) emissions. Nitrous oxide is a powerful greenhouse gas (GHG) with about 310 times the global warming potential of carbon dioxide. Nitrous oxide emissions from agricultural land are generally related to the application of inorganic and organic N fertilizer, or legume-derived N. Any factor or action that affects the availability of mineral N in the soil may impact N_2O emissions.

The Climate Action Reserve (Reserve) is developing a Nitrogen Management Project Protocol (NMPP) to provide a standardized approach for quantifying and monitoring GHG reductions from projects that reduce nitrous oxide (N_2O) emissions as a result of changes in N management on U.S. croplands. Examples of management strategies that may reduce N_2O-emissions, and are therefore project activity candidates for the NMPP, include reducing the total N fertilizer application rate, synchronizing the timing of fertilizer application with the timing of N uptake requirements of crops, the use of slow-release fertilizers, and using nitrification inhibitors. In general, any project activity that maximizes N use efficiency is a sound candidate for the NMPP.

The purpose of this background paper is to describe background research into issues to be addressed within the Reserve's nitrogen management protocol. The paper first outlines individual practices that can be considered for inclusion in the protocol (section 1), investigates additionality (section 2), provides an overview of all sources, sinks, and reservoirs related to nitrous oxide (section 3), and evaluates different options for quantification of emissions reductions from nitrogen management activities (sections 4 and 5).

This paper does not aim at providing full details on each of the topics covered, nor does it aim at providing definitive conclusions on how the GHG accounting mechanics should work. Rather, the paper aims at providing a comprehensive overview of all the approaches and options available for each of the potential challenges that may be encountered during future protocol development. Where appropriate, some recommendations are provided. The paper is meant to support the Reserve and NMPP multi-stakeholder workgroup in selecting options and developing the broad architecture of the protocol.

Additional details on each of the options and topics discussed in the paper can be found in the extensive literature list.

1.2 Overview of Individual Practices that May Reduce N_2O Emissions

As will be explained in the following paragraph, the key scientific principle of minimizing N_2O emissions is to match nutrient supply as exactly as possible with plant nutrient uptake so that the presence of excess organic N in the soil can be avoided. One of the complicating issues of creating an effective NMPP protocol is the large number of potential practices that are able to match nutrient supply with nutrient uptake and have the potential to reduce N_2O emissions. It is the task of the multi-stakeholder NMPP workgroup to define which practices and combinations of practices should be given priority. This section aims at supporting this prioritization by providing an overview of the individual practices that can lead to N_2O emission reductions. A more comprehensive overview of the potential to reduce N_2O emissions by individual practices is available in the T-AGG literature synthesis (Eagle et al., 2011).

Nitrous oxide is a by-product of two microbial processes occurring in the soil: denitrification and nitrification. A necessary but insufficient condition for nitrous oxide emissions to occur is the presence of excess mineral N in the soil. Additionally, nitrous oxide emissions are positively correlated with low pH, higher temperatures, high water-filled pore space, soil compaction, and available C substrate (Chantigny et al. 2010; Farahbakhshazad et al. 2008; Venterea and Rolston 2000). As a consequence, any management practice that reduces the presence of excess mineral N in the soil is a good candidate N_2O emission reduction strategy. Therefore, nitrogen management practices to reduce N_2O emissions must focus on improving the N use efficiency (i.e., less N applied for the same crop productivity). Table 1 contains a comprehensive list of individual activities that may be considered for inclusion in the nitrogen management protocol. Note that these individual activities would most likely be combined in practice (see next section).

A general framework on agricultural best management practices (BMPs) regarding nitrogen management was developed by the International Plant Nutrition Institute (IPNI), and is known as the “4R” nutrient stewardship concept. The 4R framework states that nitrogen management should be guided by getting the following four aspects right: source – ensuring a healthy balance of nutrients, rate – by assessing nutrient supply and demand, time – through incorporation of crop N demand, and place – recognizing root-soil dynamics. An excellent series of background papers reviewing the latest scientific literature on each of these four “rights” is available on IPNI’s website.

Given the importance of maintaining crop production to meet the world’s food demand, it is important to focus on project activities that do not lead to any yield losses. Hence, the focus of project activities should be on maximizing the efficiency of N use and minimizing losses of N so that N addition can be reduced while maintaining crop productivity. Reducing fertilizer N application rates, however, can be an effective strategy when combined with optimized

nitrogen sources, timing, and place, or by including mixed[1] cover crops in the rotation. Reducing N rates is also feasible when used within the context of precision agriculture, where nitrogen application rates are continuously adjusted according to the soil fertility of a particular location within a field. In any case, only project activities that maintain or increase productivity are sustainable in the end.

Table 1. Overview of the potential nitrogen management practices that are considered in the NMPP. Bold-face text represents an option category. Bullets represent sub-categories.

(A separate in-depth analysis and evaluation of management practices is available in the *SAC Meeting Report* summarizing recommendations of the NMPP's Science Advisory Committee.)

Individual nitrogen management practice
Reducing annual fertilizer application rate by using a Pre-Sidedress Nitrogen Test (PSNT) or other crop/soil based indicators to determine fertilizer application rate *Rationale: A significant amount of residual nitrogen from the previous growing season can be present in the soil before planting. This residual nitrogen can partially meet the crop N requirements, decreasing the amount of N that must be supplied. A PSNT determines the amount of residual nitrogen and can optimize the N application rate to avoid excess N and subsequent N_2O emissions.*
Replacing inorganic N application partially or completely by organic amendments • Partially replacing inorganic N application by manure • Partially replacing inorganic N application by compost • Complete transition to Organic Cropping Systems *Rationale: where organic amendments such as manure replace inorganic N, N_2O emissions tend to be lower. This decrease in N_2O emissions is caused by the lower availability of the N stored in organic amendments to crops compared to inorganic N, which reduces the risk for denitrification and N_2O production. However, it is not always the case that N_2O emissions are lower in systems with organic amendments. This effect is very much dependent on soil texture (Chantigny et al., 2010), climate, and the quality of the organic amendment, which is usually represented by the C-to-N ratio of the amendment.*

[1] "Mixed" refers to the combination of nitrogen fixing and non-nitrogen fixing cover crop species. Mixed cover crop systems have been shown to absorb residual N from the previous season while providing N in the subsequent cropping season (Tonitto et al 2006).

Optimizing the timing of N fertilizer application

- Apply nutrients no more than 30 days prior to planned planting date
- Apply 50% or more of the total nitrogen needs after crop emergence
- Split application technology and management

Rationale: Crop N demand changes drastically during the growing season: N demand is small right after planting, increases rapidly during vegetative growth, and drops sharply as the crop senesces or nears maturity. As a consequence, synchronizing timing of N application with plant N requirements reduces the potential for excess N to occur and reduces N_2O emissions. One strategy is to change N application from the fall, as is common practice for up to 30% of U.S. corn production (Paustian et al. 2004), to the spring instead (Hao et al. 2001; Hultgreen and Leduc 2003). Timing fertilizer application to crop uptake is one of the most effective ways to reduce N loss as N_2O, but will require more labor from the producer.

Including mixed cover crops in a rotation

- Plant a cover crop to absorb residual nitrogen post-harvest
- Use of legume cover crops as a nitrogen source

Rationale: When cover crops are used in fertilizer-based grain systems, it is common practice to use a non-legume plant such as cereal rye. However, mixed (legume and non-legume grass) cover crops can be used in legume-fertilized rotations to improve N management. The non-legume will scavenge any residual nitrogen and immobilize this nitrogen during the off-season, effectively preventing N_2O emissions from residual nitrogen. If N availability is low, the legume will fix atmospheric N into forms that can become available for subsequent crops, further reducing the need for supplying inorganic N and potential N_2O. When plowed in right before planting, the nitrogen absorbed by cover crops is slowly released upon natural decomposition of the cover crop material. In addition, cover crops will sequester atmospheric carbon by increasing soil organic carbon levels. However, in some cases increases in N_2O emissions have been observed when using cover crops, probably due to the higher presence of readily available carbon sources. Therefore, the timing between cover crop incorporation and planting should be as minimal as possible.

Note that the use of mixed cover crops was supported by some members of the workgroup. However, the Science Advisory Committee did not include mixed cover crops in the list of currently approved practices to reduce N_2O emissions.

Placement of fertilizer

- Injected near seeds during sowing
- Applied in sub-surface drip irrigation (fertigation)

Rationale: The closer fertilizer is placed near the zone of active root uptake, the greater the probability that the N is effectively taken up by the plant, resulting in less N_2O emissions. Additionally, due to local changes in soil quality within one field, the efficiency of N fertilizer can be optimized by exactly applying the amount of N fertilizer that is optimal for a specific location. This requires a GPS-based fertilizer application and is often referred to as precision agriculture.

Changing fertilizer composition • Change chemical composition (anhydrous ammonia to urea) • Change to controlled-release nitrogen fertilizer *Rationale: for denitrification or nitrification to take place, nitrogen must be in a specific chemical form (i.e., nitrate vs. ammonia, respectively). If nitrogen is applied in a chemical form that is more distant from these reagents, chances of denitrification or nitrification to occur are smaller. The three main nitrogen sources used in the U.S. are anhydrous ammonia, urea, and ammonium nitrate, While in a broad literature review, no effects of chemical composition on N_2O emissions were found (Stehfest and Bouwman, 2006), many paired studies demonstrated lower emissions using urea than with anhydrous ammonia or ammonium nitrate. Many confounding factors exist, however.* *Controlled-release fertilizers have specific coatings designed to provide a slow but steady release of mineral N so that N use efficiency is improved and N_2O emissions are reduced. Some limited studies indicate that there is some potential for slow-release fertilizer to improve N use efficiency and reduce N_2O emissions (Halvorson et al. 2010; Hyatt et al. 2010). However, it should be noted that the cost of production and transportation is higher than conventional N fertilizers.*
Use of nitrification and urease inhibitors *Rationale: nitrification and urease inhibitors are chemical compounds that slow the biological transformation of nitrate and urea, respectively, leading to greater N use efficiencies and smaller N_2O emissions. Nitrification and urease inhibitors can be very effective in reducing N_2O emissions. When used inappropriately, however, nitrification and urease inhibitors may just shift N_2O emissions to later during the growing season, causing no net decrease in N_2O emissions. In addition, they increase production costs significantly.*
Adding deep rooting plants into crop rotations (e.g. alfalfa or other hay plants) *Rationale: Deep rooting plants can scavenge residual nitrogen and once taken up by roots, redistribute nitrogen deeper into the soil profile, decreasing the potential for a local excess of inorganic nitrogen.*

1.3 Combining Individual Practices

N application rate has been identified as the most reliable predictor of N_2O emissions (Stehfest and Bouwman 2005, Millar et al. 2010). However, a straightforward reduction in N rate without changing N source, timing and/or placement might not always be possible. For example, Snyder et al. (2011) pointed out that corn producers in the North Central Region on average already apply equal or smaller amounts of N than the recommended rate. This was confirmed in a recent N use report by USDA (2011), which stated that 65% of corn producers use best management practices with respect to N rate. Nevertheless, N rate and/or N_2O emissions might be further reduced by improving other N management practices (i.e., N source, timing, and placement). Under certain circumstances, increasing rotational complexity such as including cover crops or switching from a continuous corn to a corn-soybean rotation could also allow producers to reduce N application requirements and therefore N_2O emissions. It should be noted, however, that increasing rotational complexity can achieve additional N_2O emission reductions other than reductions that would be predicted based on the reductions in N application rates alone. In general, a comprehensive and integrated approach to nitrogen management will require combining as many of the individual activities outlined above as

possible. In some cases, however, it is strongly advised to combine practices to avoid a mere shift in N_2O emissions over time and not a net decrease in emissions. More specifically, the use of cover crops should always be combined with a synchronized timing of N application with crop demand to avoid an increase in N_2O emissions due to more readily available C sources. Similarly, nitrification and urease inhibitors should also be combined with a synchronized timing of N application with crop demand. Nevertheless, controlled factorial experiments where the interactions of various management practices on N_2O are tested are scarce. Note that biogeochemical models aggregate the scientific knowledge on mechanisms underlying N_2O emissions to the best extent, but they are not exact simulations of reality. Therefore, imprecise prediction of biogeochemical process rates or inadequate simulation of certain management practices can occur. For example, it is common practice to seed red clover into winter wheat to facilitate emergence and growth of the clover immediately after wheat harvest. Nevertheless, the most current version of the biogeochemical model, DNDC, does not allow for concurrent cropping. The inability of biogeochemical models to correctly implement and combine the right practices is explicitly known, and can lead to overestimation of N_2O emissions reductions.

2 Additionality of Nitrogen Management Practices

Only carbon offsets that are generated by practices that are "additional" to what would have occurred anyway will be effective at mitigating climate change. In other words, GHG reductions used as offsets may not have occurred in the absence of a buyer that paid for this GHG reduction as a carbon credit. A range of methods exists to determine that practices that claim to generate emission reductions are effectively additional. The Reserve is committed to using standardized methods. Within the context of additionality, this means that the additionality of projects is evaluated according to explicit and objectively verifiable criteria rather than subjective methods that try to assess a project's individual circumstances. However, an extensive amount of background research is necessary to be able to come up with precise and unambiguous criteria that capture all circumstances. Additionality tests of Reserve protocols generally have two components: a Legal Requirement Test and a Performance Standard Test.

The **Legal Requirement Test** ensures that projects are not required by law. If the law requires a certain practice, such as a legal limit to nitrogen application in ecologically sensitive areas, the practice would have happened in absence of carbon finance and the project is non-additional. A Legal Requirement Test ensures that eligible projects would not have occurred anyway in order to comply with federal, state or local regulations, or other legally binding mandates. A project passes the Legal Requirement Test when there are no laws, statutes, regulations, court orders, environmental mitigation agreements, permitting conditions or other legally binding mandates requiring its implementation or requiring the implementation of similar measures that would achieve equivalent levels of GHG emission reductions. Section 2.1 aims at identifying any current or pending regulatory requirements that would mandate the implementation of the practices considered by the NMPP. Section 2.1 also reviews voluntary incentives for improving nitrogen management. While voluntary incentives are not strictly part of a legal requirement test, they are reviewed here because they may be relevant for developing additionality criteria.

Furthermore, both regulatory and voluntary incentives may be relevant for developing guidelines for ecosystem service credit stacking in the nitrogen management protocol.

Beyond any legally binding requirements, certain practices may still be non-additional if they would have been implemented because there is an economic, technological, or social advantage related to the practice. For example, practices that increase nitrogen use efficiency may reduce costs of fertilizer application. Similarly, cover crops may be planted as a necessary erosion control measure on sloping terrain. The **Performance Standard Test** is intended to evaluate all financial, economic, social, and technological drivers that may affect decisions to undertake a particular project activity. Performance Standard Tests are specified such that the large majority of projects that meet the Performance Standard Tests are unlikely to have been implemented because of these other drivers. In practice, Performance Standard Tests are often specified by analyzing the "common practice" and specifying rules so that projects "go beyond" common practice. In the case of nitrogen management practices, common practice will heavily depend on the geographical area, which means that Performance Standard Tests will differ according to the location.

2.1 Regulatory and Voluntary Incentives related to Nutrient Management

This section provides an overview of the most important regulatory and voluntary mechanisms relevant for developing additionality criteria in the nitrogen management protocol.

2.1.1 Regulatory Requirements

Generally, federal agencies such as U.S. EPA and USDA only set standards to manage pollution caused by or related to agriculture. Federal laws devolve specific rule-making on how to achieve these standards to individual states. Often, federal agencies will offer financial incentives and technical guidance to assist states in implementing the federal standards.

The Clean Water Act

The Clean Water Act (CWA) is the federal law that protects the nation's waterbodies, including lakes, rivers, and coastal areas. Nutrients such as nitrates from cropping fields that contribute to pollution through stormwater runoff or leaching into the water table are specifically included as "nonpoint source pollutants" per amendment section 319. Section 303(d) of the CWA requires that states and tribes identify waters affected by different categories of nonpoint source pollution to the US EPA. This list is commonly referred to as the "Impaired or Threatened Waters List" or the "303(d) list." Each state or tribe must set priorities to clean up waterbodies of the 303(d) list and is responsible for implementing a program to control pollution. In practice, cleaning up waterbodies on the 303(d) list is achieved by first establishing the maximum amount of a pollutant, such as nitrate, that a given waterbody can tolerate without violating water quality standards,[2] and then implementing a concrete plan to reach this limit through a combination of incentives and enforcement. This limit is commonly referred to as the

[2] Note that the term "TMDL" is often used to refer to the whole process of establishing a TMDL implementation plan, including monitoring.

Total Maximum Daily Load (TMDL).[3] If the pollution is partially related to nitrate coming from agricultural sources, the TMDL implementation plan may contain agricultural Best Management Practices (BMPs) such as practices that affect nutrient management or erosion control. The extent to which the BMPs are legally required or voluntary is likely to vary depending on the mechanisms established for a specific waterbody. Therefore, the degree to which a strict "legal requirements test" will filter out nitrogen management practices implemented to meet CWA requirements is variable.

TMDLs and TMDL implementation plans are notorious for being the subject of an often controversial legal and regulatory struggle. Therefore, TMDLs and implementation plans change frequently. Further, in cases where multiple states or other jurisdictions are covered by one TMDL, individual states maintain responsibility for developing their own management plans. Specific activities and management practices are therefore likely to vary between states and other jurisdictional boundaries. In relatively localized TMDLs, such as the TMDL established for controlling nitrate-nitrogen from croplands in the San Luis Obispo Creek,[4] a single management plan with a standardized set of allowable activities is likely to be sufficient.

In contrast, the TMDL for pollutants into the Chesapeake Bay affects 6 states, including New York, Pennsylvania, Delaware, Maryland, Virginia, and West Virginia.[5] The Chesapeake Bay Program was initiated in 1983 by the signing of the Chesapeake Bay Agreement goal to attain the water quality necessary to support the living resources of the Bay by not only reducing the nitrogen loadings, but also the phosphorus and sediment loadings, as well as erosion, to the Bay from controllable sources. In May 2009, Executive Order 13508[6] highlighted the ecological importance of the Chesapeake Bay, and established a Federal Leadership Committee charged with the "management and development of strategies and program plans for the watershed and ecosystem of the Chesapeake Bay and oversee their implementation." The Chesapeake Bay Program partners includes all the affected states, the Chesapeake Bay Commission, a tri-state legislative body, EPA, USDA, and an advisory groups of citizens, scientists and local government officials. NRCS recently came out with a report on the Chesapeake Bay Region *Assessment of Conservation Practices on Cultivated Cropland in the Chesapeake Bay Region.*[7]

The Chesapeake TMDL is the cornerstone of effort to increase the water quality of the Chesapeake Bay. The TMDL represents a far-reaching "pollution diet" that calls for reductions in

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[3] As an example, the following website contains links to all relevant TMDL-related information of California http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/index.cfm

[4] The following is stated within the relevant resolution of the regional water quality control board of the Central Coast Region: "Cropland Source: Landowners and operators of irrigated lands in Prefumo Creek watershed will implement actions needed to achieve the allocations to croplands pursuant to the Conditional Waiver of Waste Discharge Requirements to Irrigated Lands (Conditional Waiver). Implementation and monitoring requirements for parties engaged in agricultural activities are consistent with, and rely upon, the Conditional Waiver." The full text is available at http://www.waterboards.ca.gov/water_issues/programs/tmdl/docs/nitrate_draft_res_attach.pdf

[5] Additional information about the Chesapeake TMDL is available at http://www.epa.gov/chesapeakebaytmdl/

[6] A copy of Executive Order 13508 and associated information can be found at http://www.executiveorder.chesapeakebay.net

[7] Available at http://www.nrcs.usda.gov/technical/nri/ceap/chesapeake_bay/index.html

nitrogen from all states and all sectors. This overall TMDL is broken up by assigning TMDL limits for each of the states in the Chesapeake Bay and each of three sectors agriculture, urban, and wastewater treatment. Since agriculture is the largest single source of nitrogen to the Chesapeake Bay, nutrient management on agricultural lands hold the greatest potential to cleaning up the Chesapeake Bay. Individual states have the authority to decide how they will meet the limit for agriculture. In every state, the TMDL is achieved by a combination of regulatory and non-regulatory approaches discussed below. States are also allowed to work through existing agriculture sector conservation programs to increase implementation of conservation practices as a measure to reach the agriculture-related TMDL.

Most states are still in the planning phases for determining how they will comply with the TMDL limits. In practice, each state is in the process of developing Watershed Implementation Plans (WIP) describing how the TMDL will be reached. The development of the WIPs occurs in multiple phases. All states have submitted their phase I WIP at the end of 2010 and are currently developing drafts for the phase II WIP. Based on the phase I WIPs, states will involve the agricultural sector in reaching the TMDL mainly through four distinct activities: (1) require landowners to develop and implement "nutrient management plans" (Delaware, Maryland, New York) and (2) a voluntary cost-share program to incentivize growers to plant cover crops (Delaware, Maryland), (3) pilot projects to test new manure incorporation methods (Maryland), and (4) requiring growers to implement a buffer zone or setback near the edges of fields where no fertilizer or sludge may be applied (Maryland). Overall, the state of Maryland is probably the furthest along with implementing these practices. Note that in Maryland, several nitrogen management practices are already legally required, including a nutrient management plan on all farmland focusing on commercial and organic fertilizer application rates and dates as well as distance from streams, and maintenance of buffers along stream corridors.

Nutrient management plans are developed by certified private sector nutrient management planners and extension staff and cover nutrient application timing, rates, and application method and the use of soil tests to fine tune application rates. In addition, these plans attempt to maximize use of on-farm nutrients such as manure and cover crops and minimize nutrient imports such as purchased fertilizer. Plans must be updated at least every 3 years. The state of New York is exploring the idea of incentivizing farmers to reduce their N application rate by 15% through crop insurance or other incentives ("Yield Reserve" program). Within the WIP Phase I for the state of Delaware, it is mentioned that this will be required for landowners who control the application of nutrients on 10 acres or greater, other states have not yet indicated an area threshold. The Delaware Phase I WIP further states that audits will be performed and non-compliant landowners will be fined from $50 to $1,000 per violation. Final fines and penalties are addressed through special commissions, such as the Delaware Nutrient Management Commission.

The use of cover crops reduces erosion and prevents nutrients from leaching to groundwater or volatilizing by maintaining a vegetative cover on cropland and holding nutrients within the root zone. This practice involves planting and growing, but not always harvesting of cereal crops with minimal soil disturbance. The cover crop is seeded directly into the crop residue of the previous crop and captures residual nitrogen in its matter as it grows. When the cover crop is

incorporated in spring, this nitrogen is released and used by the subsequent crop. In case of commodity cover crops, the cover crop is first harvested for grain.

Currently, farmers are utilizing vertical tillage equipment such as the "turbo till" to incorporate manure. A new and more efficient injection technology is being piloted with initial funding through Conservation Innovative Grants (CIG) and National Fish and Wildlife Foundation (NFWF) grant. On fields that utilize dairy manure as fertilizer, the manure is incorporated into the soil at the time of application using low disturbance technology. This practice can reduce ammonia loss to the atmosphere by up to 95% compared to traditional surface application. Custom applicators with the equipment will be made to growers if the demand is sufficient.

Maryland is considering a requirement to implement a 10 foot required setback for all fertilizer and sludge application. This requirement would also bring consistency to several other programs regulating nutrients.

Another example of a regional program is the Ohio River Basin Trading Project. The Ohio River is a 303(d) impaired waterbody and the Ohio River Basin Trading Project is an interstate multi-credit trading program and represents a comprehensive approach to improving regional water quality and minimizing costs to the public and stakeholders. The project area covers portions of at least 8 states including Kentucky, Indiana, Pennsylvania, Ohio, West Virginia, Illinois, Virginia, and New York. The project focus is on designing and developing pilot water quality trading markets for nitrogen and phosphorus discharges and greenhouse gas emissions. The project area involves a large set of stakeholders – including regulatory agencies, power companies, farmers, wastewater treatment facilities and other industrial dischargers – working collaboratively to protect watersheds throughout the Ohio River Basin. The project may also benefit downstream water bodies as far away as the Gulf of Mexico.

The third regional example is the Pacific Northwest Region Water Quality Program. The goal of this program is to provide leadership for water resources research, education and outreach to help people, industry and governments to prevent and solve current and emerging water quality and quantity problems. The Pacific Northwest Program is a regional collaboration of federal, state, and local environmental and water resource management agencies. The collaboration places a University Liaison within the offices of EPA Region 10. This partnership is being supported in part by the USDA National Institute of Food and Agriculture (NIFA). The approach to achieving this goal is for the Partners to develop a coordinated regional water quality effort based on, and strengthening, individual state programs. The program has developed guidance for nitrogen management in the form of Best Management Practice (BMP) in agriculture to protect aquifers (groundwater) from nitrates that should be used by agricultural producers include. The BMP includes:

- Apply nitrogen at recommended rates for crop production.
- Use pre-plant soil profile nitrate testing and soil and plant nitrate testing when appropriate.
- Base nitrogen application rates on realistic yield goals.
- Credit nitrogen contributions from legumes, manures, and other organic wastes.

- Plan nitrogen applications to correspond with crop demand (in season).
- Do not apply nitrogen fertilizer in the fall on coarse textured soils or shallow soil over fractured bedrock.
- Use nitrification inhibitors when soil conditions and nitrogen application timing may promote leaching.
- Apply manure uniformly in accordance with crop nutrient requirements.
- Schedule irrigation to minimize leaching.
- Manage fertigation systems carefully.
- Diversify crop rotations to include crops that utilize deep residual nitrogen.

Note that once the identification of impaired waters is approved by the U.S. EPA, state or tribes can apply for grant money for demonstration projects and monitoring projects to assess the success of specific nonpoint source implementation projects, including nitrogen management projects.[8] The additionality of projects that have received such grant money for the explicit purpose of improving nitrogen management must be investigated as part of the Performance Standard Test of the NMPP protocol.

Limits on Trace Levels of Heavy Metals

Because commercial fertilizers and organic amendments contain trace levels of heavy metals, limits are set to the heavy metal contents in fertilizers, organic amendments, and biosolids. Most regulation regarding heavy metal contents in soil amendments is present at the state level, except for biosolids, manure and compost products, which are regulated through U.S. EPA, 40 CFR Part 503.

Under EPA regulations, managers must maintain records on cumulative loading of trace elements only when bulk biosolids do not meet EPA Exceptional Quality Standards for trace elements.[9] Therefore, in case of reducing application rates of composts and biosolids and a cumulative loading of trace elements exists, this will have to be clearly indicated in a project document to demonstrate that the proposed reduction in application rate remains legally additional. However, it is unlikely that this will exclude any projects.

Many states have adopted standards regarding trace metal composition of fertilizers meant to reduce the risk of adding harmful concentrations of trace metals to soils when fertilizer is applied. A comprehensive overview of the heavy metal addition rates for typical fertilizer application rates can be found in EPA (1999).[10] The heavy metal content of commercial N fertilizer is the smallest of all fertilizer classes. Therefore, the Association of American Plant Food Control Officials (AAPFCO), tasked with making regulation among states uniform, stated that metals in N fertilizer generally do not pose harm to the environment as long as the metal

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[8] Examples of section 319 success stories can be found at http://water.epa.gov/polwaste/nps/success319/Section319III_index.cfm

[9] These limits and how to calculate application rates can be found in http://cru.cahe.wsu.edu/CEPublications/pnw0511e/pnw0511.pdf

[10] http://www.epa.gov/oppt/pubs/fertilizer.pdf

concentration in fertilizer is below a specific threshold.[11] In addition to trace metal composition testing, state fertilizer laws generally require product registration, licensing and efficacy testing to assure that statements made on the label are correct. Also, at the state level, fertilizer is primarily regulated for quality, as for any manufactured good. These regulations are usually administered through the state's department of agriculture.

Safe Drinking Water Act

The Safe Drinking Water Act (SDWA), administered by the EPA, contains a legally enforceable limit for nitrate nitrogen in drinking water, set at 10 mg/L. The SDWA requires states and water suppliers to conduct assessments of potential contamination of water sources. States have to implement measures to protect water sources through voluntary programs or legal actions. However, to our knowledge no legally binding limits exist to control nitrogen management within the context of the SDWA. As with the CWA, the main tools to control the set limit are agricultural Best Management Practices. Nevertheless, nitrogen management from agriculture is an important tool in controlling sources of nitrate to drinking water. Even though the SDWA does not mandate any requirements affecting nitrogen management, it may do so in the future. Therefore, future updates to the NMPP protocol should include an analysis of the regulatory changes of the SDWA.

2.1.2 Incentive Programs Promoting Nutrient Management Practices

Many incentive programs exist that provide technical and financial assistance for farmers and ranchers to implement nitrogen management on a voluntary basis. Most of these programs are managed and implemented at the federal level. Other programs, such as EQIP are implemented at the state level.

Except for the Coastal Zone Management Act, all programs described below are supported by the NRCS. It is the mission of the NRCS to support a large number of programs and opportunities for financial or technical assistance with nitrogen management activities. An overview of all NRCS programs is available at http://www.nrcs.usda.gov/programs/. This background paper provides a short overview of the relevant aspects to nitrogen management in each of the programs. Projects that receive funding from these programs to improve their nitrogen management may be subject to further additionality criteria.

Coastal Zone Management Act (CZMA)

The CZMA encourages states/tribes to preserve, protect, restore or enhance natural coastal areas, including wetlands, floodplains, estuaries, beaches, and dunes. Eligible areas border the Atlantic, Pacific, and Arctic Oceans, Gulf of Mexico, Long Island Sound, and Great Lakes. Participation is completely voluntary. To encourage states/tribes to participate, the act makes federal financial assistance available to develop and implement a comprehensive coastal management program. Most eligible states/tribes participate in the program. Section 6217 of the CZMA, administered jointly by EPA and the National Oceanic and Atmospheric Agency

[11] See http://www.aapfco.org/rules.html for the specific heavy metal threshold concentrations.

(NOAA), specifically supports states to develop and implement nonpoint pollution control programs for coastal areas.[12] Within a guiding document specifying typical measures to control nonpoint source pollution published by the EPA[13] in 1993, commercial N fertilizer is identified as a pollutant to coastal areas. Management measures to reduce pollution include development and implementation of a nutrient management plan focusing on (1) applying nutrients at rates necessary to achieve realistic crop yields, (2) improving the timing of nutrient application, and (3) using agronomic crop production technology to increase nutrient use efficiency. In 2003, EPA updated and expanded the 1993 coastal nonpoint source manual to address the control of agricultural nonpoint source pollution for the entire United States.[14] National Management Measures to Control Nonpoint Source Pollution from Agriculture highlights best available, economically achievable means of combating nonpoint source pollution, and discusses monitoring techniques, load estimation techniques, and watershed approaches.

Agricultural Management Assistance (AMA)

The AMA program provides cost share assistance to agricultural producers to voluntarily address environmental issues, including water management, and water quality by incorporating conservation into their farming operations. The only measure that is supported that is relevant within the context of nitrogen management is a transition to organic farming. Other measures, only tangentially relevant to nitrogen management include production diversification and resource conservation practices. AMA is only available in 16 states (CT, DE, HI, ME, MD, MA, NV, NH, NJ, NY, PA, RI, UT, VT, WV and WY). Total AMA payments are limited to $50,000 per participant for any fiscal year. More information is available at http://www.nrcs.usda.gov/programs/farmbill/2008/pdfs/ama_factsheet2008_final.pdf.

Conservation Stewardship Program (CSP)

The CSP program provides payments for producers to address resource concerns by improving, maintaining, and managing existing conservation activities and undertaking additional conservation activities. Activities relevant to nitrogen management covered by the CSP include[15] (activity codes are provided in brackets):

- Injecting or incorporating manure (AIR01)
- Use of nitrification and urease inhibitors (AIR02)
- Using nitrogen provided by legumes, animal manure and compost to supply 100% of the nitrogen needs (ENR08)
- Continuous cover crops and Use of Cover Crop Mixes (SQL02 and SQL04)

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[12] See http://coastalmanagement.noaa.gov/nonpoint/welcome.html
[13] Available at http://water.epa.gov/polwaste/nps/czara/MMGI_index.cfm
[14] Available at http://water.epa.gov/polwaste/nps/agriculture/agmm_index.cfm
[15] A full list of activities is available at http://www.nrcs.usda.gov/programs/new_csp/2011/other-pdfs/csp-conservation-activity-list012411.pdf

- Apply nutrients no more than 30 days prior to planned planting date(WQL05)
- Apply controlled release nitrogen fertilizer (WQL06)
- Apply 50% or more of the total nitrogen needs after crop emergence (WQL07)
- Split applications of nitrogen based on a Pre-Sidedress Nitrogen Test (PSNT) or other crop-based indicators (WQL08)
- Plant a cover crop that will scavenge residual nitrogen (WQL10)
- Use of legume cover crops as a nitrogen source (WQL16)
- Transition to Organic Cropping Systems (WQL20)

The CSP program is available to producers in all states. Payments are limited to $40,000 per year.

Conservation Technical Assistance (CTA)

The CTA program is designed to provide technical assistance to individual landowners, communities, local and state governments, and others who plan and implement natural resource conservation systems on their lands. In practice, CTA is used to develop site-specific conservation management practices using NRCS standards and guidance. A wide range of conservation practices is included. Within the context of nitrogen management, the CTA program indicates it can support the development of plans that improve soil and water quality. A full description of the program is available at http://www.nrcs.usda.gov/programs/cta/.

Environmental Quality Incentive Program (EQIP)

The EQIP initiative is a voluntary conservation program established in the 1996 Farm Bill, and amended in subsequent Farm Bills, to assist farmers to implement a wide range of conservation practices, including nitrogen management practices (referred to as nutrient management practices by EQIP) over a period of maximum 10 years. Available to producers in all fifty states, the EQIP program uses a set of National and State-specific priorities to decide which practices will get funding. Program payments are limited to a person or entity to $300,000 for all contracts entered into during any 6-year period. EQIP provides an important nitrogen management opportunity to producers by providing cost share to develop a comprehensive nutrient management plan, known as a conservation activity plan (CAP),[16] which includes recommendations on nutrient application rates. By implementing a CAP, nitrate fertilizer can be reduced by 20% - 40% (Trachetenberg and Ogg 1994).

The EQIP National Organic Initiative helps producers plan and implement conservation practices that allow their organic operations to be environmentally sustainable by providing financial assistance to National Organic Program (NOP) certified organic producers as well as producers in the process of transitioning to organic production. Applicants must either have an organic system plan that meets the NOP guidelines or certify that they are working toward one. Organic

[16] More information on the EQIP CAP program is available at http://www.nrcs.usda.gov/programs/eqip/cap.html

producers can receive up to $20,000 per year or $80,000 over six years through this initiative. Organic producers may also apply for assistance under general EQIP. More information on the EQIP Support for Organic Growers program is available at http://www.nrcs.usda.gov/programs/eqip/organic/index.html.

Local-level initiatives

Local-level initiatives may exist around particularly problematic watersheds or vulnerable areas. For example, the **Indian Creek Watershed Program** in Illinois aims at implementation of water quality conservation practices in the Indian Creek watershed. It is estimated that 50-75 % of the watershed's agricultural producers will adopt comprehensive agricultural conservation systems. Project partners plan to measure water quality improvement results over a 6-year period.

2.2 Data to Establish Common Practice

Performance Standard Tests specify how projects can demonstrate that they are additional. The design of Performance Standard Tests requires an analysis of the common practice so that rules can be drafted to verify that projects go beyond the common practice. A particular challenge related to NMPP is that there are a large number of practices that may be affected by N_2O-reducing nitrogen management, including annual N fertilizer application rates, N fertilizer type (nitrate, ammonia, urea, manure, mixed cover crop or compost), or timing and application (number of fertilization events, broadcasted, injected, applied with irrigation water). A more comprehensive overview of potential practices is provided in Table 1. Ideally, common practice is analyzed for each of the potential practices. We have divided our review on the nitrogen management common practice according to the geographical scale at which the data are available.

This section is focused on providing an overview of the individual primary data sources. However, it will be evident that the available primary data sources are insufficient to fully define the common practice. Much information can be gained if the data from the primary sources is combined and secondary datasets are derived. Section 2.2.3 discusses approaches to combined primary data sources into useful secondary datasets that are of immediate relevance to the nitrogen management protocol.

2.2.1 Crop Acreage

Fertilizer rates are dependent on the specific crop planted. Often, fertilizer use data is aggregated across crops. Therefore, understanding differences in aggregated fertilizer use across geographies, such as counties or states, or over time requires knowing the crop acreage for a specific geography or year.

NASS Surveys (Annual Summaries)

The NASS annual summary surveys focus on county-level yields and acreages of crops grown and state-level expenses. The crops and practices covered by the NASS survey (annual summaries) are selected to cover over 80% of planted acres. NASS increases samples when other states contribute funds to the program. Core crops are surveyed every other year on even/odd basis for different crops. During some years, some data may be missing due to lack of

funding. For example, NASS did not survey corn in 2007. Confidentiality requirements are very strict; therefore, NASS will not divulge information for an individual producer. In practice, at least five complete reports are required for sufficient data for statistical analyses. If less than five complete reports are available, no statistics will be published for that data specific item. We do not think this is a major challenge for using the data within the context of creating a performance standard test. Results are published by political boundaries, with the state-level being the smallest level; county-level results may be possible if there are enough surveys in the state. Data can be most easily queried using the web-based QuickStats 2.0 system. The URL for QuickStats and more information on how to build relevant queries are available in Appendix C.

The NASS Cropland Data Layer

The NASS collects geospatially explicit information on crops planted for the total U.S. using 30-m Landsat data and produces a publically available GIS layer called the Cropland Data Layer. Data can be queried by state, county, or Agricultural Statistics Districts. Agricultural Statistics Districts (ASD) are defined as a contiguous group of counties having relatively similar agricultural characteristics. Data can be easily queried using the CropScape system, a geospatial data service from USDA. CropScape offers interactive visualization, web-based data dissemination and geospatial queries of acreage over multiple years. It also provides for automated data delivery to systems such as Google Earth. The system is available at http://nassgeodata.gmu.edu/CropScape/.

2.2.2 Fertilizer Use, Rates, Type, Expenses, and Sales

This section aims at providing a comprehensive overview of all available data sources related to fertilize use, fertilizer type, and application rates. At a federal level, USDA National Agricultural Statistics Service (NASS) is tasked with gathering data on a variety of issues including agricultural production, economics, and the environment. A number of data products from the NASS contain relevant information on fertilizer use, rates and sales. In addition, data products from industry organizations and state services are also available.

NASS Surveys (Annual Summaries)

The NASS annual summary surveys, detailed in the previous section also contain state-level fertilizer expenses. Only state-level data is available and the expense of fertilizer is summed with lime and soil conditioners.

NASS 2008 Organic Survey

As a follow-up to the 2007 Census, USDA conducted its first-ever, wide-scale survey of organic agriculture in 2008. [17] Survey results provide vital data on many aspects of organic farming, including production, marketing and economics. It appears to be one of the few resources providing data on use of organic mulch or compost. Relevant data includes:

- Statewide number of farms using no-till or minimum till (Table 13 and 32)

[17] http://www.agcensus.usda.gov/Publications/2007/Online_Highlights/Organics/index.asp

- Statewide number of farms that produced or used organic mulch/compost (Table 13 and 32)

NASS Agricultural Chemical Use Program

NASS Agricultural Chemical Use Program is the U.S. Department of Agriculture's official source of statistics about on-farm and post-harvest fertilizer and chemical use and pest management practices. While data availability is not consistent over time and geographic extent, it is probably the most comprehensive and detailed survey available on fertilizer use for U.S. agriculture. Only a limited number of crops are reviewed each year. Surveys are conducted in odd numbered years for fruit (latest available survey performed in 1995, reported in 1996) and even numbered years for vegetable crops. The survey is conducted in collaboration with the Economic Research Service (ERS) and the Agricultural Resource Management Service (ARMS). This release includes chemical application rates and acres treated by major producing states for field crops annually (corn, soybeans, cotton, potatoes, wheat). The sample is designed to provide coverage of all farms in the 48 contiguous States plus state level data for 15 major cash receipts states. However, some major crops are missing, e.g., processing tomatoes in California. This source only includes inorganic fertilizer, and not organic fertilizer. Data and reports are available at http://www.nass.usda.gov/Statistics_by_Subject/Environmental/index.asp. A summary is available at http://www.ers.usda.gov/briefing/AgChemicals/nutrientmangement.htm#fertilizer. Table 12 in Appendix A summarizes the available data per crop.

USDA Economic Research Service (ERS) Fertilizer Use and Price Datasets

A comprehensive overview of historical fertilizer application rate, consumer prices, and different crop acreage data at state level (i.e. state as minimum mapping unit) is available at http://www.ers.usda.gov/Data/FertilizerUse/ and summarized in Table 2. These data are partially derived from the NASS and the ARMS datasets, and as such are not a primary source for current data. However, these data do provide a unique overview of historical trends since the 1960s. Note that a long time period between surveys may exist, which increases the potential that year-to-year variation obscures underlying trends.

Table 2. Overview of data available from USDA Economic Research Service (ERS) Fertilizer Use and Price Datasets.

Data Item	Crop (if relevant)	Year	Smallest Geographical Extent
Nitrogen used on corn, rate per fertilized acre receiving nitrogen (Table 10, 16, 22, 28)	Corn, Cotton, Soybeans, Wheat	1964-2007	AL CO DE FL GA IL IN IA KS KY MD MI MN MS MO NE NY NC ND OH PA SC SD TN TX VA WI
Plant nutrient use by selected crops (tons) (Table 2)	Corn, Cotton, Soybeans, Wheat, Other	1964-2009	U.S.
Anhydrous Ammonia, Aqua, Ammonia, Nitrate Ammonium, Sulfate Ammonium, Nitrogen solutions, Sodium nitrate, Urea, Other (Table 4)	Aggregated across all crops	1960-2009	U.S.
Price of Anhydrous ammonia, Nitrogen solutions (30%), Urea 44-46%, nitrogen, Ammonium nitrate, Sulfate of ammonium (Table 7)	Aggregated across all crops	1960-2011	U.S.

Indices of Fertilizer Consumption by State, 1960-2004

The ERS has published total statewide fertilizer consumption over the last 5 decades. The data are indices and normalized according to a benchmark (i.e., the fertilizer consumption for Alabama in 1996). These data are available at: http://www.ers.usda.gov/Data/AgProductivity/.

Manure Production

In an analysis prepared by the NRCS and the ERS titled "*Manure Nutrients Relative to the Capacity of Cropland and Pastureland to Assimilate Nutrients: Spatial and Temporal Trends for the United States,*"[18] a descriptive analysis is presented of the temporal and spatial changes in the quantity of manure nutrients produced for every county. Manure production is concentrated in a limited number of regions around the country since livestock populations have become spatially concentrated in high-production areas. In these high-production areas, the amount of manure nutrients vastly exceeds the assimilative capacity of land available. Consequently, manure produced will have to be transported over a significant distance before it is applied, potentially impacting the GHG balance of manure application.

[18] Available at http://www.nrcs.usda.gov/technical/nri/pubs/manntr.pdf

Association of American Plant Food Control Officials (AAPFCO) Data

Since 1985, the AAPFCO at the University of Kentucky compiles data on the annual sales of commercially produced fertilizer from licensed fertilizer distributors who are required to report sales of all fertilizer products to State regulators. Commercial fertilizer sales data are available from AAPFCO dating back to 1985. Before 1994, the data were compiled and originally published by the Tennessee Valley Authority (TVA). Generally, the AAPFCO data are provided at the county level for most states in the U.S. However for some states and years, (some) county-level data may be missing. All data that could not be attributed to a specific county was attributed to an "unknown" county category. In practice, AAPFCO data are available for about 70 to 75% of the counties. An overview of the distribution of N over different N fertilizer types based on the AAPFCO data can be found in Table 2-2 of http://www.epa.gov/oppt/pubs/fertilizer.pdf for different main agricultural production areas in the U.S. Gaither and Terry (2004)[19] provide a detailed description of the AAPFCO data.

Data from the Fertilizer Institute (TFI)

TFI is the trade association representing the fertilizer industry. TFI sells the "Commercial Fertilizers Report," which contains historical data on U.S. fertilizer sales by nutrient back to fiscal year (July-June) 1960 (but not by state) at http://www.tfi.org/publications/stats.cfm. In addition, TFI reports information on fertilizer import and export, fertilizer production, fertilizer producer inventories and the number of fertilizer producers.

Ad-hoc and State-Level Data Sources

Many ad-hoc and state-level data sources exist. It is beyond the scope of this background paper to list all sources. We limit ourselves to provide a number of examples. Such resources may prove useful for refining performance standards to be geography and crop specific.

State departments may have different mandates and missions to collect data related to nitrogen management. In California, fertilizer sales data are collected at the county level by the California Department of Food and Agriculture (CDFA 2009). However, there is no explicit collection of fertilizer use data. Many states' extension services have published reports on the implementation of certain agricultural practices. For example, a statistical review of California's organic agriculture is available at http://aic.ucdavis.edu/publications/Statistical_Review_05-09.pdf. Sometimes extension services have also published cost-and-return studies for agricultural practices. These are usually based on limited surveys, discussions with extension agents and expert opinion. For example, estimates of fertilizer rates in California are available from the University of California's ARE Cost and Return Studies (from 2000 until present). These studies are based on conversations with producers and extension staff and are segregated according to broad regions and include most common management practices. Cost and return

[19] Gaither, Kelly, and Terry, D.L., 2004, Uniform fertilizer tonnage reporting system — version 4 — Instruction manual: Association of American Plant Food Control, 87 p., available online at *http://www.aapfco.org/UFTRSDoc.pdf*

studies often remain conservative in the cost estimates. Therefore, many growers reported applying less N than reported in the cost and return studies (Rosenstock et al., 2011).

The peer-reviewed scientific literature contains many ad-hoc studies and reviews of fertilizer use. For example, Rosenstock et al. (2011) contains a detailed review of the literature on fertilizer application rates in California from 1940-2010, with crop-specific estimates for the years 1973 and 2005. The review is based on a number of ad-hoc papers, such as Zhang et al. 1998, and ARE (2010) (with a smaller range of crops).

Another example of an ad-hoc state-level data source is Minnesota's FANMAP program, which surveys existing farm practices regarding agricultural inputs such as fertilizers, manures and pesticides and the voluntary adoption of BMPs. In 2007, about 60 farm interviews were completed. Information on total N application rate, timing of N applications (quantity of N applied at sidedress, fall, at planting, at emergence, and spring preplant), sources of N application (quantity of N applied as urea, liquid N, anhydrous ammonia, or diammonium phosphate/monoammonium phosphate). Reports are available at http://www.mda.state.mn.us/protecting/soilprotection/fanmap.aspx.

2.2.3 Timing and Number of Fertilizer Applications

Optimizing the timing of N applications is important to increase the efficiency of fertilizer N use and reduce the chance of N losses through nitrification, denitrification, or leaching. No data source that was fully comprehensive over crops, geographical areas and that was consistent in time was available to determine timing of N applications. However, a few data sources with ad-hoc or incomplete data on timing and application of fertilizer are available and discussed below.

Agricultural Resource Management Survey (ARMS)

The ARMS survey collects field-level information on (among other items) nutrient and crop residue management practices for a limited set of survey states and crops. The states surveyed are dependent on the crop. Data is not available for a limited set of years. The survey collects data on application rates, fertilizer timing, and fertilizer application method.[20] Four categories are used to determine fertilizer timing: in the fall before seeding, in the spring before seeding, at seeding, after seeding. In addition, the survey collects information on the fertilizer application method and includes the following categories: broadcast - ground without incorporation, broadcast - ground with incorporation, broadcast - by aircraft, in seed furrow, in irrigation water, chisel/injected or knifed in, banded in or over row, foliar or directed spray.

The NASS Agricultural Chemical Use Program

As part of the NASS Agricultural Chemical Use Program, the number of fertilizer applications per year is recorded. No information on timing of N applications is recorded, however. As noted in

[20] The fertilizer survey instrument is available at http://www.ers.usda.gov/Data/ARMS/app/ARMSDocs/Questionnaires/W%5E2009%5EAll%5EPhase2%20Fertilizer%20Supplement%5EQ%5ECOP_CPP.pdf

the previous section, the data are not consistent over time and geographical extent and only a limited number of crops are reviewed each year.

Cost and Return Studies from Extension Services and Land-Grant Universities

Some state-specific data are available. For example, the cost-and-return studies developed by the Department of Agriculture and Resource Economics of the University of California in Davis (first noted above) contains valuable information regarding timing and average number of fertilizer applications. The cost and return studies are created through interviews with three to five full-time and successful farmers and represent best management practices rather than true averages of practices (Klonsky, pers. comm.). The studies are updated about every five years. Cost and return studies are available at http://coststudies.ucdavis.edu/. Table 3 contains three examples of the level of detail that is contained within the cost-and-return studies.

Table 3. Example of the level of detail regarding common nitrogen management practices available in the cost-and-return studies of the Department of Agriculture and Resource Economics of the University of California in Davis.

Crop	Geographical Applicability	Common Nitrogen Management Practice
Field Corn	Mineral soils in the Sacramento Valley	• At planting in April, 151 pounds (15 gallons) of ammonium phosphate (10-34-0) is applied per acre. • Aqua Ammonia (20-0-0) is applied as a sidedress in May at a rate of 225 pounds (152 gallons) of N per acre.
Tomatoes (fresh), Melons, and Winter Squash	Placer & Nevada Counties	• A cover crop of mixed legumes and grasses for nutrients and erosion control is planted in the fall. • Every other year in April, compost applied at 10 tons per acre, and blood meal (13-0-0) at 800 pounds per acre are applied.
Wheat for Grain	Tulare, Kern & Madera Counties	• In November, prior to land preparation, anhydrous ammonia at 130 pounds per acre is applied by the grower using an injection rig. • In February, N as urea is applied top dress by air at a rate of 50 pounds per acre. • In April, 40 pounds of N as UN32 is applied in the irrigation water.

2.2.4 Secondary Data Sources

The previous section reviewed primary data sources, i.e., data sources that reported original data. A number of authors and organizations have attempted to combine different primary data sources into derived data to get to crop-specific and/or county-specific N application rates.

United States Geological Service (USGS) County-Level Nutrient Input Report

In 2006, the USGS published a report with calculated county-level estimates of total fertilizer N and manure use (in kg N per county) (Ruddy et al., 2006) for 1982 until 2001.[21] The data are available at http://water.usgs.gov/pubs/sir/2006/5012/excel/Nutrient_Inputs_1982-2001jan06.xls. The authors of this report state-aggregated fertilizer sale data from AAPFCO to individual counties using the county-specific fertilizer expense data from the agricultural census to obtain the total county-level N application use. For about half of the states, the AAPFCO data differentiates sales of products used in non-farm areas from products used on-farm. If no differentiation is made between on-farm and non-farm use of fertilizer, a farm allocation coefficient must be applied first. This "farm-to-total" fertilizer use ratio was calculated at a national level. The total county-level N application use was further divided into farm and non-farm use using a so-called "farm-to-total" coefficient calibrated based on counties for which nitrogen application use data were available.

NuGIS Project

The NuGIS initiative by IPNI (IPNI, 2010) is an on-going assessment of nutrient balance and nutrient use efficiency in crop production.[22] In technical bulletins, the NuGIS initiative integrates multiple data layers to create county-level estimates of nutrient removal by crops, fertilizer applied, and excreted and recoverable livestock manure nutrients. The most recent one is published in 2010 and is still in a draft state. Nutrient balances and recovery efficiencies by the balance method were estimated for the five U.S. Agricultural Census years from 1987 through 2007. In contrast to Ruddy et al. (2006), the NuGIS approach did not disaggregate the AAPFCO data from the county level to the state. If only state-level data were available, census data were used to allocate fertilizer sales to individual counties, even though the time basis for reporting is different. Most states report fertilizer sales to the AAPFCO not on a calendar-year basis as reported in the agricultural census, but on the basis of a year ending on June 30th. The same farm-to-total fertilizer use ratios as used in Ruddy et al. (2006) (see previous section) were used to obtain nitrogen used in farms.

General Issues with Using Secondary Data Sources

As recognized in the 2010 NuGIS report, the use of county-level N fertilizer sales data to inform county-level N fertilizer use is questionable. The county-level N fertilizer sales data portray a geographic distribution of fertilizer sales that is unlikely to match actual use because of (1) the large amount of fertilizer transported among counties (Rosenstock et al., 2011), and (2) the possibility that fertilizer purchased in one year is only applied a following year. Due to the uncertainty introduced by cross-county fertilizer transport, it is necessary to spatially aggregate individual counties into larger geographical entities (IPNI, 2010).

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[21] The report is available at http://pubs.usgs.gov/sir/2006/5012/pdf/sir2006_5012.pdf.
[22] More info on NuGIS is available at http://www.ipni.net/NuGIS

2.2.5 Temporal and Spatial Variability of Fertilizer Rates and Prices

The section above investigated potential data sources and approaches that can be used to inform common practice. An equally important and related issue is how variable common practice changes over time and space due to external factors, and what such external factors may be. This analysis will inform the optimal geographical scale at which common practice can be defined, and how often the common practice must be updated.

Temporal Variability of Fertilizer Prices and Application Rates

Retail nitrogen fertilizer prices are reported by the U.S. Department of Agriculture (USDA) and are available on USDA web site in a monthly report entitled *Agricultural Prices* accessible at http://usda.mannlib.cornell.edu/MannUsda/viewDocumentInfo.do?documentID=1003.

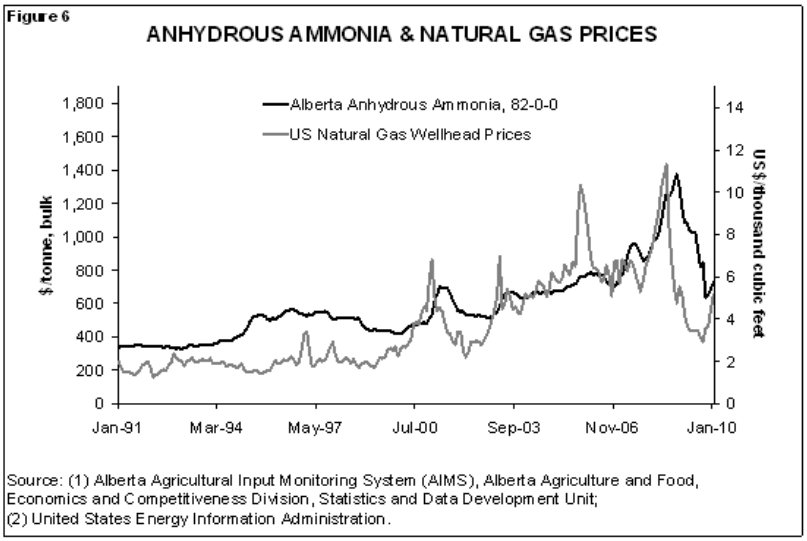

Figure 1. Relation between the price of ammonia and natural gas. (Source: Canadian Farm Fuel and Fertilizer: Prices and Expenses, 2010 - http://www.agr.gc.ca/pol/mad-dam/index_e.php?s1=pubs&s2=rmar&s3=php&page=rmar_02_07_2010-11-26.)

Nitrogen fertilizer prices have increased over 300% in less than a decade (e.g., Figure 1 for anhydrous ammonia). The increase in fertilizer price is related to the following factors:

- **Price of oil and natural gas**. Anhydrous ammonia is produced by converting natural gas into hydrogen gas, which (catalytically) reacts with air to form liquid anhydrous ammonia. Therefore, prices of fertilizer are closely linked to prices of natural gas. In addition, most nitrogen fertilizers are produced in China. A significant amount of oil is used during transportation to the end user.
- **Food demand**. Population growth in China and India particular are driving an increase in food demand, and hence, incentivize increases in agricultural production and fertilizer demand. Additionally, increases in per-capita income in developing economies drives dietary changes and increases demand for meat. Increased meat consumption increases the per-capita demand for grain and protein feeds.

- **Demand for biofuel**. The production of corn-based ethanol has created a strong demand for N fertilizer affecting worldwide N fertilizer prices.
- **Geo-political events**. For example, the dispute over natural gas pricing in Russia and the Ukraine may have decreased availability of natural gas in Europe, potentially leading to increases in fertilizer prices.
- **Exchange rates**. A weakening currency will increase prices of imported fertilizer.
- **Speculation**. Expectations of changes in any one of the factors above may motivate speculators to trade in large volumes of commodity stocks, affecting the fertilizer price.
- **Environmental factors.** Global fluctuations in weather patterns, such as the fluctuations induced by El Nino - Southern Oscillation affect agricultural production. Similarly, pest and disease outbreaks threaten global production. These environmental factors affect prices for agricultural commodities, and therefore fertilizer prices.

While N fertilizer prices are highly volatile, studies have found that fertilizer use in developed countries is price-inelastic due to lack of economic substitutes to chemical/synthetic fertilizer (Korol and Larivière, 1998; Rosas, 2011). In less developed countries, fertilizer is more price-elastic, because (1) readily available substitutes such as manure and other organic material exist and (2) capital lacks to pay for the increase in fertilizer prices.

Predictions of future fertilizer demand and application rates are based on global crop production, demand, and fertilizer use models. One such model is the WorldNPK model, which is linked to the FAPRI agricultural production model (Rosas, 2011).[23] This set-up explicitly includes fertilizer use in agriculture commodity production modeling. Rosas (2011) concluded that fertilizer application rates will most likely remain equal in the U.S. during the period 2010 until 2025,[24] even if N fertilizer prices increase as well as demand for agricultural commodities.

Historically, U.S. corn yields have continued to increase since the beginning of the 1980s, despite a slight decrease in the fertilizer application rate (Figure 2). Since 1995, application rates of N fertilizer have remained fairly constant. Increases in yields were driven by agricultural technology and increases in N use efficiency.

[23] http://www.card.iastate.edu/publications/dbs/pdffiles/11wp520.pdf
[24] http://www.fapri.iastate.edu/outlook/2011/tables/8_fertilizer.xls

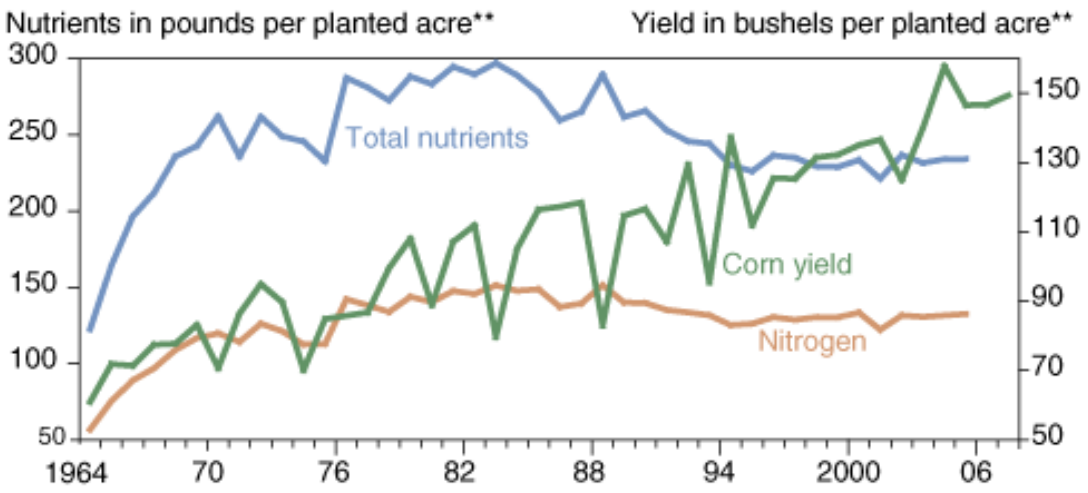

Figure 2. Historical nitrogen application rates and corn yields between 1964 and 2008. (Source: USDA ERS and NASS, copied from http://www.ers.usda.gov/AmberWaves/December09/Features/USCornYields.htm.)

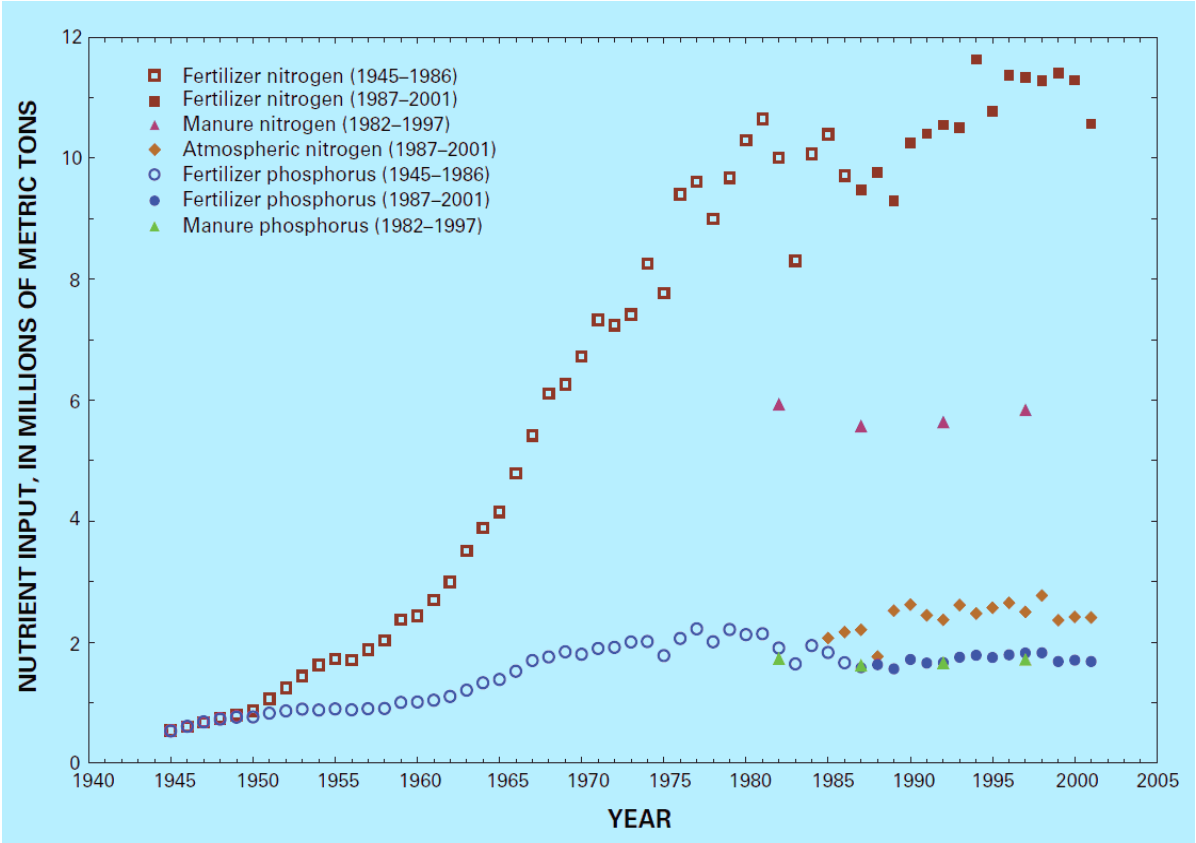

Figure 3. Annual nutrient inputs to the conterminous United States, 1945-2001. (Data from County-Level Estimates of Nutrient Inputs to the Land Surface of the Conterminous United States, 1982–2001.)

Also shown (Figure 3) are previous estimates of nitrogen and phosphorus inputs from fertilizer for 1945 through 1986 (Alexander and Smith, 1990; Battaglin and Goolsby, 1995). Nitrogen input from fertilizer increased from less than 1 million metric tons per year before 1950 to more than 10 million metric tons per year by 1980. Since 1980, annual nutrient inputs from most sources have been approximately constant; however, nitrogen input from fertilizer has continued to increase but at a lower rate and with higher variability.

Spatial Variability of Fertilizer Use

The crop response to fertilizer is strongly affected by soil characteristics. Soil and site considerations of particular importance are those that affect the potential for nutrient retention and nutrient loss from the system. Such parameters include soil texture – nitrogen leaching will be more prevalent in sandy soils compared to clayey soils, drainage and porosity, slope, and the Cation Exchange Capacity (CEC). All of these parameters are known to vary greatly among fields, and even within one field. The latter is exemplified by high-resolution satellite images of the Normalized Difference Vegetation Index (NDVI) of one field during the growing season. The NDVI is an index that is correlated to crop health. For many fields, it can be observed that the NDVI changes significantly within one field. Precision agriculture takes advantage of these data by adjusting fertilizer rates according to the location within one field and, thus, supplying exactly the amount of fertilizer that is needed based on the inherent soil fertility of that location.

2.3 General Recommendations for Establishing a Performance Standard

Performance Standard Tests are intended to evaluate all financial, economic, social, and technological drivers that may affect decisions to undertake a particular project activity in a streamlined and standardized manner. The Performance Standard Tests are specified such that the large majority of projects that meet the Performance Standard Tests are unlikely to have been implemented because of these other drivers. In practice, projects pass a Performance Standard Test by meeting or exceeding a performance threshold, i.e., a standard of performance that is established by investigating the regional common practice for a specific management activity. In other words, Performance Standard Tests are specified so that projects go (well) beyond common practice. In the case of nitrogen management practices, common practice will heavily depend on the geographical area, which means that Performance Standard Tests will have to be location-specific.

A performance standard test may combine one or both of the following requirements (1) the proposed project activity is not common practice, and (2) there is a demonstrated lack of adoption of the project activity on the agricultural field prior to the existence of carbon projects. The first requirement applies when adoption rates of the project activity are very small in the absence of carbon projects. In that case, adoption of the project activity will be additional under most circumstances. For example, if the adoption rate of switching from fall to spring fertilizer N application would be 2%, any producer who chooses to apply fertilizer N in the springtime might be eligible for of a carbon project. Currently, the highest baseline adoption rate in protocols by the Climate Action Reserve is close to 7%, for methane gas capture at landfills. With the second mechanism, projects are considered additional if the

producer can demonstrate that the project activity was not implemented on the field on which the carbon project is planned during some number of years or cropping seasons prior to the start of the carbon project. Both types of requirements could be included in the Performance Standard Test. A hybrid approach, in which a threshold adoption rate of the region in which a project is located is combined with a limit on the historical adoption on the field on which the project is planned, may be particularly sensible when adoption rates of certain practices are greater than a typical threshold percentage.

Developing a full performance standard baseline requires:

(1) Selecting the right parameters that are used to define the common practice

(2) Selecting the right geographical level over which common practice can be considered homogeneous

(3) Deciding which data sources can be used to determine values for parameters at the selected geographical level

(4) Mechanics on how and when to update the performance standard

2.3.1 Selection of Parameters Included in Common Practice Definition

A transparent, consistent and unambiguous set of rules must be created to determine whether a certain management practice is additional for a given geographical region. Using these rules, it could very well be decided that one specific management practice is eligible in one geographical region, and not in another. A good basis for rules regarding additionality for nitrogen management practices can be found in the CSP and EQIP program documents. A distinction should be made between parameters that related to practices that are either implemented or not (e.g., cover crops), referred to as “binary practices” and parameters that relate to gradual changes to existing practices (e.g., reducing the total fertilizer application rate using residual N soil tests), referred to as “gradual practices.”

Additionality of Binary Practices

The additionality of binary practices could be set in terms of a maximal adoption rate. For example, it could be decided that any practice that has a “business as usual” adoption rate above X% - where X is, for example, 15% - is ineligible within a certain geographical region. The following is a first attempt at providing the parameters that can constitute a performance standard for binary practices.

- Adoption rate of planting a mixed cover crop for at least a certain number of months to immobilize N outside of growing season and supply N in the growing season
- Adoption rate of nitrification and urease inhibitors at a certain minimal adoption rate
- Adoption rate of using a Pre-Sidedress Nitrogen Tests (PSNT) or other crop-based indicators to determine split applications

Additionality of Gradual Practices

In case of gradual practices, additionality must be set as a minimal deviation from the common practice average. Examples of additionality tests for gradual practices are:

- At least X% of N is supplied by manure and compost, where X is, for example, 30%
- More than X% of N is supplied within 30 days prior to planting, where X is, for example, 50%

2.3.2 Selection of Geographical Scale to Determine Common Practice

Selecting the proper geographical scale to determine common practice must balance the variability in agricultural practices due to changes in soil and climate and the availability of the data. Logical options for a sound geographical scale include (1) state level, (2) Agricultural Statistics Districts, or (3) individual counties. Since agricultural practices can differ drastically within one state, the state level is most likely too broad to base common practice on. On the other hand, at the county level, insufficient data are available to determine common practice. While it is technically possible to get fertilizer use data at a county level, it will likely not yield accurate results due to limitations of the derivation approach to create secondary datasets, most notably the assumption that all fertilizer sold within one county is applied within the same county (IPNI, 2010). Therefore, we recommend using combinations of individual counties as the right geographical scale to determine common practice. For example, Agricultural Statistics Districts (ASDs),[25] used by NASS for reporting, can be used, as done by IPNI (2010) and CAMCO (2011). Figure 4 shows an example of the agricultural districts for Kansas.

[25] A shapefile of ASDs is available at http://www.nass.usda.gov/Charts_and_Maps/Crops_County/boundary_maps/asds.zip

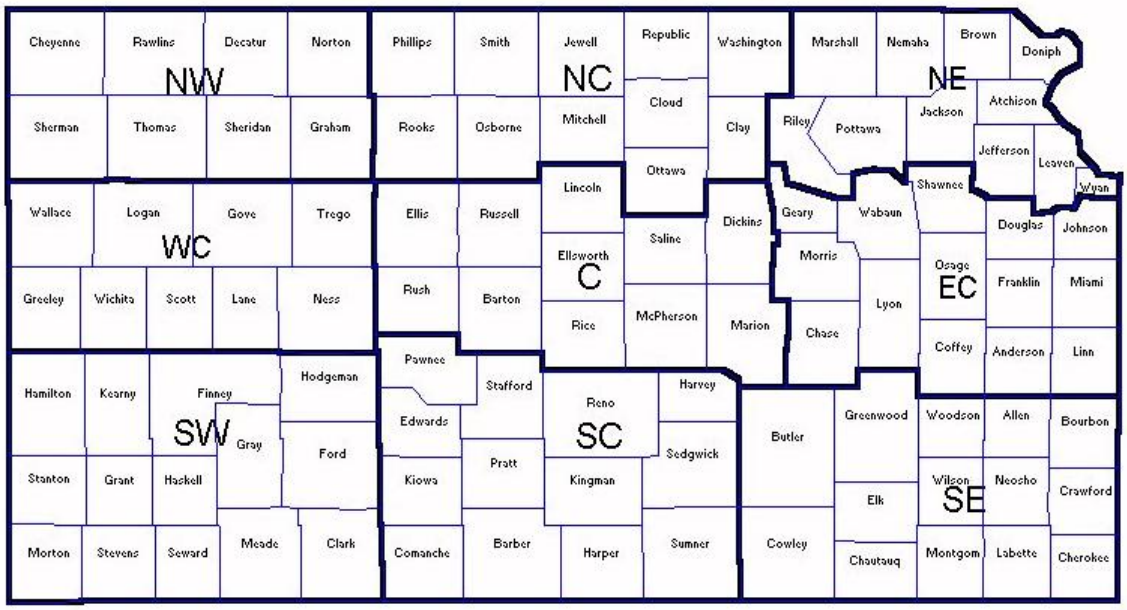

Figure 4. Agricultural Statistics Districts for the state of Kansas.

Alternatively, counties could be combined according to available data sources. For example, counties in California could be combined according to the cost and return studies from UC Davis.

2.3.3 Data Sources to Determine Common Practice

No comprehensive and field-level survey data exist on specific N fertilizer application rates, timing, or practices. State-level average N application rates, timing, or practices and relative standard errors (RSE) are available from the ARMS of the ERS/NASS/USDA. However, these data are not consistent over time and only available for some agricultural states.

A logical alternative for directly measured or surveyed N fertilizer application rates, timing, or practices is to use fertilizer recommendations from agricultural extension services to determine the common practice, as done under approach 2 of the MSU-EPRI methodology.[26]

Alternatively, secondary/derived data on N application rates at sufficient geographical accuracy can be developed by combining state- or county-level fertilizer sales data from AAPFCO with total expenditures on fertilizers per crop from the agricultural census and total fertilized acres per crop from the agricultural census. The indirect approach was followed by Ruddy et al. (2006), CAMCO (2011), IPNI (2010). The latter is the most sophisticated approach to deriving fertilizer use data. The authors of IPNI (2010) are still refining county-level fertilizer use estimates and the data are not yet publically available. However, according to Dr. Paul Fixen,

[26] The MSU-EPRI methodology is currently undergoing validation with the Verified Carbon Standard (http://www.v-c-s.org/methodologies/quantifying-n2o-emissions-reductions-us-agricultural-crops-through-n-fertilizer-rate-0) and American Carbon Registry (http://www.americancarbonregistry.org/carbon-accounting/methodology-for-n2o-emission-reductions-through-fertilizer-rate-reduction).

one of the main authors of the IPNI report, an effort to fine-tune the fertilizer use effort will probably be completed later in 2011. CAMCO (2011) reports "Nitrogen factors" for the NASS crop reporting districts, which is the nitrogen used per yield output instead of an application rate. This nitrogen factor is used primarily to investigate how an individual farm's nitrogen efficiency ranks compared to the average within a crop reporting district. This approach is not crop-specific and is currently only valid in areas in which one crop is dominant. These data can be partially updated annually when new AAPFCO becomes available and fully updated every 5 years when the agricultural census is updated. To assess the accuracy of the indirectly derived N application use data, these data must be cross-checked with directly measured or surveyed ad-hoc studies. In areas where the number of crops is limited and the crop rotations are fairly simple, total application rates can be disaggregated into crop-specific application rates by using a mathematical model. However, deducing crop-specific application rates becomes particularly challenging in areas with complex crop rotations, such as California.

2.3.4 Updating the Performance Standard

While N fertilizer prices are highly fluctuating and the total N fertilizer use is expected to increase, it is reasonable to assume that, on average, N fertilizer rates will not significantly increase in the short-to-medium term. This suggests that performance standards based on fertilizer application rates may remain relevant for at least a ten-year crediting period. However, N fertilizer rates may change in smaller areas with specific circumstances. Therefore, it will be necessary to periodically update and revisit the performance standard. As data resources that underpin the eventual performance standards are update periodically (approximately every 5 years) it would be possible to also update the performance standards periodically as well if it is determined to be necessary.

2.4 Evaluating N Use Efficiency of Reducing N Rates to Inform Performance Standard Development

Before potential metrics to quantify N use efficiency and define a performance standard are discussed, we go further into the currently used approaches to decide how much N to apply. This analysis of the common practice and the metrics that are used in that context is the best starting point for analyzing different metrics to define a performance standard.

2.4.1 Choosing and Evaluating Fertilizer N Rates

Many individual factors influence the response of a crop to the application of N fertilizers, including the quantity of residual nitrogen present in a soil, soil pH, crop species and variety, timing and number of N fertilizer applications, N application method (injection, broadcasted, applied in bands, applied in irrigation water, etc.), deficiency of other plant macro- and micro-nutrients than N, and cultivation techniques and tillage intensity. Furthermore, the impact of these individual factors on yields is strongly regulated by weather and irrigation practices. All of these factors make the development of accurate N fertilizer recommendations very challenging.

Prior to the mid-1970s, most crop N rate recommendations were based on soil–specific criteria and /or on crop management variables such as rotation and manure application. In the last 4

decades, however, extension services associated with land-grant universities have published N rate recommendations based on N response trials for many states, particularly the Midwest.

In order to develop a performance standard test for reducing N_2O emissions by reducing N application rate, it is important to thoroughly understand different strategies commonly used to determine N application rates. When recommended rates are predictably related to actual on-the-ground N application, then the recommended rates could serve as a useful proxy indicator for likely common practice in a particular setting. In what follows, four different approaches to set a fertilizer N rate are discussed.

Yield Goal

The yield-based approach recommends an N rate that will support the agronomic maximum yield presumed to be achievable on a particular field. In this approach, the producer determines his or her yield goal for a particular field based on yields achieved on that field in previous years. The N application rate for any given year is determined by multiplying the yield goal by a factor (often 1.2 lb N/bu) that expresses the amount of N required per unit of expected yield.[27]

Throughout the last four decades, yield goal has been one of the major strategies to determine N application rates for cropland. The multiplication factors are in turn based on N rate response trials, field experiments in which the yield is measured on small plots in which the N rate is varied. Nevertheless, Sawyers et al. (2006) called the yield-goal based N recommendations into question because: (1) a poor relationship between these recommendations and the most economically advantageous N rate (Figure 5 and Figure 6), (2) uncertainty about how yield goals should be determined, (3) a false underlying assumption that N use efficiency is constant across sites and years, and (4) inadequate or inappropriate adjustments for non-fertilizer N sources.

[27] In many states in the NCR, the yield-goal approach recommends to apply 1.2 pounds of N per bushel for corn following corn, with credits (adjustments) given when corn follows a legume or where manure is applied. Such yield-goal factors are usually developed by extension agents working in land-grant universities.

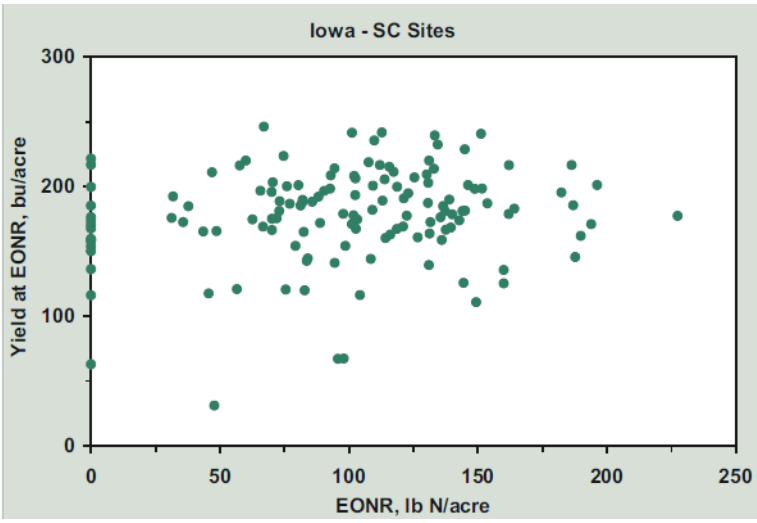

Figure 5. Relation between Yield and Economically Optimum N Rate across sites and years in Iowa. (Copied from Sawyer et al.,2006.)

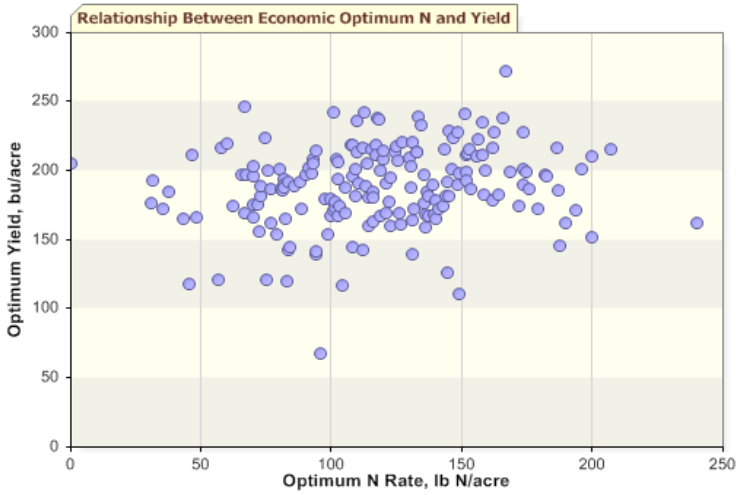

Figure 6. Relationship between site economic optimum N and Yields for continuous corn in Iowa. (Graph copied from http://extension.agron.iastate.edu/soilfertility/nrate.aspx.)

Routine N Application Rates

Alternatively to yield-goal based N rate recommendations, many producers apply 'routine' N application rates, implying that the same N rate has been applied to the field for many years, where past satisfying outcomes have entrusted producers to maintain that rate. Since routine N application rates most likely represent a strategy that has provided consistent and satisfactory yields, regardless of climatic conditions and available N at the start of the growing season, producers applying N at a routine N rate can likely decrease the amount of applied N if N rate

recommendation strategies that include adjustments for non-fertilizer N sources would be consulted.

Alternative N rate recommendations, to the traditional factors noted above, are available that aim both to maximize profits and reduce the risk of excess N being lost to the environment. Those strategies include (1) applying the most economic N rate rather than the N rate that is expected to produce a yield goal or maximize yield (see section 2.1.1), (2) adjusting applied N for residual nitrate based on a soil N test (see section 2.1.2), and (3) fine-tuning N rate applications after optimizing timing, placement and type of N, or by taking into account N inputs from legumes and crop residues or other soil characteristics (see other sections in background paper). Adoption of any of those strategies or a combination of multiple of those strategies has the potential to reduce N application rates without risking yield loss (Figure 7)

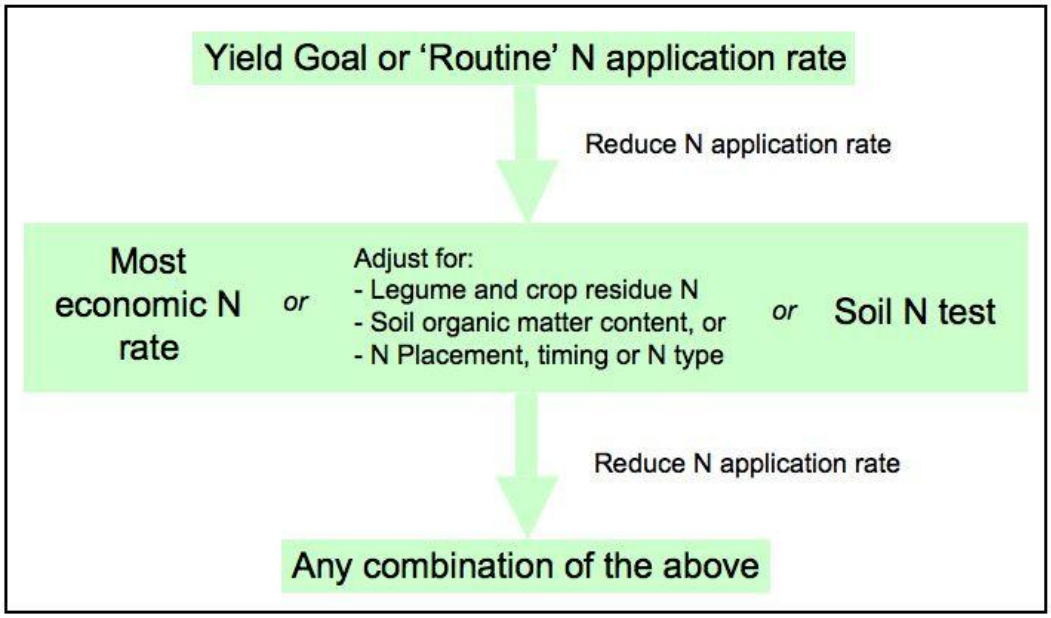

Figure 7: Visualization of the availability of strategies to improve N rate recommendations and reduce N application rates without risk for yield loss.

Optimum Economic N rate instead of yield goal or 'routine'

The yield-based approach has been questioned because the maximal yield is not always the most profitable goal as the yield-based approach does not account for the cost of fertilizer inputs. In addition, yield-based recommendations often had inadequate or inappropriate adjustments for non-fertilizer N sources (Sawyer et al., 1996). In contrast to yield-maximizing approaches, N recommendations that are based on Economic Optimum N Rate (EONR) and Maximum Return to N (MRTN) focus on maximizing profit instead of yields. The EONR is the point where the last increment of added N returns a grain yield increase large enough to pay for that N (Figure 8), whereas MRTN refers to the N rate where the economic net return to N application is greatest.

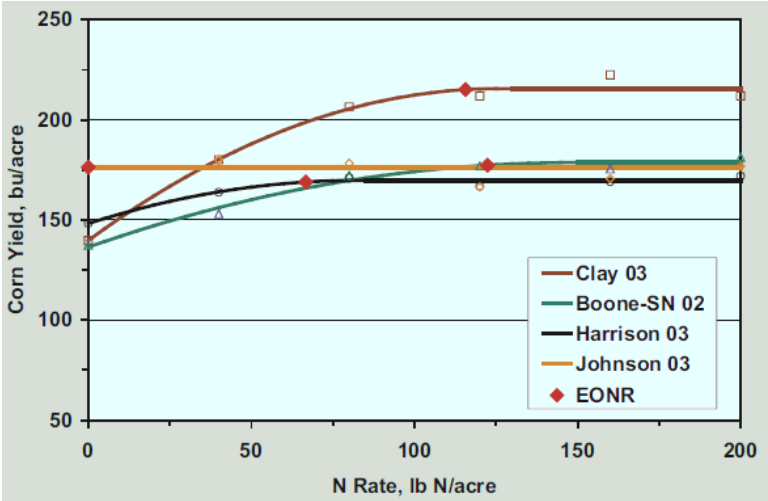

Figure 8. Demonstration of how the Economically Optimum N Rate (EONR) is calculated for 4 example field trials.

Generally, N rate recommendations based on MRTN and EONR are smaller than yield-goal based recommendations, suggesting a potential to reduce N application rates by switching from a yield-goal approach to an approach that seeks to maximize profit. Furthermore, given current crop and N prices, MRTN and EONR based recommendations are generally not associated with a significant yield loss. Note that, as discussed in the leakage section, the current economics of carbon projects are such that it is unlikely that producers will deliberately reduce N rate to values where yield is significantly affected. However, it may still be possible that N rates are reduced to a level where yields are not affected, while soil organic matter is mined. This may be the case in high SOM soils, and has important implications for soil carbon sequestration (Jaynes and Karlen 2008). It should further be underscored that neither the yield-goal approach nor approaches that seek to maximize profit (EONR and MRTN) are concerned about N use efficiency, i.e. the ratios of N outputs to inputs.[28] In other words, the approaches do not consider the agronomic and ecological efficiencies of the systems. Nevertheless, the link between N rate, yield, N use efficiency and excess N is inherently difficult. For example, Rosenstock et al. (2011) found an increase in nutrient use efficiency in California cropping systems of 37% from 1973 to 2005, while a greater amount of surplus N was applied in 2005 compared to 1973. Hence, nitrogen use efficiency had increased, but not enough to offset the increased pollution potential arising from greater N use (Rosenstock et al., 2011). While some efforts are geared towards matching N application with crop N demand through optimizing timing, placement and fertilizer formulations, difficulties concerning the link between N rate, yield, N use efficiency and N loss have certainly driven current N rate recommendations towards maximizing yield and/or profit instead of optimizing N use efficiency.

The idea underlying EONR and MRTN is essentially the same. However, EONR is a value that has commonly been determined for an individual field experiment or trial, while MRTN values are

[28] A discussion on the diversity of measures used to characterize N use efficiency can be found in Rosenstock et al. (2011).

based on a compilation of individual datasets and serves as a recommendation to farmers supported by a wide group of experts. Thus, MRTN values can be seen as economic optimum N rate for a selected number of states based on a compilation of a large data set.[29] The locations where MRTN recommendations are available include: Iowa, Illinois, Indiana, Michigan, Minnesota, Ohio, and Wisconsin. Table 4 contains an example of MRTNs for different states and price ratios published in Sawyer et al. (2006). Also, with the MRTN approach, a profitable N rate range (PNRR) can be determined, defined as the range of N application rates that will return a profit within $1.00/acre of the MRTN.

Table 4 contains an example of MRTNs and high and low limits of the PNRR for different states and prices of fertilizer (in $/lb N) and prices of corn (in $/bu). Different MRTNs have been developed for various states within the NCR. Furthermore, it was acknowledged that state boundaries do not necessarily mark out land areas with uniform yield response curves. Therefore, statistical analyses on response curves within states were used to identify regions where yield responses to added N were significantly different. This led to the development of diverse MRTNs for various geographic regions in Illinois and Indiana and for fields with contrasting yield potentials or soil types in Wisconsin.

Millar et al. (2010) suggested that applying the lower N rate within the PNRR could be incentivized by a C market and would bring about economic and environmental advantages that would far outweigh any marginal productivity benefits achieved in applying a higher N rate. In the MSU-EPRI protocol, however, the performance standard test, is passed if the N rate is reduced below baseline rates established according to one of two approaches described below. Hence, the use of MRTNs and lower limit PNRR N rates are merely a suggestion. The protocol further suggests that other methods may be used to reduce rates to better match crop needs, including the use of improved fertilizer application timing, fertilizer formulations, cover-crop N capture, or any of number of other practices known to better match N fertilizer input to crop N needs than BAU approaches. For calculating baseline N rates, the protocol includes two approaches. In approach 1, the baseline N fertilizer rate is determined from the project proponents' management records for 5 to 6 years prior to the project implementation year. Approach 2 applies when historic N rate data are not available. In that case, baseline N application rates are back-calculated based on crop yield data at the county level, available from the United States Department of Agriculture – National Agricultural Statistics Service (USDA-NASS), and equations for determining N fertilization rate recommendations based on yield goal estimates (e.g., found in state department of agriculture and land grant university agriculture department documents). Note that, where MRTN or EONR are already recommended, literature and documentation on yield-goal based N rate recommendations might not be readily available from extension services and therefore hard to access for a project developer. Furthermore, in the absence of frequency distributions of yields and N application rates based on statistically valid surveys, it cannot be guaranteed that county-

[29] MRTN values for the respective states and price ratios can be found using the N rate calculator, available at http://extension.agron.iastate.edu/soilfertility/nrate.aspx and based on Sawyer et al. (2006).

averaged yields and associated N recommendation rates are representative for the baseline scenario at a particular field.

Table 4. Example of MRTNs for different states and price ratios (adapted from Sawyer et al., 2006).

		MRTN		Low***		HIGH*	
Price Ratio* $/lb:$/bu	N Rate $/acre	Net bu/acre	Yield bu/acre	N Rate lb N/acre	Yield bu/acre	N Rate lb N/acre	Yield bu/acre
Soybean-Corn (SC) Rotation							
Illinois							
0.05	197	130.62	177	170	175	221	178
0.1	163	110.98	174	143	172	186	176
0.15	141	94.3	172	122	168	161	174
0.2	122	79.86	168	106	165	140	172
Iowa							
0.05	145	96.65	180	126	179	170	181
0.1	123	81.78	179	107	176	144	180
0.15	109	69.05	177	93	174	125	179
0.2	95	57.8	174	82	171	111	177
Minnesota							
0.05	120	77.96	161	101	159	142	161
0.1	101	65.86	159	86	157	119	161
0.15	90	55.46	158	76	155	103	160
0.2	80	46.2	156	68	153	93	158
Wisconsin							
0.05	138	80.51	171	117	170	168	172
0.1	107	66.87	169	98	167	133	171
0.15	101	55.22	168	91	166	114	169
0.2	95	44.28	167	79	163	107	169
Continuous-Corn (CC) Rotation							
Illinois							
0.05	213	156.32	154	184	152	239	155
0.1	176	135.19	152	156	149	199	154
0.15	154	117.08	149	136	146	174	151
0.2	137	101.09	146	122	142	154	149
Iowa							
0.05	200	158.98	144	179	142	234	145
0.1	174	138.36	142	153	139	196	143
0.15	152	120.53	139	138	136	171	141
0.2	140	104.35	137	125	133	156	139
Minnesota							
0.05	148	129.66	153	133	151	168	153
0.1	136	114.09	152	123	150	150	153
0.15	126	99.69	151	114	148	139	152
0.2	118	86.23	149	103	146	131	151
Wisconsin							
0.05	165	105.61	165	140	164	197	166
0.1	139	89.21	164	124	162	157	165
0.15	127	74.62	162	111	159	141	164
0.2	112	61.38	159	97	156	129	162

* Corn grain price held constant at $2.20/bu; N prices at $0.11, $0.22, $0.33, $0.44/lb N.

** LOW and HIGH approximates the range within $1.00/acre of the MRTN for each ratio.

Using soil and plant tissue N tests

A major unknown in the yield goal or economic optimum N rate recommendations is the amount of residual N that is left from the previous cropping season, which affects the actual amount of additional N input required for crops to get what they need. Soil tests offer one way to detangle some of the complexity of the response of a crop to the application of fertilizers. Soil testing has been used to guide extension recommendations for determining appropriate N rate for many years (the PSNT[30] test in particular), and data collected by the farmer using these tests are incorporated into yield goal recommendations. Using soil test results to determine N application rates is often referred to as the sufficiency approach. Under the sufficiency approach, fertilization is reduced to a small starter when soil test levels reach the agronomic critical value. No further application is recommended when a soil test indicates high residual nitrogen content. Only the additional nutrients needed to realize a crop yield response are recommended after nutrients from soil, past manure applications, tilled sods, etc. have been accounted for. The sufficiency approach tends to lead to much smaller (up to 35%) N application rate recommendations, which has led to some controversy. However, these tests are not very widely adopted for a variety of reasons, including time commitment and lack of efficacy in some regions. More specifically, as noted in the IPNI report "Selecting the right fertilizer rate: A component of 4R nutrient stewardship," the usefulness of one single soil N test at the beginning of the growing season to determine N application rates is smaller in more humid climates because weather-induced variations in inorganic N concentrations during the growing season affect the ability of a soil test to predict N availability to the crop. Examples of more humid climates are the Corn Belt, Southern, and Southeastern states.

Multiple soil N tests during the growing season that determine application rates of split applications may be more efficient in humid climates. Alternatively, a leaf color chart (LCC), a chlorophyll meter, or an optical sensor can be used to determine a crop's immediate N requirements at multiple times during the growing season without having to spend the time and effort to test soil samples. The LCC is a simple color chart that growers can compare to actual crop leaf colors, originally developed for use in rice production systems in Southeast Asia. The chlorophyll meter and optical sensor are devices that are clipped on a crop leaf and measure the "greenness." Both the LCC and the chlorophyll meter/optical sensors are based on the principle that different shades of leaf colors indicate N deficiency to excessive N content. Using the LCC or a chlorophyll meter, producers can make real-time decisions regarding N requirements on a site-specific basis. Prices of chlorophyll meters and optical sensors are significantly greater than for the LCC.

Residual N tests are fairly economical. Testing for both nitrate and ammonia usually does not cost more than $15 when the sample is delivered to a laboratory. This value does not take into account the time the producer has to take to take the soil samples and transport the samples to

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[30] Pre-sidedress Nitrogen Test (PSNT) is a soil test for nitrate-nitrogen (NO_3-N) developed for use at the 4 to 6 leaf stage of corn to help in making more accurate N fertilizer recommendations at sidedressing time (http://www.spectrumanalytic.com/support/library/rf/Presidedress_Nitrate_Nitrogen_Test_University_Summary.htm)

a soil testing laboratory. As a rule of thumb, a sample should be analyzed every 20 acres and each sample must be composed of soil from 15 to 20 cores taken within the 20 acres to account for variability (e.g., Michigan's soil nitrate test for corn data sheet available at http://stjoecountycd.com/Documents/Soil%20nitrate%20test%20for%20corn%202010.pdf).

Many states have adjusted their fertilizer recommendations to include residual soil N tests. For example, N recommendations in Iowa are based on cropping system and results of a soil nitrate test (Blackmer et al., 1997). In Wisconsin, N recommendations were revised in 1990 using a soil-specific approach based on the results of numerous N response trials conducted on the major soils used for corn production. No actual data was found to quantify adoption rates, but several experts and extension agents indicated that the actual adoption of soil tests by producers to determine N rates remains low.

N recommendations that acknowledge the complex response of a crop to the application of fertilizers also include approaches geared towards optimizing N use efficiency through fine-tuning the timing and application mode of N applications. The timing of N fertilizer applications crucially affects the efficiency of fertilizer N because the longer the duration between application and crop uptake, the greater the risk of N losses due to processes such as leaching and denitrification. Obviously, the application of N during the period of maximum crop demand may not be practical or possible. Nitrogen timing options usually include fall applications, spring pre-plant applications, sidedress or delayed applications made after planting, and split or multiple treatments added in two or more increments during the growing season. Alternatively, slow-release fertilizer can further reduce losses from leaching and denitrification. However, an additional shortcoming of the yield-goal approach as well as approaches that optimize profits is that they do not explicitly provide information on how to optimize N use efficiency.

Section 2.5.4 discusses two recent surveys on the adoption rate of soil nitrate tests. If the amount of residual soil N is available, the residual N ("N Credit") is subtracted from either the MRTN-based recommended N rate or the yield goal recommended N rate to adjust the general recommendation to better match the specific N needs at the field level.

2.4.2 Metrics to be used in Performance Standard Tests

Various metrics can be proposed for performance standard tests for N rate reductions. Here, we divide the metrics into two categories: (1) metrics based on an absolute N rate and (2) metrics that relate the N application rate to N availability and the amount of N removed by harvest (i.e. metrics that assess the level of N use efficiency). Examples of potential mechanisms of those metrics are provided for each category and advantages and disadvantages are discussed.

The first category includes metrics based on an absolute N rate. When choosing an absolute N rate for a performance standard test, it is crucial to set the maximum acceptable N rate to a value that promotes project activities to go beyond common practice and ensure additionality. State-level average N application rates and relative standard errors (RSE) are available from the ERS of the NASS/USDA. While the RSE is helpful to understand the variability around the mean N rate, it only indirectly reflects the spread of N rates across fields. A full distribution of N application rates or a standard deviation would provide more information by showing how

much deviation from the average is typical. The standard deviation can be derived from the relative standard error if the number of observations is known. Information on the spread of N application rates such as a full distribution of N application rates or the standard deviation can enable more choices for a performance standard threshold, e.g., setting thresholds based on percentiles instead of averages.

The second category of metrics is more sophisticated than using the absolute N rate and includes a partial N balance of the agricultural field. Instead of observed average N application rates reported by NASS/USDA, one of the three approaches described above for deriving recommended N rates could be used to set a performance standard threshold if it were understood that the recommendation represented better than common practice. Recommended N rates are often available from land grant universities (e.g. the Corn Nitrogen Rate Calculator from Iowa State University). Recommended N rates generally vary from state to state, or even from county to county, and are subject to change. Before using recommended N rates in a performance standard test, a good understanding of how the recommendations relate to actual on-the-ground N application rates is required, e.g., by evaluating how recommended N rates compare to actual N application rates as reported by NASS/USDA. The feasibility of average N application rates or recommended N rates for use in a performance standard test will be further evaluated in section 2.5. It should also be noted that recommended and average N rates do not account for or acknowledge variability in soil fertility between fields within a state or county. Therefore, they do not necessarily reflect the most appropriate N rate for one particular field, however as noted above, use of soil N tests is one way to better adapt general recommendations.

In contrast to absolute N rates, a performance standard test could be expressed in terms of N applied or available relative to N removed by harvest. For example, in a recent report by USDA-ERS on the use of nitrogen in agricultural systems (Ribaudo et al., 2011), best management practices for N application rate were assumed when the excess N application did not exceed 40% of N removed by harvest. These authors reported that 35% of corn producers in 19 states surveyed did not meet this N rate criterion (hence 65% did meet the best management definition). The high adoption rate for application of an appropriate N rate either suggests that most producers do not apply excessive fertilizer N under currently available technology, or indicates that the threshold of 40% excess fertilizer application is not very exclusive. It was not immediately clear what the rationale was behind the 40% threshold used in Ribaudo et al. (2011). Applying N in 40% excess of N removed by harvest coincides with a nutrient use efficiency of 71% (see section 2.4.2). In comparison, recovery of applied N in crop plants is often times less than 50% worldwide (Fageria and Baligar 2005). For the 33 most important crops observed in California, nutrient use efficiencies ranged between 13 and 75% (Rosenstock et al. 2011). At first sight, comparison with other systems suggests that N application rates observed in corn cropping systems in the NCR are reasonable. Nevertheless, it is important to note that nutrient use efficiencies are relative measures. As corn is known to receive high N application rates compared to other crops, the absolute amount of excess N in those systems may still be substantial, despite the relatively high nutrient use efficiency. Furthermore, interpretation of nutrient use efficiencies can be misleading, given the diversity of approaches that have been used to determine nutrient use efficiencies, each leading to slightly different results

(Rosenstock et al., 2011). Nevertheless, nutrient use efficiencies are interesting and potentially useful as a performance standard metric because they take into account a simple field-specific N balance and can promote N application rates appropriate for individual fields based on producers' experience and knowledge of soil characteristics. Information on achievable nutrient use efficiencies with current technology on farmers' fields for various crops and regions could be gathered through expert surveys.

We have expanded the idea of basing an appropriate metric on a partial N balance and present different "flavors" of metrics in the remainder of this section. However, to fully understand the value of the different metrics, some understanding of the N budget of agricultural systems is required. A simplified overview of all inputs and outputs of N within an agricultural system is presented in Figure 9.

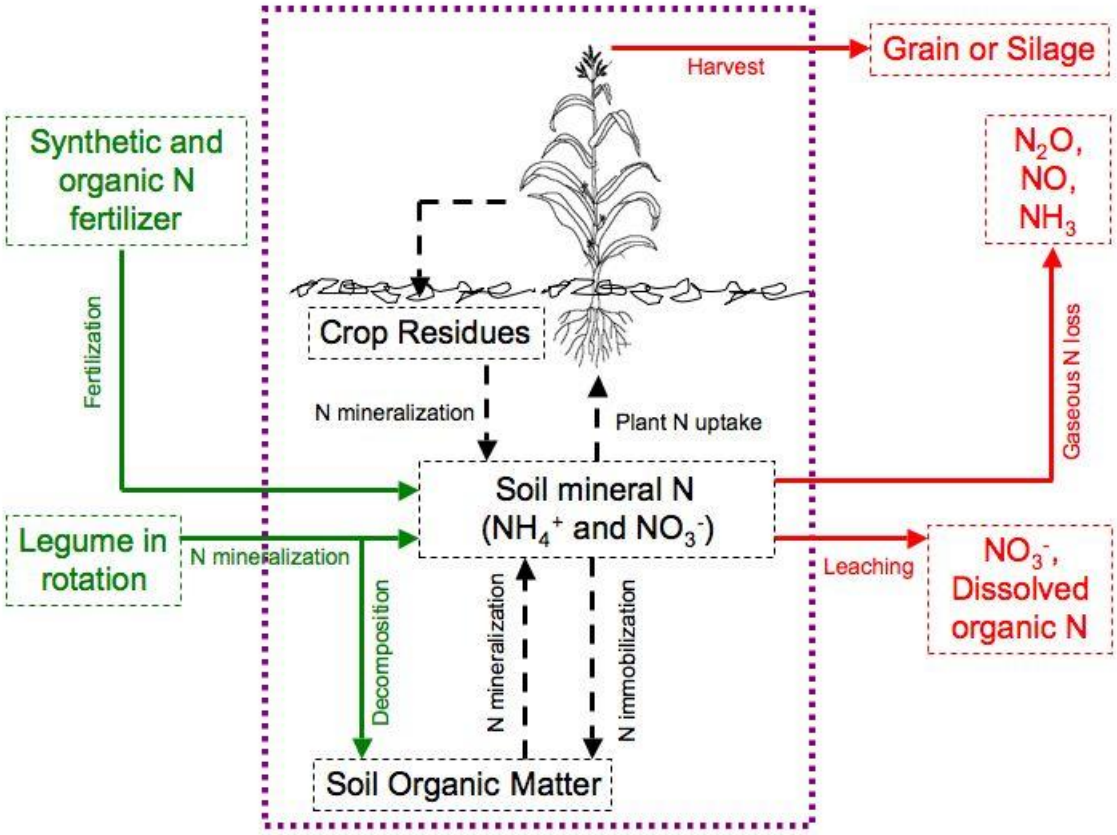

Figure 9: Nitrogen sources, cycling and losses in agricultural systems. Red arrows represent losses from the system, green arrows external inputs and dashed arrows internal recycling. The purple dotted line marks the accounting boundary. (Drawing of corn plant was obtained from www.inra.fr.)

In most agricultural systems, synthetic N fertilizer (e.g., anhydrous ammonia or urea) and organic fertilizer (e.g., manure, compost, or sewage sludge) represent the major inputs of nitrogen in the system. Those are the N sources referred to when the phrase N application rate is used. N can be lost from the system through leaching, NH_3 volatilization or emission of NO,

N_2O or N_2. Furthermore, N is taken away from the system through harvest. The amount of N removed by harvest depends on the crop, and on the usage of the crop. For example, more N is removed from the system when corn is used for silage compared to grain. In addition, there is recycling of N in the system, and residual N can carry over from one season to the other. Agricultural systems contain N in the form of soil organic matter and crop residues. Upon mineralization, this N becomes available to plants. If soil organic matter contents are high and a large amount of crop residues return to the soil, less N from external inputs might be required. Note that timing of mineralization should match the timing of plant N needs or mineralized N should be retained well in the system under this scenario. Crop residues can be derived from a cover crop, a leguminous crop or a cash crop that was cultivated in previous season. In the case where leguminous crops are cultivated, as in corn-soybean rotations, the N credit (carryover) to the next crop can be particularly high. Often times, high concentrations of residual NO_3^- from the previous season are measured at the beginning of a growing season. This NO_3^- is readily available for plant N uptake.

In order to assess excess N in the system, partial N balances can be constructed. Following Fixen (2010), the partial N balance can be expressed as the ratio of removed to available N, further referred to as RTA ("Removed To Available"). RTAs smaller than 1 suggest excess N after harvest, RTAs close to 1 suggest a relatively well balanced system and RTAs larger than 1 indicate that a considerable amount of N other than applied N was taken up by the plant. The simplest partial N balance is an N balance that only takes into account the amount of N applied and N removed by harvest (RTA_1). In that case, RTA_1 can be calculated as follows:

RTA_1 = N_removed/N_applied

Where,
N_removed = Amount of N removed by harvest (lb N/acre)
N applied = Amount of N applied to the field (lb N/acre)

N removed by harvest can be calculated as follows (Fixen 2010):

N_removed = crop_yield x N_content

Where,
For corn grain:
crop_yield = The crop yield (bushels/acre)
N_content = The N content of the harvested crop (0.7 lb N/bushel, Fixen 2010[31])

[31] This value represents an average. Corn grain N content is known to vary drastically. Boone et al. (1984) reports an average of 1.54%N (dry matter basis), with 92% of samples between 1.34 and 1.74%N. Assuming a standard moisture content of 15.5%, this corresponds to an average of 0.73 and a range of 0.63 to 0.82 lbs N/bushel. Note that a different reference from IPNI indicates a removal rate of 0.9 82 lbs N/bushel (see http://nanc.ipni.net/articles/NANC0005-EN).

For corn silage:
crop_yield = Crop yield (U.S. tons/acre)
N_content = N content of the harvested crop (9.7 lb N/ton, Fixen 2010).

Data to calculate RTA_1 at the state level are readily available. Unfortunately, no data to calculate RTA_1 at the field level are readily available. There is a direct relation between RTA_1 and the "nitrogen excess" approach in Ribaudo et al. (2011). More specifically:

RTA_1 = 1/(N_excess/100+1) and N_excess = 100/RTA_1+100

Where,
N_excess = N_excess in %
The 40% N excess used in Ribaudo et al. (2011) corresponds with an RTA_1 of 0.71. The RTA_1 can give a skewed perception of the system's N budget, especially if N mineralization rates or residual N are high.

In systems where soil N tests are conducted at the beginning of the growing season, the residual soil NO_3^- content could be included in the partial N balance. The RTA in that case is calculated as follows (RTA_2):

RTA_2 = N_removed/(N_applied + residual_N)

Where,
N_removed = As above
N_applied = As above
residual_N = The residual soil NO_3^- as indicated by a soil test in kg N/ha.

Note that RTA_2 reflects better the N balance in the system than RTA_1. A disadvantage is that information on residual NO_3^- in the beginning of the growing season is not always readily available. Across cropping systems and agricultural regions, 27% of producers used a soil or plant tissue test to determine an appropriate fertilizer N rate (Ribaudo et al. 2011). In a different survey conducted in Minnesota (Bierman et al., 2011), 49.6% of the farmers indicated they used a soil nitrate test at least once in the past 5 years. These observations imply that soil N tests, while fairly common, are not used annually by the majority of farmers. The RTA_2 metric requires the availability of sufficient data on residual soil N in U.S. cropping systems in order to select a critical RTA_2 value to be used in a performance standard test. Alternatively, residual N can be approximated by calculating excess N from the previous season, and assuming that 30% of N applied during that season was leached.[32]

[32] This is consistent with IPCC 2006. However, it is likely that this value has a large amount of uncertainty around it.

residual_N_x = (0.7 x N_applied$_{x-1}$) – N_removed$_{x-1}$

Where,
N_applied$_{x-1}$ = Amounts of N applied by harvest in the previous growing season (x-1) in lb N/acre
N_removed$_{x-1}$ = Amounts of N removed by harvest in the previous growing season (x-1) in lb N/acre

In cropping systems where legumes are included in the rotation, N from mineralization of leguminous crop residues can be included in the budget. RTAs can be calculated without (RTA_3) and with (RTA_4) consideration of residual soil NO_3^-:

RTA_3 = N_removed/(N_applied + legume_N)

RTA_4 = N_removed/(N_applied + residual_N + legume_N)

Where,
N_removed = As above
N_applied = As above
residual_N = As above
legume_N = The N credit (carryover) from mineralization of legume crop residues (lb N/acre)

An N credit from soybean cultivated before corn of 40 lb N/acre is typical in the Midwestern USA.[33] In rotations where legumes are cultivated, included N credit from the legume in the calculation of RTAs for the non-leguminous crop is more representative for the system. RTA_3 and RTA_4 are expected to be close to 1, whereas RTA_1 and RTA_2 might be larger than 1 if the producer accounted for the N credit from the soybean crop when choosing an appropriate N application rate. While RTAs close to 1 are desirable when all sources of N available to the plant are accounted for, it should be noted that RTAs larger than 1 can indicate mining of soil organic matter, implying a risk for loss of soil organic matter (Snyder et al. 2011, Jaynes and Karlen 2008). As this could occur without severe yield loss, at least not in the first years after the change in management, an upper limit for RTAs might have to be considered.
If RTAs were to be used as metrics in a performance standard test, it would entail selecting critical minimum RTA values, which eligible projects would have to exceed. Those critical values will depend on RTAs achieved under "business as usual" scenarios. Options for critical RTA values under various scenarios will be assessed in section 2.5. In addition, the use of RTAs in performance standard tests would require the calculation of RTAs for individual projects. This can be time consuming for the project developer or participant. Furthermore, crop yield necessary for determining the amount of N removed by harvest is unknown at the start of the

[33] For example: www.soils.wisc.edu/extension/wcmc/2005/pap/Bundy2.pdf and http://www.extension.umn.edu/cropenews/2004/04MNCN26.htm

project. Also residual N is unknown prior to the start of the project. This implies that a project's RTA, and consequently its eligibility, would only be calculated after the project has started. One way to get around this is by determining the project's expected RTA (and its eligibility) based on the proposed N application rates and the average amount of N removed by harvest for crop yields in the five years or cropping seasons observed in the relevant field prior to the start of the project. Similarly, residual N could be approximated by determining excess N based on the proposed N application rates and the average amount of N removed by harvest for crop yields in the five years or cropping seasons observed in the relevant field prior to the start of the project, corrected for leached N as discussed above.

2.5 N Use Data by State

This section lays out an assessment of available data to calculate N use and (partial) N balances. Furthermore, adoption rates and economic, social and technical barriers for each of the strategies that facilitate reducing N rates are summarized. Consistent with the MSU-EPRI protocol to quantify N_2O emission reduction from reducing N application rates used for corn in the NCR, the following states are included in our analysis of the data for the NCR: Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota, and Wisconsin. Note that the International Plant Nutrition Institute (IPNI) only considers Iowa, Illinois, Indiana, Minnesota, South Dakota, and Wisconsin to be included in the NCR. N use in California cropping systems is discussed based on Rosenstock et al. (2011). Their analysis includes N use data on the 33 most dominant crops in California.

2.5.1 Actual Versus Recommended N Rates

North Central Region

The most economically rational approach to determine the optimal N rate is the MRTN. The MRTN increases with decreasing fertilizer price and increasing corn price. Therefore, the ratio of the price of N fertilizer and the price of corn is used to calculate the MRTN. Figure 10 indicates corn prices and anhydrous ammonia fertilizer prices between 1999 and 2011, based on data from USDA surveys. We took into account the price difference between anhydrous ammonia and urea, the two major fertilizer sources for corn cropping systems in the NCR, by assuming that 50% of fertilizer use consists of urea and 50% consists of anhydrous ammonia. This is consistent with research conducted in Minnesota, which indicated that corn systems are fertilized equally with urea as with anhydrous ammonia (Bierman et al., 2011). However, the ratio of urea vs. anhydrous ammonia used is highly dependent on the soil type. The rate of increase in price for corn is similar to the rate of increase in price for fertilizer. As a consequence, there did not seem to be a consistent trend in the price ratio since 1999. The price ratio fluctuated between 0.07 and 0.014 and averaged at 0.10. This range in the price ratio is narrower than the range for which default tables were made in Sawyer et al. (2006), which is 0.05 to 0.20. This difference in price ratio ranges may be partially explained by the fact that the price ratio is partially dependent on the source of nitrogen. If anhydrous ammonia is the sole fertilizer source, price ratios since 1999 would range between 0.04 and 0.09. In contrast, exclusive use of the more expensive fertilizer urea would show price ratios between

0.09 and 0.18. It was not immediately clear what the fertilizer source assumptions were in Sawyer et al. (2006).

MRTN-based N recommendations for each of these price ratio points, as well as average N fertilization rates, were included in Table 5. The MRTNs were calculated based on the N rate calculator from Iowa State University,[34] which is also the basis for Sawyer et al. (2006). In Illinois, different MRTN values are available for North, Central and South Illinois. In Indiana, there is a distinction between West and Northwest, East and Central, and the remainder of the state. In Wisconsin, different MRTN values are available for fields with very high to high yield potential and medium to low yield potential, and fields on irrigated sands and non-irrigated sands. Note that, in contrast to Sawyer et al. (2006) our data is not aggregated over individual states. Table 6 also shows USDA reported actual fertilizer application rates at the state level, for comparison to the MRTN recommended rates. If actual N fertilizer application rates are greater than the MRTN recommended rates, this would suggest some potential for reducing N rates as a project action and the MRTN recommendations could be a useful proxy for indicating performance that goes beyond common practice.

[34] Available at http://extension.agron.iastate.edu/soilfertility/nrate.aspx

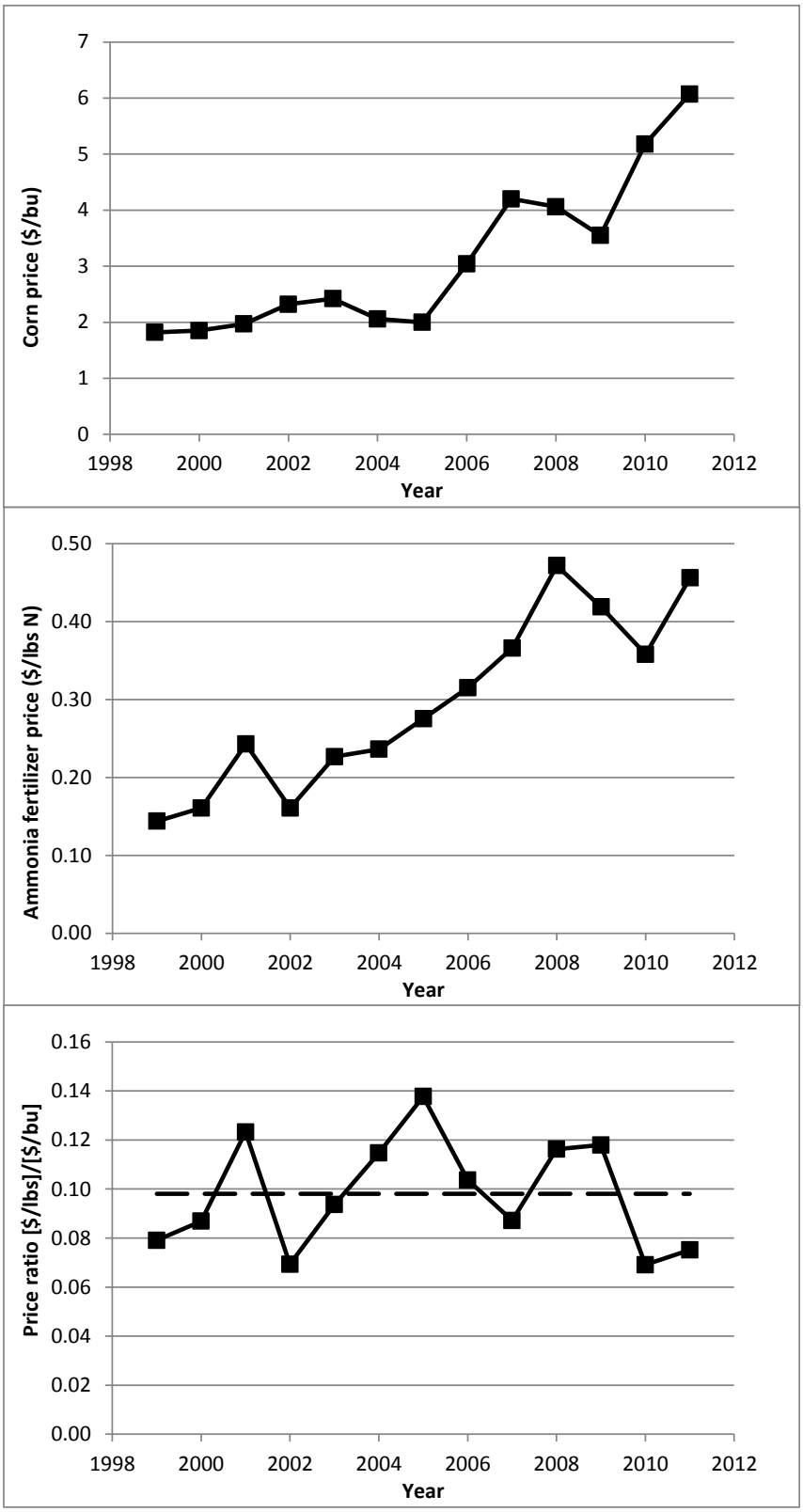

Figure 10. Corn price, fertilizer price, and resulting price ratio for the NCR. Data from NASS surveys.

Table 5. Actual and recommended N rates for corn the states in the NCR.

States	Actual Corn N fertilization rate [lbs N acre-1] 2005	2010	Region within state	recommended N rate - MRTNs at different price ratios [lbs N acre-1] Average price ratio (0.10) SC	CC	Low price ratio ~2010 (0.07) SC	CC	High price ratio ~2005 (0.14) SC	CC	Source of recom-mended rate
Illinois	164	187	North	145	185	157	201	132	167	(1)
			Central	168	185	183	200	152	169	
			South	172	188	190	205	155	171	
Indiana	165	200	West & North-west	169	NA	177	NA	156	NA	(1)
			East and Central	202	NA	214	NA	191	NA	
			Remainder	176	NA	189	NA	161	NA	
Iowa	158	159	State	133	190	145	199	120	176	(1)
Kansas	152	147	State	NA	NA	NA	NA	NA	NA	(2)
Michigan	143	137	State	131	NA	141	NA	122	NA	(1)
Minnesota	156	140	State	109	148	120	154	103	144	(1)
Missouri	179	141	State	NA	NA	NA	NA	NA	NA	(3)
Nebraska	155	157	State	NA	NA	NA	NA	NA	NA	(4)
North Dakota	136	179	State	NA	NA	NA	NA	NA	NA	(5)
Ohio	180	158	State	175	197	190	214	158	182	(1)
South Dakota	127	145	State	NA	NA	NA	NA	NA	NA	(6)
Wisconsin	120	103	VH/HYP	125	151	131	160	107	139	(1)
			M/LYP	94	109	107	118	89	94	
			Irr. Sands	209	209	209	209	197	197	
			non-irr. Sands	130	130	130	130	122	122	

Red cells indicate MRTN N rates that are greater than the actual corn N fertilization rate at a specific year

Green cells indicate MRTN N rates that are smaller than the actual corn fertilization rate at a specific year

With:

SC	Soy-corn rotation
CC	Continuous corn
NA	not available
VH/HYP	very high and high yield potential
M/LYP	medium to low yield potential
Irr.	irrigated
non-irr.	non-irrigated
(1)	Recommended N rates from http://extension.agron.iastate.edu/soilfertility/nrate.aspx using a MRTN for fertilizer-corn price ratio between 0.05 [$/lbs]/[$/bu] and 0.020 [$/lbs]/[$/bu]
(2)	Extreme ranges for varying yield goals and soil OM content, from http://www.oznet.ksu.edu/agronomy/soiltesting/
(3)	No N rate recommendations besides http://extension.missouri.edu/p/IPM1027#Fertilizer could be found
(4)	from Nebraska Lincoln extension fact sheet, extremes for high and low yield goal under different % of OM, assuming no residual N. source: http://cropwatch.unl.edu/web/soils/home/
(5)	for yield goal between 50 and 200 bu/acre, from http://www.ag.ndsu.edu/pubs/plantsci/soilfert/sf722w.htm
(6)	for silage corn with yield goal between 6 and 26 U.S. tons per acre, from http://pubstorage.sdstate.edu/AgBio_Publications/articles/EC750.pdf

When an average fertilizer-to-corn price ratio was employed to calculate the MRTN, there was no consistent relation between the recommended N rate and the actual N rate across states when MRTNs for corn following soybean cultivation are considered (Table 5). For continuous corn systems, the recommended MRTN rates were often times larger than the actual corn N

fertilization rates. It should be noted, however, that the actual corn N fertilization rates adopted from USDA/NASS and presented in Table 5 aggregate corn fields in soybean-corn and continuous corn rotations. Disaggregated data is not available. In 2010, when the price ratio was small, the actual N fertilization rate tended to be smaller than the recommended rates in most states and regions. In 2005, when the price ratio was large, the actual N rates were generally greater than the recommended N rates. A number of conclusions can be made. First, whether the actual N rate is above or below the recommended N rate depends greatly on the price ratio. While this comparison is dependent on the source of nitrogen that is assumed for corn cropping, similar trends were found when price ratios assumed anhydrous ammonia as dominant fertilizer source (data not shown).

Snyder et al. (2011) concluded that farmers in leading corn-producing states do not apply more N than would be prescribed for the maximum economic return to N (MRTN), based on the Corn N Rate Calculator. Second, actual N rates were generally smaller than the N recommendation, regardless of the price ratio, for Ohio and Wisconsin, and greater in Iowa, Michigan and Minnesota. Third, based on the years 2005 and 2010, there was no relation between the change in recommended N rates and the change in actual N rates: the N recommendation were greater in 2010 than in 2005, while the actual N rate was greater in 2010 than in 2005 for only 6 out of 12 states.

California Central Valley

In California, assessment of N use is severely limited by the lack of information with appropriate resolution (Rosenstock et al. 2011). This is especially problematic considering the large variety of crops, each of which have different N needs. A thorough review of N use in California indicated that faculty, farm advisors and facility managers (further referred to as “experts”) estimate that producers apply approximately 38 lbs N acre^{-1} more than growers report (Rosenstock et al. 2011). Experts’ opinion might be biased towards highly intensive cropping systems (Rosenstock et al. 2011). Alternatively, the discrepancy between experts’ opinion and grower reports might be an artifact of method and scale of data collection (Rosenstock et al. 2011). In any case, this observation warrants that the use of experts’ N rate recommendations as well as current information on state-averaged N application rates per crop type are likely unreliable metrics to use in a performance standard test.

2.5.2 N application rate versus N removed by harvest

Assuming 0.7 lb N removed per bushel (Fixen 2010), one can calculate the N removed by the corn harvest based on the average corn yields (Table 6).

Table 6. N removed by corn harvest and corn N fertilization rates for 12 states in the NCR.

	Corn yield (grain)		N removed by corn harvest (grain)		Actual corn N fertilization rate (averaged over grain and silage)		Actual - removed	
	[bushels per acre]		[lbs N acre-1]		[lbs N acre-1]		[lbs N acre-1]	
States	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**
Illinois	143	157	100	110	164	187	64	77
Indiana	154	157	108	110	165	200	57	90
Iowa	173	165	121	116	158	159	37	44
Kansas	135	125	95	88	152	147	58	59
Michigan	143	150	100	105	143	137	43	32
Minnesota	174	177	122	124	156	140	34	16
Missouri	111	123	78	86	179	141	102	55
Nebraska	154	166	108	116	155	157	47	41
North Dakota	129	132	90	92	136	179	45	87
Ohio	143	163	100	114	180	158	80	44
South Dakota	119	135	83	95	127	145	43	50
Wisconsin	148	162	104	113	120	103	16	-10
Average	***144***	***151***	***101***	***106***	***153***	***154***	***52***	***49***
	Corn yield (silage)		**N removed by corn**		**Actual corn N**		**Actual - removed**	
	[US tons per acre]		**[lbs N acre-1]**		**[lbs N acre-1]**		**[lbs N acre-1]**	
States	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**
Illinois	15.0	18.0	146	175	164	187	18	13
Indiana	20.0	21.0	194	204	165	200	-29	-4
Iowa	18.5	21.5	179	209	158	159	-21	-49
Kansas	16.0	14.0	155	136	152	147	-3	11
Michigan	17.5	18.5	170	179	143	137	-26	-43
Minnesota	16.0	20.0	155	194	156	140	1	-54
Missouri	13.0	15.0	126	146	179	141	53	-4
Nebraska	15.5	18.5	150	179	155	157	4	-23
North Dakota	11.0	14.0	107	136	136	179	29	44
Ohio	17.0	17.0	165	165	180	158	16	-7
South Dakota	11.0	13.5	107	131	127	145	20	14
Wisconsin	17.0	19.0	165	184	120	103	-45	-81
Average	***16***	***18***	***152***	***170***	***153***	***154***	***1***	***-15***

Shades of green are proportional to the value of the difference between actual and removed.

Table 6 indicates that more N is removed by silage than by corn for grain. The difference between the actual and removed was mostly positive for corn for grain and negative for corn for silage. In addition, this differed greatly among states and was not related to the average yields for that state. Illinois, Indiana, and North Dakota were among the states for which this difference was the greatest, while Wisconsin had the smallest difference.

2.5.3 Partial N Balances Using the Metrics Defined Above

In this section, we present RTAs reported by Fixen et al. (2010) and RTAs for corn in the NCR calculated based on N application rate data and corn yields as found in the USDA/NASS database and presented above.

Fixen et al. (2010) developed a preliminary Nutrient Use Geographic Information System (NuGIS) for the U.S., in which state-specific RTAs are reported, aggregated across crops (Table 7). In their analysis, RTAs were calculated as follows:

$$\mathbf{RTA_{INPI} = N\_removed/(N\_applied + N\_fixed)}$$

Where,
N_applied = The amount of synthetic and manure N applied (lb N/acre)
N_removed = Amount of N removed by harvest (lb N/acre)
N_fixed = Amount of N fixed during the growing season in case leguminous crops are cultivated (lb N/acre).

The results presented in Table 7 represent the average field in these states. No data are available to investigate the variability among fields. Considering potential losses of N due to leaching, runoff, and volatilization, the reported RTA_IPNI values do not indicate that over-fertilization is widespread on an average field in the NCR. A similar conclusion was made by Snyder (2011). Obviously, the presented approach is very coarse and the results are highly dependent on the nitrogen content of crops, which determines the nitrogen removed from a field. The nitrogen content of corn grain, for example, is known to be highly variable and dependent on cultivar, location, management, and grain yield (Boone et al., 1984). In addition, in case of corn-soybean rotation there may be an underestimation of the amount of fixed N during the soybean growing season, leading to an overestimation of the RTA. Furthermore, the N credit for corn following soybean is not accounted for in the RTAs reported by Fixen et al. (2010). Failing to account for N credit from a legume in the previous cropping cycle may lead to high estimated RTAs. In the survey conducted by Bierman et al. (2011), the average fertilization rate for corn on continuous corn systems (145 lbs/acre) was only slightly higher than for corn-soybean systems (140 lbs/acre), indicating that this N credit from the soy phase in the rotation does not lead to a significant decrease in fertilization rate. Overall, the use of data aggregated at the state level and across crops is limited within the context of developing of a performance standard test, and should only be interpreted as a rough indication.

Table 7: RTAs reported by Fixen et al. (2010) for the 12 states included in the NCR. RTAs were calculated across crops for the years 1987, 1992, 1997, 2002 and 2007. Green cells represent RTAs that pass the N rate criterion suggested by Ribaudo et al. (2011).

	RTA (across crops)				
	[dimensionless]				
States	**1987**	**1992**	**1997**	**2002**	**2007**
Illinois	0.73	0.77	0.78	0.87	0.87
Indiana	0.68	0.68	0.77	0.82	0.79
Iowa	0.74	0.71	0.8	0.86	0.8
Kansas	0.85	0.76	0.85	0.73	0.65
Michigan	0.66	0.7	0.68	0.76	0.77
Minnesota	0.76	0.74	0.8	0.86	0.85
Missouri	0.77	0.67	0.74	0.71	0.72
Nebraska	0.73	0.71	0.73	0.72	0.81
North Dakota	0.97	1.11	0.76	0.78	0.79
Ohio	0.71	0.79	0.82	0.75	0.82
South Dakota	1	1.1	1.16	0.88	0.84
Wisconsin	0.74	0.77	0.86	0.91	0.83
Average	***0.78***	***0.79***	***0.81***	***0.80***	***0.80***

Green cells indicate RTAs that meet the USDA criterion (i.e. RTA > 0.71). As noted before, RTAs higher Red cells indicate RTAs that are greater than 1 and are, therefore, at risk for mining soil organic matter.

We used average N application rates and corn yields per state in the NCR (USDA/NASS) to obtain corn specific RTAs for this region. RTA_1, RTA_2, RTA_3 and RTA_4 were calculated, which have increasingly stringent underlying mechanisms (Table 8). RTA_1 only accounts for applied fertilizer N, RTA_2 accounts for applied N and residual N, RTA_3 accounts for N credit from previously cultivated legumes and RTA_4 accounts for both legume N credit and residual N. The stringency in underlying mechanisms of the different RTAs is reflected by the number of observations for which the USDA criterion for best management practices is met, i.e., N application rates are such that excess N application does not exceed 40% of N removed by harvest, or the RTA is greater than 0.71. In other words, fewer acres pass this test when the calculation of excess N is more comprehensively estimated in accordance with the full N balance. This suggests that RTA_2, RTA_3 and RTA_4 could be useful in formulating a performance standard test. Nevertheless, less aggregated data are essential to explore this option more thoroughly. For example, N rates in continuous corn fields versus corn fields that are part of a soybean corn rotation would be needed to determine more truthful values for RTA_3 and RTA_4. Also a distribution of RTAs could be computed if paired yield and N rate data on a per field basis would be available.

Table 8: RTA_1, RTA_2, RTA_3 and RTA_4 for corn in the 12 states of the NCR.

	RTA_1 (corn grain)		**RTA_2 (corn grain)**		**RTA_3 (corn grain)**		**RTA_4 (corn grain)**	
	[dimensionless]		**[dimensionless]**		**[dimensionless]**		**[dimensionless]**	
States	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**
Illinois	0.61	0.59	0.48	0.46	0.49	0.48	0.40	0.39
Indiana	0.65	0.55	0.53	0.42	0.53	0.46	0.44	0.36
Iowa	0.77	0.73	0.66	0.61	0.61	0.58	0.54	0.50
Kansas	0.62	0.60	0.49	0.46	0.49	0.47	0.41	0.38
Michigan	0.70	0.77	0.58	0.66	0.55	0.59	0.47	0.53
Minnesota	0.78	0.88	0.68	0.82	0.62	0.69	0.55	0.65
Missouri	0.43	0.61	0.31	0.48	0.35	0.47	0.27	0.39
Nebraska	0.70	0.74	0.57	0.63	0.55	0.59	0.47	0.52
North Dakota	0.67	0.52	0.54	0.38	0.51	0.42	0.44	0.33
Ohio	0.55	0.72	0.42	0.60	0.45	0.58	0.36	0.50
South Dakota	0.66	0.65	0.53	0.53	0.50	0.51	0.42	0.43
Wisconsin	0.86	1.10	0.79	1.10	0.65	0.79	0.60	0.79
Average	***0.67***	***0.70***	***0.55***	***0.60***	***0.53***	***0.55***	***0.45***	***0.48***
	RTA_1 (corn silage)		**RTA_2 (corn silage)**		**RTA_3 (corn silage)**		**RTA_4 (corn silage)**	
	[dimensionless]		**[dimensionless]**		**[dimensionless]**		**[dimensionless]**	
States	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**	**2005**	**2010**
Illinois	0.89	0.93	0.84	0.68	0.71	0.77	0.59	0.62
Indiana	1.18	1.02	1.00	0.63	0.95	0.85	0.79	0.67
Iowa	1.14	1.31	1.13	0.83	0.91	1.05	0.80	0.91
Kansas	1.02	0.92	0.70	0.81	0.81	0.73	0.67	0.59
Michigan	1.18	1.31	1.03	0.90	0.92	1.01	0.79	0.90
Minnesota	1.00	1.38	1.08	1.03	0.79	1.08	0.71	1.01
Missouri	0.70	1.03	0.58	1.00	0.57	0.80	0.43	0.66
Nebraska	0.97	1.14	0.96	0.83	0.77	0.91	0.66	0.80
North Dakota	0.79	0.76	0.81	0.56	0.61	0.62	0.51	0.48
Ohio	0.91	1.04	0.70	0.96	0.75	0.83	0.60	0.72
South Dakota	0.84	0.91	0.83	0.71	0.64	0.71	0.54	0.60
Wisconsin	1.37	1.79	1.40	1.16	1.03	1.29	0.96	1.29
Average	***1.00***	***1.13***	***0.92***	***0.84***	***0.79***	***0.89***	***0.67***	***0.77***

Note: 40 lb N/acre was assumed for legume_N in the calculations of RTA_3 and RTA_4.
Green cells indicate RTAs that meet the USDA criterion (i.e., RTA > 0.71). As noted before, RTAs higher Red cells indicate RTAs that are greater than 1 and are, therefore, at risk for mining soil organic matter.

It is important to note that the approximation of residual N was rather challenging. We first calculated residual N as the difference between N applied and N removed, corrected for leached N (30% of applied N[35]) similarly to the calculation of RTA_2 when no measurement of residual N is available. This calculation returned negative residual N values (Table 9 option 1). The magnitude of those negative values is very unlikely, as residual N is often observed in the NCR. We suspect that the estimation of 30% leaching of applied N as outlined in the IPCC (2006) guidelines might be too high for this region. Alternatively, in a second approximation, we assumed that only the excess N (N applied – N removed) was subject to 30% leaching, leaving 70% of the excess N in the soil (See Table 9 option 2). As this approximation returned more realistic results, residual N data calculated following option 2 were used to calculate RTA_2 and RTA_4 in Table 8. Where residual N was negative, residual N was considered 0 in the calculation of RTA_2 and RTA_4. While the RTA values reported in Table 8 demonstrate the potential of more stringent RTAs to be used in a performance standard test, it is clear that the most appropriate determination of residual N used in those equations should be investigated more thoroughly. Perhaps, residual N data from soil test laboratories could be obtained.

Table 9: Residual N calculated following to suggested approximations

	Residual_N / option 1		Residual_N / option 2	
	(N_applied - N_removed) - 0.3* N_applied		0.7*(N_applied - N_removed)	
	[lb N/acre]		[lb N/acre]	
States	**2005**	**2010**	**2005**	**2010**
Illinois	14	21	44	54
Indiana	8	30	40	63
Iowa	-10	-4	26	31
Kansas	12	15	41	42
Michigan	0	-9	30	22
Minnesota	-13	-26	24	11
Missouri	48	13	71	39
Nebraska	0	-6	33	29
North Dakota	5	33	32	61
Ohio	26	-3	56	31
South Dakota	5	7	30	35
Wisconsin	-20	-41	11	-7

2.5.4 Adoption rates of N tests

Based on the survey results reported in the USDA ARMS N use report, it is clear that most farmers (~ 70%) are deciding on an N fertilization rate based on their historical practice (Table 10). Social factors including perceptions by landlords and neighbors, demonstrated success with a certain N rate, tradition, and comfort level of the producer and fertilizer supplier may play a role in determining the rate of N used by an individual producer (Sawyer et al., 2006). Up to 27% of farmers conduct a soil or N tissue test to inform their N fertilization rate decision. This

[35] following IPCC (2006) guidelines

percentage increased significantly (from 19% to 27%) between 2001 and 2005, however. In a different survey (Bierman et al., 2011[36]), conducted in spring 2010 in Minnesota, the percentage of fields on which soil testing was employed as a fertility management tool at least once in the previous 5 years was 84%. Large differences in the adoption of soil testing were noted. These differences were explained by differences in soil texture: coarsely textured soils have less ability to store mineral nitrogen, so that a residual nitrogen test is less relevant on such soils. It is not clear what explains the difference in adoption of soil testing between the USDA N Use report and the Bierman et al. (2011) study. The cost of nitrogen or the expected commodity price, the basis of the MRTN, influenced farmers' decision for N application rates only marginally (< 5%). This is corroborated by the lack of a consistent relation between actual N rates and MRTN (see previous). Interestingly, the fertilizer dealer influenced farmers' N rate decisions more than crop consultants or extension agents. It is likely that fertilizer dealers will recommend N rates based on yield goals and not the MRTN.

Even though no data were available to quantify the proportion of farmers using different discrete approaches, a likely order of prevalence of different approaches to decide N rates is: yield goal and routine practice > yield goal and routine practice + soil N test > MRTN > MRTN + soil N test > more elaborate approaches. Likewise, it is suggested in the MSU-EPRI protocol that yield-goal based N rates are the most common practice among farmers.

As most farmers are using routine practice to set their N fertilization rate, it would be sensible to set a performance standard based on historical practice. In addition, there is some potential for improving N efficiency by farmers by incentivizing the use of soil or tissue tests or more integrated approaches. In addition, residual N tests are fairly cost-effective. Testing for both nitrate and ammonia N does not cost much more than $15 per sample when a sample is delivered directly to a laboratory by a farmer. Michigan State University Extension Service recommends taking one sample every 20 acres that is composed of 15-20 cores within the 20 acres. More variable soils would require a more detailed sampling approach. As a consequence, the price per acre of soil mineral N testing is less than 1$. Obviously, this price does not take into account the time the farmer needs to sample the soil and transport the samples to the laboratory. Sampling and compositing soil samples could easily take one hour for every 20 acres, or two days for an average farm of 200 acres.

[36] Available at http://www.mda.state.mn.us/protecting/cleanwaterfund/~/media/Files/protecting/cwf/nfertilizersurvey2011.ashx

Table 10. Factors influencing farmers' nitrogen fertilizer application decision. Copied from USDA ARMS N Use report.)

Application used	2001	2005
	Percent of farmers	
Soil or tissue test	18.8	27.0*
Crop consultant recommendation	13	17.6*
Fertilizer dealer recommendation	28.7	41.2*
Extension service recommendation	3.2	4.6*
Cost of nitrogen and/or expected commodity price	11.4	17.3*
Routine practice	70.9	71.7*
	Number	
Observations	1,646	1,344

*Statically different from 2011 at the 1-percent level, based on pairwise two-tailed delete-a-group Jackknife t-test (Dubman, 2000)
Source: USDA, Economic Research Service using data from the USDA 2011 and 2005 Agriculture Resource Management Survey, Phase II, Cost of Production and Costs Report.

2.5.5 Data availability and Options for acquiring further data

There is very little non-aggregated or primary data available at a level below the state, let alone the field level. The following three options are available to acquire more detailed data: (1) use the existing ARMS dataset which requires meeting the conditions of a USDA data release, (2) informal survey of extension agents in some states, and (3) development of a statistically valid survey.

If we were to use the ARMS data and request a data release, we envision that the following questions could be answered.

- What is the distribution of N rates and yields within a county, state, etc.?
- Are N rates and yields significantly affected by the factors influencing farmers' nitrogen fertilizer application decision?
- What is the distribution of RTAs within a county or state?
- Are the RTAs significantly affected by factors influencing farmers' nitrogen fertilizer application decision?

Note that the most appropriate N rate can vary depending on field properties. If a stringent performance standard threshold were to be used, meaning that excellence in nitrogen management is incentivized, answers to the above questions are essential to select a scientifically sound threshold value. Knowledge on the distribution of N application rates, yields, and RTAs within states, counties, groups of farmers influenced by diverse decision tools, etc. is important to select performance standard thresholds that take into account variability in properties that can affect productivity and N cycling as observed in farmers' fields.

The disadvantage is that access to the ARMS data is quite restrictive and there is no guarantee that the data can be updated in time and according to the frequency required by the protocol. In addition, only very few crops and regions are included in the survey. Note that a similar survey has been conducted by the NASS, the National Resources Inventory - Conservation Effects Assessment Project (NRI-CEAP). This survey included the application of commercial fertilizers (rate, timing, method, and form) for crops grown the previous 3 years and was conducted from 2003 through 2006. The reporting of the data is still in draft format as noted in "Assessment of the Effects of Conservation Practices on Cultivated Cropland in the Upper Mississippi River Basin" by the NRCS. It was unclear when the final report will be published and if a release for primary data is available.

Alternatively, the Reserve can conduct an informal survey among extension agents in some states and for some crops. Extension agents could be asked one or more of the following questions:

- What is the percentage of farmers that uses a soil N test?
- What is the spread/range in N fertilization rates among farmers?
- What do fertilizer dealers base their recommendations on?

Such an informal survey would be cost-effective option, but would be largely qualitative and may not have rigorous statistical validity.

The third option is to develop a statistically valid survey and conduct the survey among farmers in key regions and for key cropping systems. This option is likely the most expensive, but also provides the most flexibility.

Lastly, some soil laboratories may have compiled their results and the results may be made available for a fee.

2.6 Options for Performance Standards for Reducing N Rate in NCR and CCV

We present a number of options for setting the common practice of nitrogen management.

The simplest approach is to set the performance threshold based on absolute N application rates, which could be calibrated by using state averages of N application rates derived from USDA ARMS data. Since these data include standard errors, it may be possible to integrate the variability of N application rates across fields within a state and define thresholds based on the percentile of farmers that exceeds the threshold under common practice. The N application rates would have to be adjusted for different crops, obviously. This could happen proportionally to the average N demand of a crop. The merit of this approach is simplicity. Using absolute N rates as a threshold disregards site-specific characteristics that impact the N efficiency, such as texture and the N removed by harvest.

Second, the performance threshold can be set as the historical average for a specific field over the past ~5 growing seasons. Any N application rate below the field-specific historical average is considered additional. Requiring historical data may be challenging for some producers. Therefore, it would be optimal to provide an alternative requirement that is not based on a

historical average. One of the two other criteria (RTA-based, and back-calculated from average county yields) outlined below could be added as alternatives to the historical average.

Third, the performance threshold can be set based on RTA_1 such that the N rate should not exceed the amount of N removed by harvest plus a fixed value, such as 40% as used in the USDA BMP, equivalent to a N efficiency ratio that is greater than 70%. The RTA approach is based on a better representation of the nitrogen cycle in the field, and is, therefore more scientifically valid than a threshold based on absolute N rates, at the expense of more complexity. The exact threshold value for RTA_1 can be either based on a frequency analysis so that only a small percentage of the producers would pass the additionality test based on their current baseline practices, and/or could be based on scientific arguments. The ARMS USDA report indicated that only 35% of corn-planted acres of land did not meet the criterion when the cutoff was set to 40% excess N (in other words, 65% do meet this criterion already). The 40% threshold leads to a relatively high baseline adoption rate (65%), and should probably be made more stringent by increasing the threshold RTA to 80% or 90%, depending on a curve such as the one illustrated in Figure 11 (fictional data). More complete RTA values will represent the N cycle better at the expense of complexity. For example, it would be very sensible to have the RTA metric take into account residual nitrogen from a soil test as explained above.

Using state-level data, we compared RTAs to the criterion set in the recently published N use report by USDA (Ribaudo et al., 2011), where the N rate was considered consistent with BMP if excess N was less than 40% of N removed by harvest, which is equivalent to an RTA greater than 0.71, as explained above. RTAs that met the USDA criterion with RTAs greater than 0.71 showed up as a green cell in Table 7 and Table 8. As noted before, RTAs reflect the N use efficiency. Therefore, RTAs greater than 1 involve a risk for mining of soil organic matter. Therefore, RTAs larger than 1 are highlighted in red in Table 7 and Table 8. In about half of the states, the USDA criterion is met in 2010 for corn for grain. In all states the criterion is met in 2010 for corn for silage.

The fourth option to set a performance standard is based on yield goal (i.e. consistent with MSU-EPRI when no historical yields are available) and back-calculate fertilizer rates for average county yields. In the case study presented in Annex C in MSU-EPRI, the yield-goal based recommend N rate was 174 kg N/ha. The question remains how much smaller it should be so that yields are not impacted. In addition, as noted above in section 2.2, fertilizer recommendations are not necessarily a good indication of actual practice. In addition, there is a lack of sufficient county-specific data to back-calculate N application rates. Back-calculating fertilizer rates based on yields is challenging since there is not always a clear and direct relation between fertilizer rates and yields due to the presence of residual nitrogen in a soil, climate variability, and variability in other soil parameters. However, as is pointed out by the authors of the MSU-EPRI methodology, this approach remains conservative since, likely, the baseline N application rates are underestimated. Since differences in yields among counties are likely caused by a complex array of factors, including soil type and management, it seems that there is not much information added compared to using an average N application rate across counties by back-calculating fertilizer use based on yields even if this approach is conservative.

Finally, it is possible to combine two or more of the options above in a hybrid approach. For example, it could be required that project N application rates not only go below historic rates but also meet a level of performance that is "significantly better than average." The question then becomes on how to set the threshold for better than average, and if the added complexity justifies the increase in environmental integrity.

As an illustration of the difference between option 1 and option 2, we conducted the tests in option 1 and option 2 for the case study contained in the MSU-EPRI protocol in Annex C. The yield of the case study was 156.1 bushel/acre, so assuming an N content of 0.7 lbs N/bushel, 140 lbs N are removed per acre. The N rate criterion of 40% higher than harvested N would be 196 lbs N/acre. The threshold N rate used in the MSU-EPRI example could be above or below harvested N plus 40%, granted that the yield goal is achieved.

It is clear that the presented options to set the exact performance standard threshold need a substantial amount of data. As a simple illustration of what would be possible if a dataset of 50 fields with yields, residual soil N and N fertilizer rates would be available, we created a dataset of 50 fields through simulation. Figure 11 indicates the distribution of farmers across the range of RTA_1 and RTA_2. The figure should be interpreted as following. Twenty percent of the farmers have an RTA_1 greater than 100% and an RTA_2 greater than 80%. RTA_1 is always greater than RTA_2.

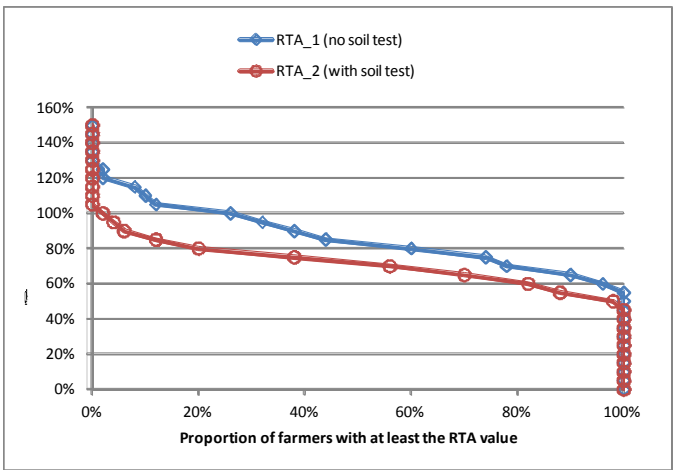

Figure 11. Distribution of farmers according to two removed-to-available (RTA) ratios defined above. Distribution is based on simulated data and includes 50 fields.

2.7 Conclusions on Performance Standards for Reducing N Rate

Surveys indicate that farmers determine application rates mostly by "routine practice," which may be informed by yield goal-based approaches. Therefore, the MSU-EPRI protocol requires back-calculation of N rates based on yield-goals and county average yields as a proxy for BAU N application rates in case historic N application rates are not available. The MRTN is a more economically sensible approach to determine the N rate compared to yield-goal, but there was

no consistent relation between MRTN and actual application rates, at least at a state level. As a consequence, the MRTN may not be an ideal proxy for BAU application rates and would, therefore, not be an ideal metric for a performance standard used to ensure projects do better than average or BAU.

Since most farmers indicate that "routine practice" determines their N application rates, it makes sense to express a performance standard relative to routine practice. In other words, if a project it is doing better than routine practice, they will exceed the performance standard.

Still, a significant amount of farmers use more innovative approaches than routine practice or pure yield-goal approaches. Such more innovative approaches do include the MRTN, but also soil nitrate tests, etc. to determining N application rates. Therefore, in addition to a performance standard expressed relative to the routine practice, there seems to be potential for using metrics that are based on N balances (removed-to-available, RTA) to determine whether projects go beyond common practice in case routine practice is already relatively small.

The USDA developed a BMP criterion stating fertilizer should not be applied greater than 40% above the N removed by harvest. It was not immediately clear what the physical basis was for the value of the threshold. In a recent survey, it was determined that 65% of all farmers meet the BMP criterion. It would be ideal to include a measurement of the residual nitrogen at the beginning of the growing season to calculate the excess N. Surveys indicate that soil N tests, while fairly common, are not used annually by the majority of farmers. Without detailed field-level data available, it is not known what the percentage of fields is that would meet the residual-nitrogen corrected excess N criterion. State-level data suggests that most areas would not meet the USDA BMP criteria if the calculation of excess N accounted more completely for the full N cycle. One of the RTA metrics would help to reflect the full N cycle more fully.

2.8 Ecosystem Service Stacking

Ecosystem service stacking refers to situations where credits or incentive payments for actions taken on a single parcel of land are established to provide more than one ecosystem service, which in some cases may lead to issuance of multiple "credits" for a single action. The main consideration for developing the nitrogen management protocol is how additionality of an offset project that reduces N_2O emissions through nitrogen management is affected when the project may also receive incentive payments for adopting nitrogen or other nutrient management practices and/or sell water quality credits for reducing nitrogen leaching/run-off. Options for addressing ecosystem service stacking will need to be further considered during the protocol development process. Additional background research will be prepared in support of such considerations.

3 GHG Sources, Sinks, and Reservoirs

The Reserve commissioned a background paper, developed in the context of the Cropland Management Protocol, which comprehensively describes the sources, sinks and reservoirs (SSRs) relevant to agriculture projects. Most of the SSRs related to cropland management will

also be SSRs for nitrogen management. Therefore, this section elaborates on the SSRs that are different for NMPP, direct and indirect N_2O emissions.

3.1 Overview of all GHG Sources, Sinks, and Reservoirs Related to Nitrogen Management

A substantial number of on-site and off-site (upstream and downstream) emissions sources, sinks and reservoirs (SSRs) are potentially affected by project activities that are implemented to reduce N_2O emissions. Nitrous oxide emissions are usually separated into direct emissions and indirect emissions. Direct emissions originate directly from the soil system to which the N fertilizer was added through either nitrification or denitrification, while indirect emissions are produced off–site as a result of the application of fertilizer N (e.g., beyond the field site to which fertilizer N has been applied). Figure 12 provides an overview of all direct and indirect emissions related to nitrogen management. As indicated in the introduction of this section, most of the indirect emissions outlined in Figure 12 are discussed at length in the cropland management background paper commissioned by the Reserve. Therefore, we will limit the discussion to the following sources of indirect N_2O emissions: (1) NO_3 leached into groundwater and, subsequently, denitrified to N_2O, (2) NO_3 lost through run-off from fields and, subsequently, denitrified to N_2O, and (3) mineral nitrogen converted to NH_3 and NOx through volatilization and, subsequently, deposited onto aquatic and soil surfaces and converted to N_2O.

The uncertainty related to direct N_2O emissions is well described. However, the uncertainty in indirect N_2O emissions from crop and livestock production is even greater than for direct emissions (Mosier et al., 1998; Groffman et al., 2002). Not only are indirect N_2O emissions dependent on the amount of nitrogen leachate, run-off, or volatilization, but also on the denitrification conditions of the soil, sediment, and aquatic environment where the nitrogen ends up. To add even more uncertainty, nitrogen gas fluxes from streams and groundwater are currently poorly constrained (Fennel et al., 2009).

Note that emissions related to the production or transportation of fertilizer may be affected by projects that reduce nitrogen use; however these impacts may be omitted from the accounting of GHG emission reductions as this is conservative (the omission could only lead to an underestimation of GHG reductions).

Up-stream	On-site	Off-stream
• Transport of labor to farm (if labor is increased) • Emissions during production of lime and N fertilizer • Emissions from transportation of lime, inorganic N or manure to the field • Emissions from transportation of seed for cover crops • GHG emissions from storing manure or any other off-site manure management	• Reduction in N_2O emissions from soil ("direct emissions") • Changes in soil carbon content • Changes in emissions from machinery use from N fertilizer application • Changes in emissions from machinery use from planting and incorporation of cover crops	• GHG emissions from indirect land-use changes • Leaching and run-off of applied nitrogen, followed by denitrification into N_2O • Volatilization of applied nitrogen to NH3 and NOx, followed by deposition onto aquatic and soil surfaces and conversion to N_2O

Figure 12. General Illustration of the up-stream, on-site, and down-stream SSRs related to nitrogen management.

3.2 Defining GHG Accounting Boundary

3.2.1 Indirect GHG Sources and Quantification for Existing Protocols

The accounting for emissions from run-off is often wrapped together with the accounting of emissions from leaching, even though they are completely separate phenomena.

Source	ACR	MSU-EPRI	Alberta
Leaching	Included. Quantification through IPCC Tier 1 default emission factors based on the leaching amount simulated by DNDC.	Included. Quantification through IPCC Tier 1 default emission factors based on total annual fertilizer.	Included. Quantification through emission factors based on total annual fertilizer, but calibrated for different eco-regions in Canada.
Run-off	Wrapped within the leaching quantification.	Wrapped within the leaching quantification.	Unclear, most likely wrapped within the leaching quantification.
Volatilization	Included. Quantification through IPCC Tier 1 default emission factors based on the leaching amount simulated by DNDC.	Included. Quantification through IPCC Tier 1 default emission factors based on total annual fertilizer.	Included. Quantification through emission factors based on total annual fertilizer, but calibrated for different eco-regions in Canada.

3.2.2 Analysis of the Gaps in Existing Approaches to Define GHG Accounting

All three existing protocols, which are reviewed in section 4 of this paper, include the three sources of indirect N_2O emissions. As pointed out before, the quantification of indirect N_2O emissions is very challenging, in part because leaching, run-off and volatilization heavily depend on soil properties, weather conditions as well as the mode and rate of fertilizer application. For regions where the DNDC model has been calibrated rigorously for estimating N leaching, run-off and volatilization, the ACR methodology is likely more sensitive to effects of N management and environmental conditions on N losses, leading to a potentially more accurate quantification of indirect N_2O emissions. However, regions where the DNDC model has been successfully calibrated for predicting leaching and volatilization are limited, mostly due to the lack of appropriate data. In the MSU-EPRI and Alberta protocols, leaching and volatilization are a function of total N applied. While scientifically-based default factors underpinning the estimation of leaching and volatilization within those protocols are expectedly robust, they are insensitive to potential effects of environmental conditions and N management besides N application rate. Within the Alberta protocol, emission factors are specific for different eco-regions, allowing for a potentially more accurate quantification of indirect N_2O emissions than the MSU-EPRI protocol.

3.2.3 Conditions Under Which SSRs Can Be Excluded Because They Are Insignificant or Doing So Is Conservative

The MSU-EPRI methodology assumes that when the total rainfall and irrigation during the growing season is smaller than the potential evapotranspiration during the growing season, indirect emissions from leaching and runoff are insignificant. Definitions of a growing season are included in the methodology. However, as with denitrification, leaching is an event-driven process. The amount of nitrogen leached is strongly correlated with the intensity of a rainfall

event (Sugita and Nakane, 2007). Even when the total potential evapotranspiration during the growing season remains greater than the total rainfall and irrigation during the growing season, the daily potential evapotranspiration may be smaller than the daily rainfall and irrigation during at least some days. Therefore, there may be circumstances under which the test to omit indirect emissions through leaching will yield a conservative estimate of emissions. On the condition that sufficient data for calibration and validation of a biogeochemical process model are available and such a model can be properly calibrated, leaching can be simulated based on actual rainfall and irrigation schedules, potentially leading to greater accuracy of indirect N_2O emission estimates.

4 Review and Comparison of the GHG Accounting in Existing Methodologies

Three programs have developed distinct, comprehensive and relevant nitrogen management offset protocols for North America:

- N_2O Emission Reduction through Changes in Fertilizer Management (American Carbon Registry)
- Quantifying N_2O Emissions Reductions in U.S. Agricultural Crops through N Fertilizer Rate Reduction, (MSU-EPRI methodology,[37] in second assessment phase with the Verified Carbon Standard)
- Quantification Protocol for Agricultural Nitrous Oxide Emissions Reductions (Alberta Offsets Program)

The Reserve summarized the main components of these three protocols in a separate methodology analysis papers that compare the main architecture, leakage, additionality, and GHG Boundaries of the three protocols.[38] This background paper focuses on comparing the quantification methodology of these methodologies with the goal of informing the quantification approach of the nitrogen management protocol. An additional forthcoming paper on quantification approaches will build off of this quantification methodology comparison by comparing the accuracy and uncertainty associated with various quantification approaches for N_2O emission reductions from cropland.[39]

4.1 Challenges of Measuring Nitrogen Gas Fluxes

Most commonly, N_2O emissions are measured in the field using static flux chambers. Gas sampling from simple but effective chambers in the field is relatively straightforward, and a

[37] Methodology developed by researchers at Michigan State University (MSU) and the Electric Power Research Institute (EPRI)

[38] The Reserve's "Background Paper: Methodology Synthesis for Nitrogen Management" is available at: http://www.climateactionreserve.org/how/protocols/agriculture/nitrogen-management/

[39] This forthcoming Quantification Approaches paper will be available in early 2012 at: http://www.climateactionreserve.org/how/protocols/agriculture/nitrogen-management/.

relatively small investment in equipment is required. Chamber techniques are used in an ever-expanding number of research studies.

Issues associated with sampling and analysis are well known and documented, and readily overcome. Gas chromatography for analysis of N_2O is a well-established technology and costs of machinery are continually decreasing alongside continually increasing analytical sophistication. One of the major drawbacks of determining annual N_2O emissions using field measurements is the large time-commitment and associated cost for personnel. The time and workload required for field measurements of N_2O emissions increases exponentially with more management practices, cropping systems, geographic locations, and combinations of those under investigation. Another difficulty associated with field measurements stems from the high spatial and temporal variability associated with N_2O emissions (e.g., Mathieu et al. 2006, Lee et al. 2009). Nitrogen gas fluxes may change drastically within one field reflecting very local changes in porosity and water-filled pore space. Such local changes in porosity and water-filled pore space can originate from naturally occurring changes in soil type. Similarly, N_2O fluxes are primarily event-related, meaning that a great proportion of the annual emissions are emitted immediately after individual precipitation or irrigation events, especially after fertilization or soil cultivation.

Models can bridge some of the challenges associated with measuring N_2O fluxes. Biogeochemical process models were designed to quantify annual biogenic gas fluxes based on detailed input data in a way that is arguably faster and more cost-effective than measuring than N_2O gas fluxes. When calibrated well and used under the correct circumstances, models can estimate N_2O fluxes in a sufficiently accurate and cost-effective manner for a variety of environmental conditions and agricultural management practices. Costs for N_2O quantification using biogeochemical models are mainly associated with technical expertise required to operate these complex models and collection of field data to calibrate and validate the models. Field measurements remain the benchmark quantification method and all model predictions must be validated using empirically field measurements.

Models are usually divided into (1) biogeochemical process models, which simulate the processes driving emissions, and (2) statistical or empirical models, which statistically interpolate field measurements as a function of basic input variables. There is no generally accepted approach to modeling N gas flux. Empirically derived regional emissions factors, as used in both the MSU-EPRI and Alberta methodologies, combine field-based measurements and straightforward statistical models, in which nitrous oxide emissions are assumed to be only dependent on the N application rate.

Each model requires a different set of input data. The right model for a specific application will depend on the available data and the required accuracy. Models that receive more input data will not necessarily yield more accurate results. Due to over-parameterization, many examples exist in which more simple models perform better than more complex models. One effective way to avoid over-parameterization is by using different data to calibrate the model than to validate a model, and only comparing models using data that was not used during model calibration. Note that an emission factor is a specific case of a statistical model, where N_2O emissions are calculated as a factor of total N applied. The IPCC has suggested that N_2O

emissions from crop production systems can be quantified based on an emission factor of 1% of total N applied (Mg N_2O–N (Mg N input)$^{-1}$), with an uncertainty range of 0.3 to 3% (2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4, Chapter 11).

The three broad approaches to quantify N_2O emission reductions: direct flux measurements, biogeochemical process models, and emission factors, represent different trade-off between cost and accuracy. While the use of emission factors is extremely straightforward and transparent, it is often unknown how accurate they are under varying soil properties, timing and placement of N fertilizer, tillage type, or weather conditions. When global emission factors (i.e., Tier-1) are used, particularly at a local or even field scale, the (relative) uncertainty around the resulting N_2O emissions can be substantially large, so that the use of a single global average emission factor is inadequate (Berdanier and Conant 2011). Regional emission factors (i.e., Tier-2), developed for a specific region and cropping system, may yield substantially more accurate results than global emission factors when used at local or field scale. Biogeochemical process models require expert knowledge in costly input data but, if well calibrated based on field measurements, can increase the accuracy of GHG emissions significantly and take into account impacts of weather and soils. Direct field measurements to quantify baseline and project N_2O emissions would require skilled personnel to both operate and to analyze the resulting data, as well as appropriate equipment, but will have the greatest accuracy. The three quantification approaches in existing protocols use either global or regional emission factors or a biogeochemical process model. Note that field N_2O measurements are required for the development of regional emission factors as well as the calibration of biogeochemical process models.

4.2 General Evaluation of Process Models and Emission Factors

As a prelude to the review of the existing GHG protocols, we will discuss general challenges and advantages of process models and emission factors first.

4.2.1 Evaluation of Biogeochemical Process Models

Biogeochemical process models have the ability to account for specific crop characteristics, climate/weather conditions, and soil traits when calculating GHG emissions. However, biogeochemical process models are far from perfect and only include a simplification of all processes involved in SOM dynamics. In particular, accurately characterizing the soil moisture of the vadose zone (the unsaturated root zone in the soil profile) remains an unsolved problem in the earth sciences. So, while it is mechanistically understood that the aerobic state of the soil is the main driver of N_2O gas fluxes, it is challenging to model N_2O gas fluxes since biogeochemical process models are not great in modeling the soil moisture state in terms of both spatial and temporal scales (Groffman et al. 2009). Therefore, it is not surprising that process models do not model daily N_2O fluxes very well. While both the DNDC and DAYCENT models have been shown to simulate the timing of peak and low N_2O flux values fairly well (Kariyapperuma et al. 2011), the models are not able to reliably capture the magnitude of peak events. For example, Del Grosso et al. (2002) reports a weak relation (r^2 values rarely exceeding 25%) between simulated and measured daily N_2O fluxes within one site. A similar weak relation between simulated and measured daily N_2O fluxes was reported in Li et al. (2005) when using

the DNDC model (r^2 value ranging from 0.14 and 0.35). Del Grosso et al. (2002) posed that effects of (micro-)topography, aspect, wind, humidity, microsite heterogeneity, gas diffusion, and other factors on soil water and temperature are not included in general-purpose biogeochemical process models, such as DNDC or DAYCENT, but are likely important on a daily basis and may explain the reported deviation between observed and simulated values at a daily time scale. However, when aggregated at an annual time scale, total modeled N_2O fluxes do show a much better correspondence with measured fluxes. For example, Del Grosso et al. (2005) reports an r^2 of 0.74 based on simulations at 12 different sites across North America. The inherent uncertainty in the biogeochemical process model is referred to as the structural uncertainty, as opposed to the uncertainty introduced by imperfect input values.

An additional challenge is that biogeochemical process models are notoriously difficult to calibrate. Results can be biased if one single variable is not parameterized well. Due to the large number of input variables, it is not straightforward to verify a simulation run. A recent comparison of N_2 and N_2O fluxes simulated by various widely applied biogeochemical models demonstrated broad disagreement across the models tested (David et al., 2009), raising questions about how to correctly parameterize process models. In conclusion, running process-based models accurately and consistently requires a certain level of sophistication and expertise. In order to avoid bias and unreasonable costs, standardization would be needed in order to make process-based models available to non-expert users (project developers, verifiers), who would use the models to set baselines and quantify N_2O emission reductions for individual projects.

4.2.2 Evaluation of Emission Factors

The IPCC tier 1 factor was developed specifically for GHG inventories at a broad geographical scale such as individual countries. To simplify, the emission factor represents an average across different crops, soils, climate and management, all of which are known to affect nitrification–denitrification and N_2O production and emission (Mosier et al., 1998). The IPCC tier 1 emission factor is not tested to be used to calculate emission reductions due to decreases in fertilizer. Errors around estimates of emission reductions will be further inflated by subtracting two results (e.g. project minus baseline GHG emissions) of the emission factor approach. Errors are reduced when multiple fields are included in the quantification. A comparison on quantification approaches developed by the Reserve contains an analysis of the reduction in uncertainty when multiple fields are included in the quantification and a certain level of correlation is assumed between the emissions at project and baseline N addition levels.

An important difference between emission factors and biogeochemical process models relates to assumptions regarding nitrogen cycling. Emission factors assume that there is no carry-over effect from one year to the next and that all nitrogen added to cropland during one season cycles completely during that season and contributes to N_2O emission for only that year. Emission factors do not allow for residual N to be stored in soil or biomass and contribute to N_2O emission in subsequent years. In contrast, biogeochemical models do incorporate soil organic nitrogen storage, even though simulation results indicate that this does not always lead to an improvement of simulation accuracy. Nevertheless, most biogeochemical models assume that the majority of N added due to management will be taken up by the crop, lost as N gas, or lost as nitrate. While this can be problematic in years where crop growth is simulated poorly, biogeochemical models are generally capable of simulating the impacts of previous management. This results from their ability to take into account (1) the effect of management practices that increase SOM mineralization and therefore may increase N_2O emissions, e.g., intensive cultivation, or summer fallows, and (2) the exact weather.

An additional drawback of the emission factors is that they likely underestimate indirect N_2O emissions from nitrogen-fixing crops. Emission factors assume that only nitrogen from synthetic fertilizer and organic matter amendments are susceptible to leaching. However, some authors have reported a significant amount of nitrate leaching on unfertilized years of a cropping rotation during which soybeans were grown (Jaynes et al., 2001, Di and Cameron, 2002). Nitrogen immobilized in the soil N pool can be stored in organic form for many years until its mineralized, reabsorbed by plants, or lost from the plant/soil system via leaching or N gas emission (Follett, 2001; Paul and Clark, 1996). In soybean-corn rotations, residual nitrate from corn cultivation can be leached through planting of soybean the following year and beyond, and should be accounted for when quantifying indirect N_2O emissions from such cropping systems. More research is necessary, however, to fully understand the dynamics of leaching. Current biogeochemical models have not been rigorously tested for their ability to reproduce soil solute properties. This is mostly caused by a lack of availability of lysimeter data to calibrate the models and the high spatial and temporal variability in solute dynamics. Better spatial and temporal quantification of nitrate leaching in tile-drained systems facilitated the simulation of seasonal dynamics of nitrate loss in those systems. Even though process models do not perform very well in predicting nitrate leaching, they do take into account storage in soil N pools across years, making process models arguably more accurate than emission factors.

In conclusion, there is not one single quantification approach available that is cost-effective, simple, and accurate for all soils, climatic conditions, and cropping systems at the field level. There are many potential benefits and challenges to using either emission factors or process model. Regional emission factors and regionally calibrated process models are good candidates for quantification approaches in the NMPP on the condition that their validity can be demonstrated for the regions and cropping systems they are intended to be used for. It is expected that the accuracy of various quantification approaches will increase in the near future with the availability of more field measurements. Therefore, it is recommended to allow sufficient flexibility in the NMPP.

4.3 Review & Comparison of Existing GHG Accounting Methodologies

This section reviews each of the three protocols according to a standard set of criteria, including accuracy, cost, ease of use, and transparency. Both the MSU-EPRI and Alberta protocols use emission factors. The ACR protocol is the only one that applies a biogeochemical process model to estimate N_2O emissions for the baseline and project (DNDC). The general advantages and drawbacks of using emission factors vs. biogeochemical models are discussed in the previous section. This section provides only a high-level but comprehensive review of the three protocols.

ACR protocol	
Accuracy	**Medium**. The ACR protocol uses the DNDC biogeochemical process model. When calibrated well, the DNDC model can be relatively accurate due to the ability to directly account for the many real-time complexities and inter-related dynamics of GHG emissions from agricultural soils. Most notably, the actual weather and dates of management practices are used to calculate N_2O emissions. However, the uncertainty around modeled results may still be significant due to structural errors in biogeochemical process models. The ACR protocol only requires procedures to calibrate the optimal yield parameter and parameters related to heat or water stress. For parameters related to C:N ratios and nutrient allocation of the different plant compartments, the crop-specific default values must be used. However, it is unknown what the validity is of the latter parameters for some less widespread crops. In addition, specific crop varieties may have significantly different characteristics compared to more generic default values within the model. For example, many crops have short-season varietals that mature faster and need less thermal degree days to mature. The protocol could be improved by specifying how the latter parameters could be calibrated as well. The approach to account for uncertainty from variability in input factors is valid in principle. There are four aspects that could be improved to the approach, however. First, it is assumed that no correlation exists among input parameters, potentially overestimating the uncertainty. Second, the approach requires fixed probability distributions when values from databases, such as the SSURGO database, are used. There is almost no background data available to substantiate these distributions and the uncertainty around the SSURGO data. It is known that the SSURGO database contains significant inaccuracies. Third, the approach assumes that the model's structural uncertainty is insignificant when at least 5 fields are included within one project. However, background data lacks to substantiate that the structural uncertainty is insignificant when 5 fields are included. In addition, it remains unclear how a field is defined. The approach could be improved by (1) allowing correlation between input factors, if sufficient data are available, (2) the use of databases such as SSURGO should be made more restrictive or conservative by increasing the inherent uncertainty of the probability distribution, or including some field check of the SSURGO data, (3) adding an additional deduction factor to represent the model's structural uncertainty and which is calibrated using empirical field data, and (4) adding specific requirements for model validation against field flux measurements per crop type for various geographic scales.
Cost	**Medium-high**. Data inputs for biogeochemical process models are very elaborate. Specific inputs related to amount, timing, application method, type, and rate of synthetic fertilizer application, as well as separate inputs for organic fertilizer application details must be supplied. There is a strong need for proper and standardized validation and calibration procedures to achieve accurate estimates and homogeneity across model runs. Cost will be a higher if an external consultant must be hired to run DNDC.
Ease of use	**Low**. Working with a biogeochemical process model requires experience. Detailed procedures must be developed to exactly explain how each variable is selected. Furthermore, standardization of soil parameter inputs is a large barrier for model

	implementation.
Transparency	**Low**. As discussed before, biogeochemical process models are difficult to calibrate and existing model runs are challenging to verify. Additionally, the ACR protocol allows a significant amount of flexibility in selecting data sources that can be used for each of the model inputs and model parameters. For example, soil parameters can be either directly measured or looked up in the SSURGO soil database. The procedures to account for the uncertainty are different based on whether the parameter was measured or looked up in an existing database. This degree of flexibility makes the transparency and verifiability of credits challenging. For example, for field measurements, the outcome of the procedure will depend on how many and where field samples are taken. Likewise, in cases where values are looked up from the existing database, some ambiguity exists on how to aggregate values if a field spans multiple soil types. In addition, this flexibility could potentially be resource intensive for a project to determine which variables should be default values, measured on-site, and so on. This is an area where a more rigorous Reserve protocol could improve upon.
Completeness	**High**. Process models have a very complete GHG accounting because they describe all processes leading to volatilization and leaching and integrate soil carbon dynamics as well as nitrogen cycling.

MSU-EPRI protocol	
Accuracy	**Medium**. The MSU-EPRI protocol allows two emission factors: the standard tier-1 (linear) IPCC emission factor can be used if the cropland is within the United States or a specific (exponential) emission factor for corn systems on cropland within the North Central Region (NCR) of the USA. The emission factors take only annual nitrogen application rate as an input. The use of the IPCC tier 1 emission factor for quantifying field-level emission reductions is not perceived as accurate due to reasons indicated in the previous section. Even though the use of this factor may be conservative in many cases, as claimed by the authors of the MSU-EPRI methodology, there is very little data to exactly substantiate this claim and understand in which cases the use of this factor is not conservative (e.g., organic soils, extreme weather events,…). MSU-EPRI's regional NCR emission factor, derived from a 3-year field study in Michigan, conducted by Michigan State University (MSU) is without doubt a notable improvement over the IPCC global default emission factor. The results of the field study and the principle of the accounting procedure were published in Millar et al. (2010). Despite the more rigorous approach compared to the tier-1 emission factor, the NCR emission factor remains limited in geographic scope, applicable cropping system (only corn), and applicable management practices (only practices that are associated with a reduction in fertilizer). The claim that the results from the Michigan field studies are broadly applicable to corn systems throughout the North Central Region (NCR)[40] has not been well documented. Millar et al. (2010), the peer-reviewed article supporting the MSU-EPRI protocol, reports: *Nevertheless, these N gradient datasets are geographically limited and further studies of N_2O emissions responses from multiple N gradients throughout the Midwest would provide greater confidence for applying the non-linear relationship Hoben et al. (2010, in review) documented for Michigan.* Whatever emission factor is used, however, it is clear that N_2O emission varies temporally and spatially and that any emission factor used would have uncertainty of at least 50% (Mosier et al., 1999; Lim et al., 1999). Millar et al. (2010), clearly indicates that there is simply a lack of empirical data to draw broadly applicable and consistent conclusions on the impact timing of N application, fertilizer N placement or fertilizer N type. It is not debated that these factors may have a significant impact under certain circumstances or weather conditions. It is well documented that annual variations in N_2O emission due to weather are often greater than management-induced variations (Clayton et al., 1997; Kaiser et al., 1998). In addition, emission factors likely underestimate indirect N_2O emissions from

[40] Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota and Wisconsin

	nitrogen-fixing crops, as explained before in Section 4.2.
Cost	**Low**. The simplicity of an emission factor makes its use extremely cheap.
Ease of use	**High**. Emission factors do not require specialized experience.
Transparency	**High**. Only a very limited set of input variables are needed. Emission factors are easy to verify.
Completeness	**High**. Procedures for all indirect emissions are included.

Alberta protocol	
Accuracy	**Medium**. The protocol uses an emission factor specifically calibrated for Canadian eco-regions. The factor for the Alberta protocol takes into account the ratio of precipitation and irrigation to potential evapotranspiration and is therefore perceived as more comprehensive than the MSU-EPRI emission factors, which are purely based on annual nitrogen application rate. The factors were developed within the context of Canada's National Inventory Report quantification method using N_2O flux measurements in three regions with different precipitation characteristics.[41] Note that the Alberta approach is based on reducing GHG emissions relative to the crop yield. Therefore, offsets can be created under the protocol when project activities lead to an increase in absolute GHG emissions but crop yields increase proportionally even further.
Cost	**Low**. The simplicity of an emission factor makes its use extremely cheap.
Ease of use	**High**. Emission factors do not require specialized experience.
Transparency	**High**. Only a very limited set of input variables are needed. Emission factors are easy to verify.
Completeness	**High**. Procedures for all indirect emissions are included.

Even though COMET-VR is not a GHG accounting protocol, but rather a GHG quantification tool, we include a short review of COMET-VR below. COMET-VR is a graphical web-based front-end for the DAYCENT model already parameterized for many relevant management options, crops, and soil types. It fills a gap between a detailed biogeochemical process model, and a quick and easy-to-use emission factor by reducing the complexity of the model parameterization to a number of straightforward steps within a graphical user interface.

[41] See Canada's National Inventory Report Part 2 available at http://www.ec.gc.ca/Publications/492D914C-2EAB-47AB-A045-C62B2CDACC29%5CNationalInventoryReport19902008GreenhouseGasSourcesAndSinksInCanadaPart2.pdf

COMET-VR and COMET-FARM	
Accuracy	**Medium**. The COMET-VR tool simplifies the use of a biogeochemical process model by limiting the model parameters that can be entered. Only a limited set of management schedules, crop parameters and soil parameters emission are included. This has the advantage of simplifying the use of the model, at the expense of flexibility in the input parameters. However, the authors of the COMET-VR tool went through great lengths to focus on the most important parameters and included all major management practices and crops. Therefore, only for fields where more explicit input data is available (measured soil texture, soil organic carbon, or the exact dates of management decision), will using the original DAYCENT model have an advantage over the tools.
Cost	**Low**. The required input data should be readily available for a producer. The simplicity of the graphical user interface makes the use of these calculation tools cheap. However, some time must be spent in going through all the screens and filling out all necessary data.
Ease of use	**High**. Calculation tools do not require specialized experience to parameterize.
Transparency	**High**. A much more limited set of input variables are required, compared to the original DAYCENT model. The tools are pre-populated with a set of input parameters for different types of management and crops. Therefore, the results provided by the tools are easily verifiable. However, currently, the system does not allow for saving model runs or exporting a model run to an auditor for verification. However, we suspect that this would be fairly straightforward feature which can be implemented if demand for this feature is sufficient.
Completeness	**High**. Most GGH SSRs are included.

4.4 Conclusion and Recommendations for Quantification Approaches

Tier-1 emission factors are useful tools for GHG inventories at large geographic scales, but are often inadequate to predict emissions at a local or even field scale. Direct flux measurements are too costly to be used as the sole quantification method within the context of any carbon protocol. Since protocols developed by the Reserve generally tend to be based on methods that have the highest accuracy within reasonable costs, tier-2 emission factors and biogeochemical process models are recommended as valid candidates for quantification approaches. Tier-2 emission factors can be a valuable quantification approach if they are derived from data collected in cropping systems and at geographic locations representative for the project. In addition, the conditions of their use in terms of allowable cropping systems, regions, soil types, and/or agronomic management must be clearly specified and they must be properly discounted for uncertainty, preferably using field measurements that were not used for the development of the emission factor. Similarly, biogeochemical process models should only be used when properly calibrated with field measurements for the conditions under which they are intended to be used and when the uncertainty can be quantified using independent field measurements. As with emission factors, uncertainty discounting can be a powerful mechanism to ensure the use of biogeochemical process models remains conservative. If biogeochemical process models

were to be included as a quantification approach, unambiguous and strict procedures will be necessary to ensure the correct and objective use of such a model.

Undoubtedly, many more field measurements will become available in the near future. These new data will be the basis of new tier-2 emission factors as well as procedures to employ biogeochemical process models in certain cropping types and regions. Therefore, it is recommended to allow a modular and sufficiently flexible approach to emissions quantification in the NMPP. As noted above, a forthcoming quantification approaches paper from the Reserve compares global and regional emission factors with field measurements for two cropping systems. [42]

5 Review of Risk Potential, Magnitude and Quantification of Leakage

Leakage is defined as a secondary effect leading to changes in GHG emissions caused by a shift in cultivation activities from inside of the project area to outside of the project area, as a result of the project activities. Within the context of agricultural production, the most likely cause for such a shift in cultivation activities is a decrease in the production (yield) of an agricultural commodity as a result of implementing the project activity. This decrease in production could cause leakage in a direct manner, meaning with project proponents, or an indirect manner. For example, if project activities lead to a decrease in yield, a grower could increase production in a non-project area to compensate. Indirectly, if lower production and therefore lower supply of a commodity from the project areas increases the price of the commodity within a region expanding beyond the project area, a grower located outside of the project area may increase production of the commodity. While direct production shifts can – to some extent – be evaluated by monitoring the lands under management of the project proponents (both inside and outside of the project area), indirect production shifts that are caused by a change in production on an individual field are very challenging to quantify. Economic models exist to quantify shifts at a macro level, however.

Project activities can affect agricultural yield in various ways. Yield can be increased, in which case the project activities provide an additional benefit beyond N_2O emissions reductions. In this case there may be even positive leakage, in which the project activities lead to decreases in emissions due to an increased supply of the commodity. Then, N_2O emissions reductions estimated for the project activity will remain conservative when changes in emissions from leakage are omitted. Project activities that have no significant effect on yield also hold no leakage risk. Whenever reduced yield is observed on fields where project activities are implemented, it is important to determine if the decrease in yield is causally related to the project activity or is a result of inherent variability caused by weather, pests, changes in management unrelated to project activities, etc. Only when the project activity effectively causes the yield loss, should the magnitude of leakage and its effects on net N_2O emission reductions be deducted from the overall emission reductions. The next question then becomes

[42] This forthcoming Quantification Approaches paper will be available in early 2012 at: http://www.climateactionreserve.org/how/protocols/agriculture/nitrogen-management/.

whether the GHG intensity of production on other lands in greater, less than, or equal to the GHG intensity on the project land.

One important determinant of the risk of leakage is the crop acreage elasticity of the commodity, the impact of an increase in price for the commodity on the acreage of the commodity. Leakage risk is assumed to increase with increased crop acreage elasticity of the commodity. An added complexity is that a decrease in production of one commodity may lead to an increase in production of another commodity, which is referred to as the cross-price elasticity. In this section, we discuss the risk and potential for leakage according to four main questions:

(1) What is the risk and magnitude of yield loss for the various project activities that are considered?
(2) What is the risk that a change in yield leads to a production shift, using information on crop acreage elasticity?
(3) How can it be determined whether an observed change in yield is causally related to project activities or just the effect of inherent variability in weather and management?
(4) What is the GHG intensity of production on the other lands and how can one quantify the emissions from activity-shifting leakage?

5.1 Effects of Selected Project Activities on Yield and Leakage Risk

5.1.1 Reducing N Rate

As noted in section 2.2 in this background paper, low N prices relative to crop prices have encouraged some farmers to apply N beyond the strict N crop requirements as a form of risk insurance for seasonal variability in yield associated with variability in weather conditions, soil characteristics or pest damage. There is no indication that this practice is still widespread, however (Snyder et al., 2011). However, under some conditions, there may be some potential for achieving N_2O emissions reductions by decreasing N application rates without decreasing yield or profit. However, there is a risk for leakage if the N application rate is decreased below a critical level where yield is significantly jeopardized. In general, two mechanisms for reductions in N application rates have been recommended: (1) Switching from applying the Agronomic Optimal N Rate (AONR) to applying the amount of N that will maximize profits, and (2) basing the N application rate on soil N or plant tissue N tests.

Recommended N rates based on maximizing profits have been referred to as economic optimum N rate (EONR) and Maximum return to N (MRTN), depending on the state where the idea is promoted. For example, an EONR is suggested by Nebraska Lincoln extension services,[43] while MRTN has been developed for Iowa, Illinois, Indiana, Michigan, Minnesota, Ohio and Wisconsin[44]. Those approaches have the recommended N application rate be dependent on the

[43] http://cropwatch.unl.edu/web/soils/home/
[44] http://extension.agron.iastate.edu/soilfertility/nrate.aspx

price of N, the forecasted price of the crop and the crop yield response to N addition. While the MRTN rate is the average recommended N rate, an economically Profitable N Rate Range (PNRR) was defined as the range of N rates resulting in a net return to N within ± US$ 2.45 ha^{-1} of the MRTN (Sawyer et al. 2006). In general, profits are sustained without significant yield loss as long as the N rate is not decreased below the lower limit of the PNRR. Hence, N rates that are within the PNRR are assumed to hold no risk for leakage. As N rates below the lower limit of the PNRR explicitly suggest a risk for yield loss, it is very unlikely that a producer will deliberately reduce N application rates below the lower limit of the PNRR in the absence of other financial intensives.

Now, it is pertinent to assess the risk that a farmer will jeopardize yield by reducing N application rates below optimal levels if financial intensives such as C-credits are in place. In other words, if the producer can make more money by saving on fertilizer costs and selling carbon than he/she could by selling actual crops, he/she might choose to reduce crop production. In Hoben et al. (2011), yield and N_2O responses to N rates in corn cropping systems in Michigan and the PNRR for the region in which the research was conducted are presented. Given average N and corn prices for the period 2006-2009 from USDA NASS,[45] the risk of leakage due to reduction of N rates in the presence of C financing with variable C prices can be assessed as follows:

If the following condition holds, the potential for leakage is real:

$$\Delta_{N2O} * C\_price + \Delta_{N\_rate} * N\_price - \Delta_{yield} * crop\_price > 0,$$

Where,

Δ_{N2O} = The emission reduction beyond the lower limit PNRR N application rate, calculated as the difference in N_2O emissions between the lower limit PNRR N application rate and the project

Δ_{N_rate} = The difference between N rate for the lower limit PNRR and the project

Δ_{yield} = The change in yield beyond the lower limit PNRR N application rate, calculated as the difference in the yield at an N rate representing the lower limit PNRR versus the project

In other words, there is a risk for leakage if the combined profits from N_2O emission reductions and savings in N application are larger than the economic value of the lost yield. For corn cropping systems studied by Hoben et al. (2011), yield loss and associated loss in profits caused by decreasing N rates below the lower limit PNRR were high compared to the savings made by reduced N application and N_2O emissions reductions (Figure 13). Even at a price of $60 per tCO_2-eq, there was a loss in profit when N application rate was reduced below the lower limit PNRR. This suggests that the risk that producers would reduce N rates to a point where yield is significantly affected in the presence of a C market is negligible in corn cropping systems in Michigan, and presumably, therefore, in the whole North Central Region.

[45] http://quickstats.nass.usda.gov/

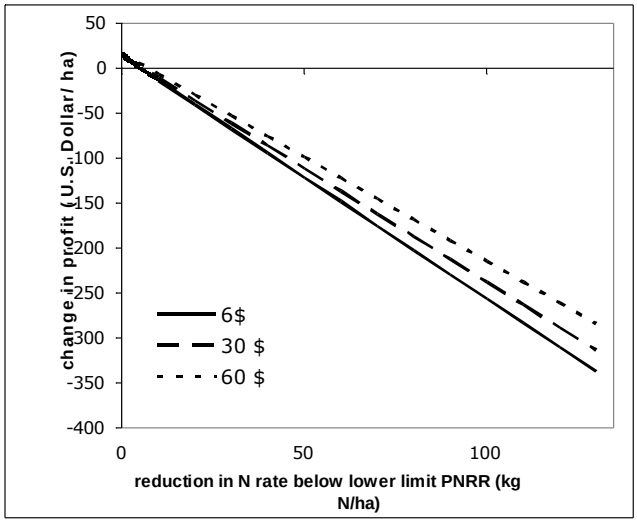

Figure 13. Change in profit from yield and carbon by decreasing N rates below the lower limit PNRR for a field in Michigan and different price points of carbon. (Yield and N_2O data underlying trends adopted from Hoben et al. 2011, prices for N fertilizer and corn adopted from USDA NASS.)

Soil N or plant tissue N quantification tests have shown to sustain yields with significantly lower N application rates, because N fertilizer needs can be predicted more accurately (Schröder et al. 2000). Nevertheless, it is important to note that N rate recommendations based on these tests rely on correlations between soil N, plant tissue N, fertilizer rates, and crop yields, which require reliable field calibration data and data collection methods (Colorado State University Extension[46]). Calibrating and correlating these several inputs is highly specific to the specific circumstances. To avoid calibration uncertainty, reliable recommendations can only be made for selected crops, environmental conditions and regions where calibrations were optimized (e.g. University of Minnesota Extension,[47] Figure 14). Thus, under the condition that adequately calibrated N recommendation tests are available, N application rate reductions based on soil N or plant tissue N quantification tests will result in N savings and N_2O emissions reductions without yield loss and consequently no risk for leakage.

[46] http://www.ext.colostate.edu/pubs/crops/00501.html
[47] http://www.extension.umn.edu/distribution/cropsystems/dc6514.html

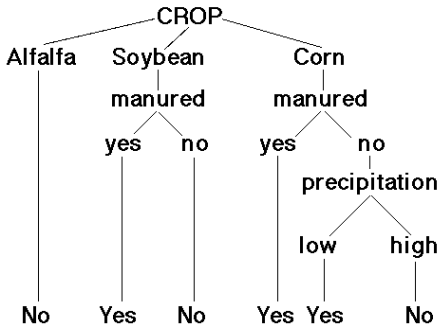

Figure 14. Flow chart decision-aid for determining the probability of having significant residual N in the soil, resulting in justified use of soil N based fertilizer N recommendations. (Adopted from University of Minnesota extension.)

In conclusion, it is highly unlikely that producers will deliberately reduce N applications rates to a point where yields are jeopardized. Reducing N rates based on N recommendations that optimize profit (e.g., EONR and MRTN) as well as adjusting N application rates guided by results from a soil N test are appropriate options to reduce N_2O emissions without risk for leakage. These strategies, however, are limited to regions where the relevant extension materials are available.

5.1.2 Switching from Fall to Spring N Application

Research has shown higher corn yields with spring fertilizer N application compared to fall application (Ohio State University Extension,[48] Shapiro et al., 2008). This is mostly related to the risk of N loss during the winter when N is applied in the fall, leaving less N left for plant growth in the spring and summer. Moreover, because of the avoided loss of N during the winter when switching from fall to spring application, it is recommended that N application rates be reduced by 5% of the fall total when applied in the spring (Shapiro et al., 2008). Nevertheless, if conditions are very wet in spring, the producer might miss his/her window of opportunity to apply N at this time of the year, causing significant negative yield effects if no fertilizer was applied in fall. Consequently, a positive yield effect of switching from fall to spring N application suggests no risk for leakage in years with relatively dry springs, while yield loss can be dramatic if wet conditions in spring were to prohibit fertilizer application.

5.1.3 Applying N Closer to the Root

Research over a period of 15 years demonstrated that subsurface drip irrigation in combination with fertigation increases yields in various crops in the San Joaquin Central Valley and Imperial Valley in California (Ayars et al. 1999) including tomato, cotton, sweet corn, alfalfa, and cantaloupe. These higher yields under subsurface drip irrigation and fertigation result from placement of the fertilizer closer to the root of the crops and timing of fertilizer application in

[48] http://ohioline.osu.edu/e2567/

accordance with the crop's nutrient needs. As subsurface drip irrigation and fertigation generally increase crop yields, there is no risk for leakage associated with this project activity.

5.1.4 Use of Nitrification and Urease Inhibitors

By preventing the conversion of NH_4^+ to a more mobile and loss-prone form of N, i.e. NO_3^-, nitrification inhibitors conserve fertilizer nitrogen in the root zone where it may be utilized by the crop, eventually increasing crop yield (Wolt, 2004). However, nitrification inhibitors can also negatively affect yield by altering the availability of NH_4^+ versus NO_3^- as crop N source and by inducing phytotoxicity (Prasad and Power, 1995). Prevention or retardation of nitrification results in increased NH_4^+:NO_3^- ratios in soil. While plants can metabolize NH_4^+ more efficiently than NO_3^-, NH_4^+ is toxic to all plants at high concentrations. It was observed that the growth of most upland crop plants is best when both NH_4^+ and NO_3^- forms of N are available, but optimal NH_4^+:NO_3^- ratios vary with species, cultivar and age of the plant (Prasad and Power 1995). Phytotoxicity is dependent on the crop type and nitrification inhibitor used. A suite of nitrification inhibitors is commercially available, including dicyandiamide (DCD), nitrapyrin, Ca-carbide, 3,4-dimethyl pyrazole phosphate (DMPP), thiosulfate and neem. In addition to the cropping system and nitrification inhibitor type, the efficiency of nitrification inhibitors generally depends on temperature, soil texture and soil organic matter content (Subbarao et al. 2006).

A meta-evaluation of the agronomic and environmental effectiveness of the nitrification inhibitor, nitrapyrin in field corn, wheat, grain sorghum, and sweet corn cropping systems in the Midwestern USA indicated that nitrapyrin can increase crop yield by 7% (Wolt, 2004). However, increases, decreases and no effects of nitrification inhibitors on crop yield have also been observed (Prasad and Power, 1995). Based on an extensive literature review, Prasad and Power (1995) suggested that yield benefits of nitrification inhibitors in corn production systems are limited to coarse textured soils and in situations where excessive soil water leads to heavy N leaching. Yield effects of nitrification inhibitors on sorghum were limited. Wheat yields responded positively to nitrification inhibitors in the Pacific Northwest and the southern Midwest, but not the southeastern states and the western Midwest. Increases, as well as decreases, in potato yield due to nitrification inhibitors were common, while cotton showed phytotoxicity symptoms after DCD application. Based on the variable responses of yield to the use of various nitrification inhibitors, a risk of leakage exists. Therefore, it is recommended to evaluate yield effects for individual projects using nitrification inhibitors. Furthermore, it is important that project developers consult extension agencies and manufacturers' guidelines in order to make well-informed choices of nitrification inhibitors and application rates appropriate for the cropping system and environmental conditions under consideration.

5.1.5 Changing Fertilizer Composition from Anhydrous Ammonia to Urea

Various experiments have shown sustained yields when switching from anhydrous ammonia to urea as a fertilizer N source (University of Minnesota Extension[49]). However, yield penalties can

[49] http://www.extension.umn.edu/distribution/cropsystems/dc0636.html

occur following inappropriate timing and placement of urea application, mostly as a result of the volatile and readily hydrolysable nature of urea, increasing the risk of unintended N and subsequent yield loss (Camberato 2009). Volatilization can be prevented by incorporating urea in the soil during, or shortly after, application to the field. Incorporation can be accomplished by a tillage operation, by blending urea into the soil with irrigation water, or by knife injection of the fertilizer (Scharf and Lory 2006). Urea can be treated with a volatilization inhibitor before broadcasting. Fall application of urea will cause significant yield loss, and is therefore, discouraged. Furthermore, seed-placed urea can be toxic to seeds of various crops, even at low concentrations (Minnesota extension). Therefore it is recommended that urea is side-placed rather than seed-placed. In summary, the use of urea instead of anhydrous ammonia does not imply leakage risk, granted appropriate timing and placement of the fertilizer.

5.1.6 Using Slow or Controlled Release Fertilizer

Slow or controlled release fertilizers are intended to increase nutrient use efficiency and decrease the risk of N loss to the environment by matching the timing of plant-available N with crop N needs. Synthetic slow-release fertilizers operate on one of the following strategies (Chien et al., 2009): (1) The slow release fertilizer is formed by condensation products of urea and urea aldehydes, of which the most significant types on the market are urea formaldehyde (UF), isobutylidene diurea (IBDU), and crotonylidene diurea (CDU). Nitrogen slowly becomes plant-available as the fertilizer is mineralized; there are complex factors such as solubility of polymer–N, soil temperature, moisture, microbial activity, texture, organic matter, etc., that can affect the mineralization rate and agronomic effectiveness of polymer– N; (2) Alternatively, slow release fertilizers can be comprised of coated or encapsulated fertilizers, such as S-coated urea (SCU) or polymer-coated urea (PCU). The release pattern of these fertilizers is usually related to the coating composition and thickness and can be affected by soil moisture and temperature. As slow release fertilizers will only benefit yield if release patterns match plant N needs, it can be expected that complex effects of environmental conditions on release patterns for both types of slow release fertilizers result in varying effects on crop yield.

Controlled-release coated urea products enhanced grain yield and N uptake compared to regular fertilizers in rice in Spain (Carreres et al., 2003), winter wheat in China (Fan et al., 2004), peanuts in Japan (Wen et al., 2001), potatoes in the USA (Munoz et al. 2005), and maize in Japan (Shoji et al., 2001). However, urea coated with a semi-permeable polymer had no effect on corn yields in Michigan compared to 3 other traditional N sources (Michigan State University Extension[50]). Similarly, a coated urea slow release fertilizer and a polymer-N slow release fertilizer did not affect grain yields in corn and winter wheat cropping systems in North Carolina (Cahill et al., 2010). The same was observed for vegetables, where slow release N fertilizers as a pre-plant treatment did not decrease crop yield, but yield was rarely increased when compared with standard split applications of soluble N (Guertal, 2009). In conclusion, yield is mostly unresponsive to slow release N fertilizers, therefore risk for leakage due to decreased crop yield is unlikely.

[50] http://news.msue.msu.edu/news/article/slow_release_nitrogen_fertilizers

5.1.7 Adding N Scavenging Crops

While replacing bare fallow periods with non-leguminous cover crops could improve N retention of post-harvest surplus inorganic N, there is a wide-spread perception that cover crops would severely reduce cash crop yields (Tonitto et al. 2006). Nevertheless, meta-analysis of 69 yield studies and 92 soil studies indicated that yields under non-legume cover crop management are not significantly different from those in the conventional, bare fallow systems, while leaching is reduced by 70% on average (Tonitto et al. 2006). This implies that leakage risk is minimal when adding N scavenging crops to a rotation.

5.2 Elasticity of Crop Acreage to Fluctuating Crop Demand and Prices

It is evident that farmers' expectations of future commodity prices shape their decisions of which crops to plant on available area. Nerlove (1956) created a model that related the desired acreage of a crop as a function of one-year lagged crop price and one and two-year lagged crop acreages (Braulke, 1982). More recent studies have improved upon this model by adding factors related to a farmer's expected utility maximization behavior. The most common quantity used to express the dependency of crop acreage with price is the crop acreage elasticity. A crop acreage elasticity of 0.25 indicates that, when a certain percent increase in the price of the crop (relative to input cost) occurs, this price increase results in an increase in crop acreage that is equivalent to 0.25 times the price increase. In other words, the higher the crop acreage elasticity, the greater the increase in acreage in response to a price increase for a specific crop. In addition, there can be further impacts on the acreage of other crops and the acreage of non-farming land going into crop production. Therefore, three different categories of crop acreage elasticity are distinguished:

(1) elasticity of the crop acreage in response to changes in the price of that crop
(2) elasticity of the crop acreage in response to changes in the price of a different crop ("cross-price elasticity")
(3) elasticity of bringing non-agricultural land into agricultural production ("land transformation") in response to changes in the price of a specific crop.

An overview of the elasticity and cross-price elasticity for some key crops is provided in the following table. The elasticity of corn ranges between 0.05 and 0.95, wheat between 0.05 and 0.35, and soybean between 0.25 and 0.95. The value of 0.95 for the elasticity of corn and soybeans was reported in one study by Miller and Plantinga (1999). Huang and Khanna (2010) have the most sophisticated model with the most data. As a consequence, we have the greatest confidence in their elasticities, ranging from 0.067 for wheat to 0.510 for corn. Most cross-price elasticities are negative, and can therefore be conservatively omitted from leakage calculations, except for a curiously positive cross-price elasticity for wheat in response to the price of corn. Given the complexity of the econometric models that were used to estimates of the elasticity, it is likely that all estimates are associated with a substantial amount of uncertainty. The impact of this uncertainty on the increase in emissions due to leakage will depend on the magnitude of the decrease in leakage and the GHG intensity of the baseline management.

Table 11. Overview of crop acreage elasticity reported by Huang and Kanna (2010) for changes in prices of different crops.

Study	Crop	Own-price elasticity	Cross-price elasticity
Chavas and Holt (1990)	Corn	0.15	-0.15 (Soy)
	Soy	0.45	-0.30 (Corn)
Chembezi and Womack (1992)	Corn	0.1	-0.05 (Soy)
			-0.05 (Wheat)
	Wheat	0.05	-0.05 (Corn)
			-0.10 (Soy)
Lee and Helmberger (1985)	Corn	0.05	-0.15 (Soy)
	Soy	0.25	-0.15 (Corn)
Lin and Dismukes (2007)	Corn	0.17-0.35	-
	Soy	0.3	-
	Wheat	0.25-0.34	-
Miller and Plantinga (1999)	Corn	0.95	-0.45 (Soy)
	Soy	0.95	-0.40 (Corn)
Morzuch et al. (1980)	Wheat	0.35	-
Orazem and Miranowski (1994)	Corn	0.05	0.00 (Soy)
	Soy	0.25	0.00 (Corn)
Tegene et al. (1988)	Corn	0.2	-
Huang and Khanna (2010)	Corn	0.51	-0.118 (Soy)
			-0.345 (Wheat)
	Soy	0.487	-0.295 (Corn)
			0.00 (Wheat)
	Wheat	0.067	0.306 (Corn)
			0.054 (Soy)

As indicated above, an increase in price for a commodity may lead to an increase in uncultivated land being brought into cultivation. Even though the probability of this occurring remains fairly small, the emissions from bringing uncultivated land into cultivation are potentially high, especially in cases of grasslands or forestlands being brought into cultivation. Unfortunately, very little data are available to quantify the elasticity of land transformation, which is the elasticity related to bringing uncultivated land into cultivation. In a study investigating the land-use change impacts of soy biodiesel production, the California Air Resources Board used a range in land transformation elasticity of 0.1 to 0.3 (ARB, 2009). It is likely that yields on land that is taken into production will remain small since most productive land has been taken into production already, as pointed out by ARB (2009).

5.3 Evaluating the Cause of the Yield Effects

The most straightforward way to evaluate whether a yield is anomalous or outlying is by first determining the likely distribution of normal yields according to past observations of yields and

using a simple statistical test to verify if a specific yield is anomalous when compared to the normal distribution. In investigating farm-level historical yield series, Pease et al. (1992) concluded that for most farms, the historical yield series followed a normal distribution. Some authors, however, noted that non-normality is not uncommon due to right or left skewness (e.g., Ramirez, 1997). However, it seems that most authors agree that in most circumstances a normal distribution is approximating real yield distributions satisfactorily. As a consequence of the near-normal distribution of yields on a single field, it is tempting to use a simple statistical t-test to check whether a future yield is anomalous compared to past yields.

However, using a simple t-test to check future yields is fundamentally flawed and proves to be an approach with very little statistical power. Even without project activities, yields fluctuate annually as a function of climate, management factors, or pests and weeds (Figure 15). In addition, crop yields are expected to increase over time because of technological progress such as the adoption of new varieties, optimization of management practices, increases in application rates, and efficiencies of fertilizers and irrigation. In empirical studies, technological progress is usually modeled by a linear or quadratic function with time (e.g., McCarl et al., 2008). A linear increase of crop production over time can be observed for soybean yields in Mason County, Michigan (Figure 15 – bottom panel).

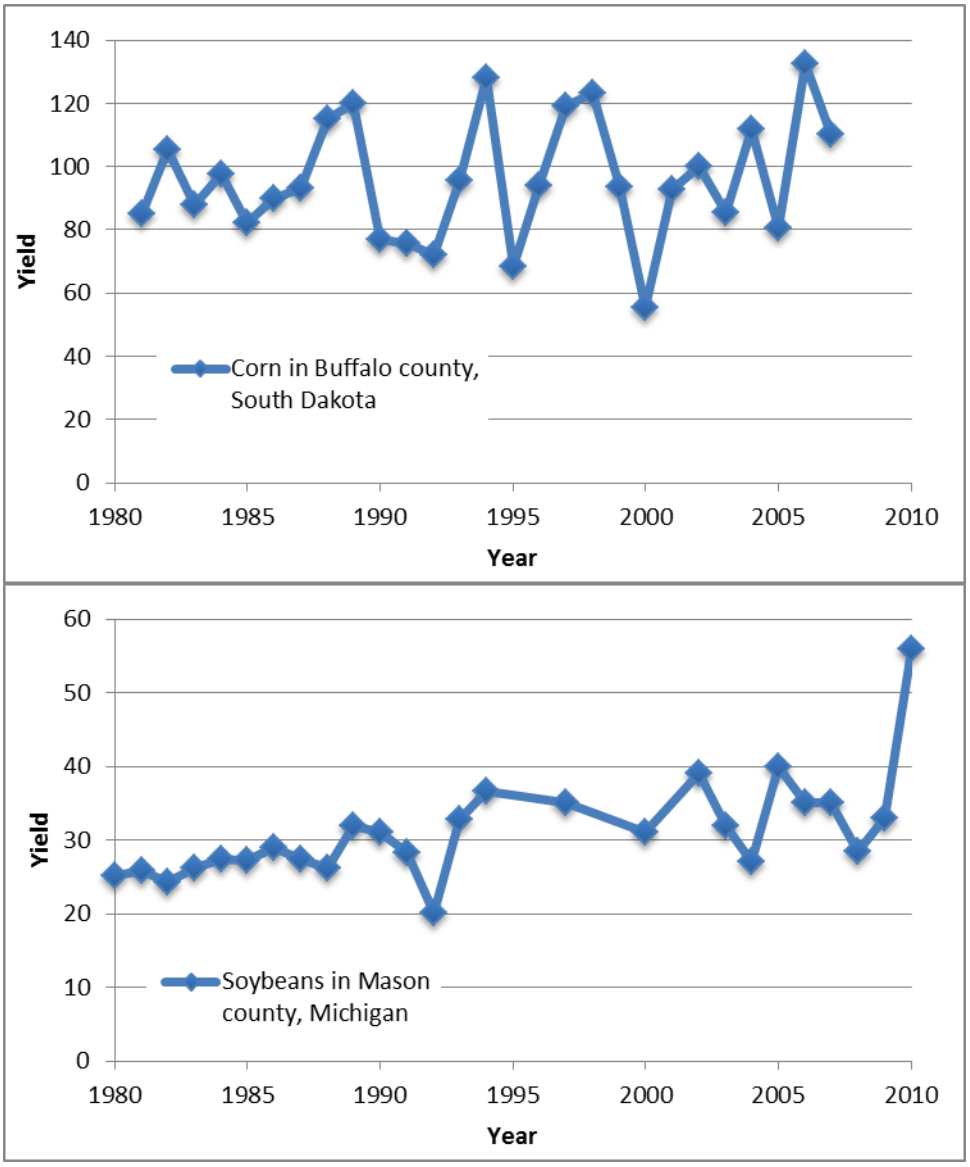

Figure 15. Average county yields for corn in Buffalo county, South Dakota (top panel), and Soybeans in Mason county, Michigan (bottom panel). Data from NASS/USDA surveys.

Therefore, a simple t-test will not have a lot of statistical power to determine to what extent an anomaly in future yields is due to natural fluctuations or due to a change in agricultural management related to a carbon project. It is possible to improve the simple t-test approach and divide an observed change in yields into (1) a change in yield caused by project activities, and (2) a change in yield unrelated to project activities. This is done by assuming that the main component that is causing fluctuations in yield over time is weather, so that any fluctuations in yield on one field are likely correlated with fluctuations in yield across a large number of fields within the same region. Therefore, the annual fluctuations of the average county yield, a statistic readily available from the NASS/USDA, can be used as a proxy of the changes in baseline yield, and can be used to normalize yields. This is the approach that was adopted in

the Reserve's Rice Cultivation Project Protocol. By normalizing field-level yields with county averages, it can be calculated what the baseline yield, the yield in absence of project activities, would have been.

The assumption behind the normalization of yields is the recognition that yields of an individual field are correlated with county yields. In other words, some fields are consistently above the county average, and some fields are consistently below the county average due to inherent differences in the fertility of a certain field and the experience of the grower. This observation is used for the basis of crop insurance contracts that pay claims based on county indices (Chaffin, 2009). The following procedure can be followed to normalize yields:

(1) Normalize the historical yields for a specific field by dividing the yield by the mean yields for the county, obtained, for example, from the USDA NASS
For the years t before t_0 ("historical yields") normalize the yield and calculate the standard deviation and mean of the normalized yields as follows:

$$y\_norm_{t,i} = \frac{y_{t,i}}{y\_county_t}$$

$$s_i = stdev(y\_norm_{t,i})$$

$$\overline{y\_norm_{t,i}} = mean(y\_norm_{t,i})$$

Where:

$y\_norm_{t,i}$	=	Normalized yield at time t for individual field $\mathbf{i}$ [Mg ha^{-1}]
$y_{t,i}$	=	Actual yield at time t for individual field i [Mg ha^{-1}]
y_county_t	=	Average yield of the county at time t for individual field i [Mg ha^{-1}]
s_i	=	Standard deviation of the historical normalized yields for individual field i [Mg ha^{-1}]
$\overline{y\_norm_{t,i}}$	=	Average of the historical normalized yields for individual field i [Mg ha^{-1}]

It is known that the ratio of two normally distributed random variables is not normal, but follows a so-called ratio distribution. However, no practical analytical solution for the ratio distribution exists, and it is assumed that the deviation introduced by assuming that this ratio is normally distributed, is minimal given the correlation between the field-level yields and county yields. In any case, it is good practice to verify that the distribution of y_norm_t is effectively

normal. In many cases, the distribution will be log-normal and an appropriate statistical transformation must be applied to y_norm_t before taking standard deviation and means.

(2) The “minimum yield threshold" below which normalized yields are significantly smaller can now be calculated assuming a student t-distribution.

$$y\_min_i = \overline{y\_norm_{t,i}} - t(0.05, n-1) \cdot s_i$$

Where:

y_min_i	=	Minimum yield threshold for individual field i
$\overline{y\_norm_{t,i}}$	=	Average of the historical normalized yields for individual field i [Mg ha^{-1}]
n	=	Number of historical observations
$t(0.05, n-1)$	=	t-distribution value with 95% confidence (for a one-tailed test) and $n-1$ degrees of freedom [-]
s_i	=	Standard deviation of the historical normalized yields for individual field i [Mg ha^{-1}]

(3) For every year of the reporting period, calculate y_norm_t and compare this value to y_min. If $y\_norm_{t_0}$ is significantly smaller than y_min, yields were significantly smaller than under pre-project conditions, even normalized for inter-annual differences. In this case, the theoretical yield that could have been attained without project activities, the baseline yield:

$$y\_baseline_{t,i} = \overline{y\_norm_{t,i}} \cdot y\_county_t$$

The best estimate of the decrease in yield caused by project activities is, therefore:

$$y\_baseline_{t,i} - y_{t,i}$$

The greatest challenge with this approach is that the power of the test (i.e., the ability to detect anomalies) is insufficient if only a few data points are available. In addition, if the yield of a field is uncorrelated to the county yield, this approach will not improve upon the simple t-test described in the beginning of this section. This is illustrated using fictional yields presented in Figure 16. Field A is highly correlated with county yields, and has yields that are consistently smaller than the county yields for most years. Field B is uncorrelated with county yields, but has the same average yield over time. The average standardized yield of field A is 85% of the county average with a standard deviation of 17%. The average standardized yield of field B, in contrast,

is 102% of the county average with a standard deviation of 34%. Applying the t-test with the normalized yields (2-sided and 95% confidence) gives a yield threshold of 51% for field A and 33% for field B, corresponding to absolute values of 48.7 bu/acre for field A and 31.6 bu/acre for field B assuming that the yield for that year was the long-term average. If a simple t-test were applied without normalization, the yield thresholds would be 38.8 and 43.1. In case of Field A, which is highly correlated to the county yields, normalization provides a more sensitive test. However, this is not the case for Field B, which is uncorrelated to the county yields. Therefore, it may be a good idea to only apply the normalization when the correlation between field-level yields and county yields is significant, using standard significance tests of the correlation coefficient.

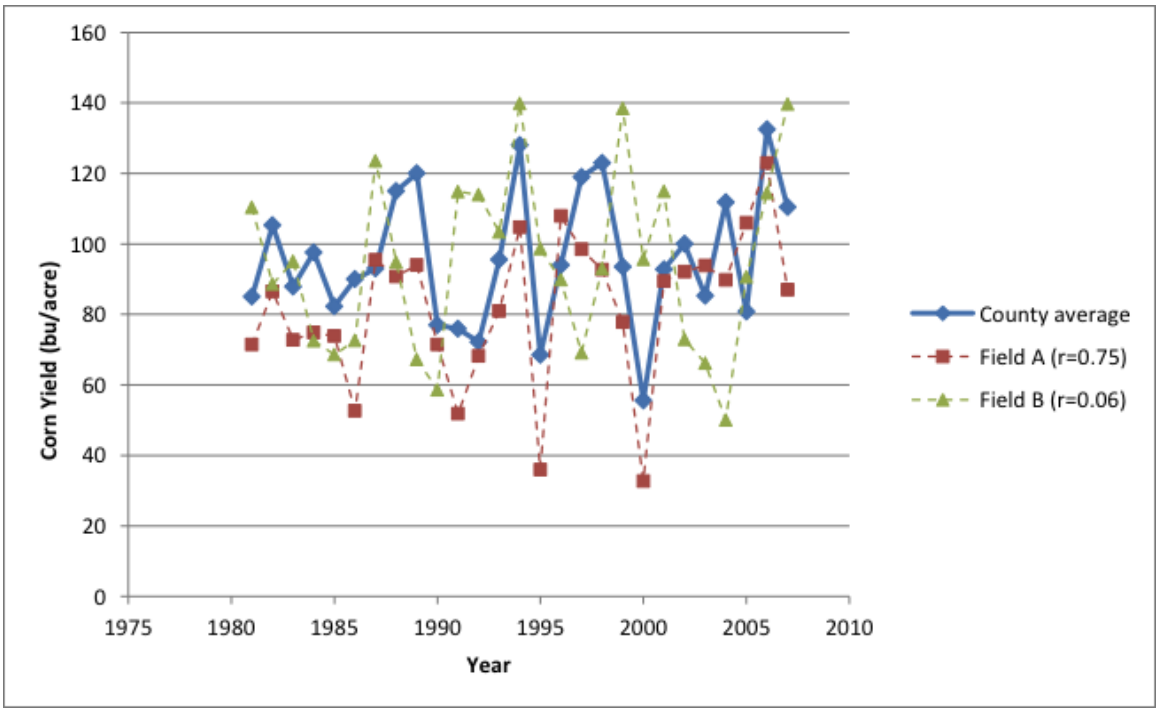

Figure 16. Yield fluctuations of corn over time for (1) average of Buffalo county, (2) a hypothetical field correlated with r=0.75 to the county yield, and (3) a second hypothetical field uncorrelated to the county average.

5.4 Accounting for Leakage

When accounting for leakage, a number of assumptions have to be made and some values must be estimated. First, it is assumed that the demand of most commodities is relatively inelastic. This is likely the case in a developed economy such as in the U.S. On the other hand, the crop acreage elasticity values from Table 11 indicate that crop acreage elasticity cannot be omitted and there may be a shift due to changes in price that farmers respond to. The crop acreage elasticity is expressed as an effective change in acreage caused by a change in the price of a commodity. However, for simplicity, it is further assumed that a change in price is directly

proportional to the yield loss caused by the project activities. Therefore, the elasticity of the supply can be simply expressed as the proportion of production that substitutes a change in yield.

The change in the GHG intensity of production in the area in which the activity shifting occurred is referred to as the GHG intensity shift. For example, when a change in the production of one commodity is compensated by a shift in the production of a different commodity with a greater GHG intensity – due to a positive cross-price elasticity, or land transformation elasticity, the GHG intensity shift will be greater than 1. Similarly, when a change in the production of one commodity is compensated by a shift in the production of the same commodity but in an area where the GHG emissions are greater due to differences in weather, soils, or management, GHG intensity shift will be greater than 1 as well.

The previous paragraph outlined how a difference in production caused by the project activity can be quantified. The amount of leakage can then be expressed as:

[Elasticity coefficient] x [Project area] x [Change in yield] x [Baseline intensity of GHG production] x [GHG intensity shift]

Where:

[Elasticity coefficient]	=	See section 5.2. A value of 100% is the most conservative assumption. A value of 60% is likely conservative for soy, wheat, and corn, based on the values presented in Table 11
[Project area]	=	The size of the project area.
[Change in yield]	=	The change in yield causally related to the project activities: $y\_baseline_{t,i} - y_{t,i}$
[Baseline intensity of GHG production]	=	The intensity of GHG emissions per unit of production in the baseline scenario, expressed as tons of CO_2-equivalents per unit of production.
[GHG intensity shift]	=	The GHG intensity shift must be determined by analyzing the area/crop type to which the activity is likely to shift and quantifying the GHG intensity of the shifted production system. The GHG intensity shift must take into account potential substitution effects assumed during cross-price elasticity, when for example, an increase in the price of corn production impacts the area of soy production. In most cases where sufficient “similar” land is available for the production of the commodity in the vicinity of the project area, the GHG intensity will be similar to the baseline

GHG intensity, and the GHG intensity shift is 1. However, when insufficient similar land is available, one can use a common practice intensity of GHG production, instead of the product of [Baseline intensity of GHG production] and [GHG intensity shift] in the equation above.

For example, if yields are (significantly) reduced from 170 bushels per acre to 150 bushels per acre, and the GHG intensity of production is 25 kg CO_2-eq per bushel, the emissions from leakage are:

(170-150) x 25 x 0.60 = 300 kg t CO_2-eq

This calculation conservatively omits any cross-price elasticity.

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7 Appendix A: Overview of Available Data in the NASS Agricultural Chemical Use Program

Table 12. Crops, years and extents for which area on which N fertilizer was applied, number of individual applications per year, total annual N application rate per area are available from the NASS Agricultural Chemical Use Program. States in brackets may not have data for all years noted.

Crop	Year	Smallest Geographical Extent
Almonds	1991, 1999	CA
Apples	1991, 1993, 1995, 1997, 1999, 2003, 2007, 2009	(AZ) (CA) (GA) MI (NC) (NJ) NY OR PA (SC) (VA) WA
Apricots	1991, 1993, 1995, 1999, 2003, 2009	CA
Artichokes	1990	CA
Asparagus	1990, 1992, 1994, 1998, 2002, 2006	CA (IL) (MI) (NJ) (OR) (WA)
Avocados	1991, 1993, 1995, 1999, 2003, 2009	CA (FL)
Barley	2003	CA ID MN MT ND PA SD UT WA WI WY
Beans (Lima, Fresh)	1992, 1994, 1998	GA
Beans (Lima, Processing)	1990, 1994, 1998	CA (IL) (MI) (NC) (NJ) (NY) (OR) (WA) (WI)
Beans (Snap, Fresh)	1992, 1994, 1998, 2002, 2006	(CA) FL GA (MI) NC (NJ) NY (TN)
Beans (Snap, Processing)	1992, 1994, 2002, 2006,	(CA) (FL) (GA) IL MI (NC) (NJ) NY OR (PA) (WA) WI
Blackberries	1991, 1993, 1995, 1999, 2003, 2009	OR
Blueberries	1991, 1993, 1995, 1999, 2003, 2009	(GA) MI (NC) (NJ) OR (WA)
Broccoli	1992, 1994, 1998,2002, 2006	(AZ) CA (OR) (TX)
Cabbage (Fresh)	1990, 1992, 1994, 1998, 2002, 2006	CA (FL) (GA) (MI) (NC) (NJ) (NY) (OH) (PA) (TX) (WI)
Cabbage (Processing)	1992, 1994	(FL) (MI) NY WI
Carrots (Fresh)	1998, 2002, 2006	(AZ) CA (FL) MI (NY) TX (WA)
Carrots (Processing)	1998, 2002	(AZ) (CA) (FL) (MI) (NY) (OR) (TX) (WA) (WI)
Cauliflower	1990, 1992, 1994, 1998, 2002, 2006	(AZ) CA (MI) (NY) (OR) (TX)
Celery	1990, 1992, 1994, 1998, 2006	CA (FL) (MI) (NY) (TX)
Cherries (Sweet)	1991, 1993, 1995, 1999, 2003, 2009	CA MI OR WA
Cherries (Tart)	1991, 1993, 1995, 1999, 2003, 2009	MI NY (OR) (PA) (WA)

Corn (Unspecified or Grain)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2005	(AL) (AR) (AZ) (CA) (CO) (CT) (DE) (FL) (GA) IA (ID) IL IN (KS) (KY) (LA) (MA) (MD) (ME) (MI) MN
Corn (Sweet, Fresh)	1990, 1992, 1994, 1998, 2002	CA (CO) (FL) (GA) (IL) (MI) (NC) (NJ) (NY) (OH) (OR) (PA) (TX) (WA) (WI)
Corn (Sweet, Processing)	1992, 1994, 1998, 2002, 2006	(IL) (MI) MN NY OR WA WI
Cotton (Upland)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2003, 2005, 2007	(AL) AR (AZ) CA (GA) LA (MO) MS (NC) (SC) (TN) TX
Cucumbers (Fresh)	1992, 1994, 1998, 2002, 2006	CA FL GA MI NC (NJ) (NY) (OR) (TX) (WA)
Cucumbers (Processing)	1992, 1994, 1998, 2002, 2006	(CA) FL (GA) MI NC (NJ) (NY) (OH) (OR) (SC) TX (WA) (WI)
Dates	1991, 1993, 1995, 1999, 2003, 2009	CA
Eggplant	1992, 1994, 1998, 2006	NJ (FL)
Figs	1991, 1993, 1995, 1999, 2003, 2009	CA
Garlic	1990, 2002, 2006	CA
Grapefruit	1991, 1993, 1995, 1999, 2003, 2009	(AZ) CA FL (TX)
Grapes (All)	1991, 1993, 1995	CA (IN) (MI) NY (OR) (PA) WA
Grapes (Raisins, Table and Wine)	1993, 1999, 2003	CA
Hazelnuts	1991, 1999	OR
Kiwifruit	1991, 1993, 1995, 1999, 2003, 2009	CA
Lemons	1991, 1993, 1995, 1999, 2003, 2009	CA (AZ)
Lettuce (Head)	1990, 1992, 1994, 1998, 2002, 2006	(AZ) CA (FL) (MI) (NH) (NJ) (NY) (TX)
Lettuce (Other)	1990, 1992, 1994, 1998, 2002, 2006	(AZ) CA (FL)
Lettuce (Romaine)	1990	CA
Limes	1991, 1995, 1999	FL
Melon (Cantaloupe)	1990, 1992, 1994, 1998, 2006, 2002	(AZ) CA (DE) (GA) (IN) (MI) (PA) (TX)
Melons (Honeydew)	1990, 1992, 1994, 1998, 2002, 2006	(AZ) CA (TX)
Melons (Watermelon)	1990, 1992, 1994, 1998, 2002, 2006	(AZ) CA (DE) (FL) (GA) (IN) (NC) (SC) (TX)
Nectarines	1991, 1993, 1995, 1999, 2003, 2009	CA
Oats	2005	CA ID IL IA KS MI MN MT NE NY ND PA

		SD TX WI
Olives	1991, 1993, 1995, 1999, 2003, 2009	CA
Onions	2006	(AZ) CA (GA) (MI) (NY) (OR) (TX) (WA) (WI)
Oranges	1991, 1993, 1995, 1999, 2003, 2009	(AZ) (CA) FL (TX)
Peaches	1991, 1993, 1999, 2003, 2009	CA (GA) MI (NC) (NJ) (NY) PA SC (TX) (VA) (WA)
Peanuts	1991, 1999, 2004	(AL) (FL) GA (LA) NC TX (VA)
Pears	1991, 1993, 1995, 1999, 2003, 2009	CA (MI) (NY) OR (PA) WA
Peas (Green, Processing)	1992, 1994, 1998, 2002, 2006	(CA) (IL) (MI) MN (NJ) NY OR WA WI
Pecans	1991, 1999	(AZ) (CA) (FL) GA (NC) (SC) TX
Peppers (Bell)	1990, 1992, 1994, 1998, 2002, 2006	CA (FL) (GA) (MI) (NC) (NJ) (OH) (TX)
Pistachios	1991, 1999	CA
Plums	1993, 1995, 1999, 2003, 2009	CA
Pomegranates	1991	CA
Potatoes (Fall)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2001, 2003, 2005	(CO) (ID) (IN) (ME) (MI) (MN) (ND) (NY) (OR) (PA) (WA) (WI)
Prunes	1993, 1995, 1999, 2003, 2009	CA
Prunes and Plums	1991	MI OR WA CA
Pumpkins	2002, 2006	CA IL MI (NY) (OH) (PA)
Raspberries	1991, 1993, 1995, 1999 2003, 2009	(MI) OR WA
Rice	1990, 1991, 1992, 2000, 2006	AR (CA) LA (MO) (MS) (TX)
Sorghum	1991, 1998, 2003,	(CO) KS (MO) (NE) (OK) (SD) (TX)
Soybeans	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002, 2004, 2006	(AL) AR (DE) (FL) (GA) IA IL IN (KS) (KY) (LA) (MD) (MI) MN (MO) (MS) (NC) (ND) NE (NJ) OH
Spinach (Fresh)	1992, 1994, 1998, 2002, 2006	(AZ) CA (NJ) TX
Spinach (Processing)	1992, 1994, 1998	(CA) (NJ) (NY) TX
Squash	2002, 2006	CA FL GA MI NC NJ (NY)
Strawberries	1990, 1992, 1994, 2002, 2006, 2008	CA (FL) (MI) (NC) (NJ) (NY) (OR) (WA) (WI)

Sugarbeets	2000	CA CO ID MI MN MT NE ND OR WA WY
Sunflower	1999	KS ND SD
Tangelos	1991, 1993, 1995, 1999, 2003, 2009	(AZ) (CA) FL
Tangerines	1991, 1993, 1995, 1999, 2003, 2009	(AZ) CA FL
Temples	1991, 1993, 1995, 1999, 2003	FL
Tobacco (flue-cured)	1996	GA NC SC VA
Tomatoes (Fresh)	1990, 1992, 1994, 1998, 2002, 2006	CA (FL) (GA) (MI) (NC) (NJ) (NY) (OH) (TN) (TX)
Tomatoes (Processing)	1990, 1992, 1994, 1998, 2006	CA (MI) (NJ) (NY) (TX)
Walnuts	1999	CA
Wheat (Durum)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2002, 2004, 2006	(CA) (IN) (MT) (ND) (SD)
Wheat (Other Spring)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 2002, 2004, 2000, 2006	(ID) MN MT ND (OR) (SD) (WA)
Wheat (Winter)	1990, 1991, 1992, 1993, 1994, 1995, 1996, 1997, 1998, 2000, 2002, 2004, 2006, 2009	(AR) (CA) CO (GA) (ID) (IL) KS (LA) (MN) (MO) (MS) (MT) (NC) NE (OH) OK (OR) (PA) (SD) TX WA

8 Appendix B: Resources on Fertilizer Recommendations for the Ten Most Important Agricultural States

Table 13. Cooperative extension service and resources available to determine fertilizer application rates for the 10 most important agricultural states.

State	Cooperative Extension Service	Main website and Fertilizer Recommendation
CA	University of California Cooperative Extension	http://ucanr.org/ Agronomy Research & Information Center: http://agric.ucdavis.edu/ UC Nutrient Management for Vegetable, Fruit and Nut Crops: http://ucanr.org/sites/nm/
IA	Iowa State University Extension	http://www.extension.iastate.edu/ Soil Fertility Homepage: http://www.agronext.iastate.edu/soilfertility/

State	Cooperative Extension Service	Main website and Fertilizer Recommendation
IL	University of Illinois Extension	http://web.extension.illinois.edu/state/index.html Illinois Fertilizer Conference Proceedings http://frec.cropsci.illinois.edu/
MN	Minnesota Extension Service	http://www.extension.umn.edu/ BMPs for N use in MN (corn): http://www.extension.umn.edu/Corn/nitrogen.html
NE	University of Nebraska Cooperative Extension	http://www.extension.unl.edu/ Soil Management to Optimize Crop Production: http://cropwatch.unl.edu/web/soils/home
TX	Texas AgriLife Extension Service	http://agrilifeextension.tamu.edu/ Crop Nutrient Needs in South and Southwest Texas http://lubbock.tamu.edu/cottoncd/east/docs/fertility/B-6053%20Crop%20Nutrient%20Needs.pdf
IN	Purdue University Extension	http://www.ag.purdue.edu/extension/pages/default.aspx Soil Fertility: http://www.agry.purdue.edu/ext/soilfertility/ Indiana Nitrogen Rate Recommendations for Corn - A Historical Perspective (1953 – 2007): http://www.agry.purdue.edu/ext/soilfertility/historical-recommendations.html
FL	University of Florida IFAS Extension	http://solutionsforyourlife.ufl.edu/ Fertilization and Nutrition http://edis.ifas.ufl.edu/topic_fertilization
KS	Kansas State University Research & Extension	http://www.ksre.ksu.edu/ Fertilization Recommendations: http://www.agronomy.ksu.edu/soiltesting/p.aspx?tabid=32
MO	Montana State University Extension Service	http://www.msuextension.org/ Soil Fertility Recommendations http://www.sarc.montana.edu/php/soiltest/ Fertilizer application guidelines, Montana State University Extension Service Publication #EB 161 http://www.mt.nrcs.usda.gov/technical/ecs/agronomy/nutrient/fert

State	Cooperative Extension Service	Main website and Fertilizer Recommendation
		ilization/fertilizer.html
OH	Ohio State University	Tri-State Fertilizer Recommendations for Corn, Soybeans, Wheat and Alfalfa http://ohioline.osu.edu/e2567/
WA	Washington State University	Washington State University Fertilizer Guides http://benton-franklin.wsu.edu/agriculture/WashingtonStateFertilizerGuides1.htm
OR	Oregon State University	Oregon State University Fertilizer guides http://extension.oregonstate.edu/catalog/pdf/fg/fg52-e.pdf
MI	Michigan State University	Developing Fertilizer Recommendations for Agriculture http://msuextension.org/publications/agandnaturalresources/mt200703AG.pdf
AZ	Arizona State University	Nitrogen Fertilizer Management in Arizona http://cals.arizona.edu/crop/soils/nitfertmgtAZ.pdf

9 Appendix C. Useful queries for NASS QuickStats 2.0

Data elements can be queried from the QuickStats 2.0 web-based tool available at http://quickstats.nass.usda.gov/. The following is a list of relevant data items and how to build queries.

- **Yield and acreage per county** and per crop
  - [*Survey*] > [*Crops*] > [*Field Crops*] > [*Crop*] > [*Area Harvested, Area Planted, Production measured in Bu*]
- Total calendar-year **fertilizer expense** per state (including lime and soil conditioners) (USD per state)
  - [*Survey*] > [*Economics*] > [*Expenses*] > [*Fertilizer totals*] > [*Expense*] > [*fertilizer totals, incl. lime & soil conditioners - expense, measured in $*]
- **Fertilizer prices**
  - [*Program: Survey*] > [*Sector: Economics*] > [*Group: Prices Paid*] > [*Commodity: Nitrogen*].