Appendices

Chapter 7: Responses: Technologies and practices

For online publication only

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A7.0 Introduction

This appendix describes the scientific basis, capacity, and applicability of management practices and technologies used to manage nitrogen (N) in California agriculture[1]. Countless methods have been developed to this end; and the discussion here is not intended to be an exhaustive review. Instead, we direct attention toward N management approaches that have one or more of the following characteristics: are commonly used, have high potential to mitigate N effects, are receiving some research attention but have uncertain effects, have the potential for unintended consequences by transferring N from one medium to another, or were of particular interest to various stakeholder groups (Box A7.1). Additional information on N management in agriculture and the mechanisms to manage N from other drivers (e.g., industry) can be found in the resources listed in Table A7.1.

[Box A7.1] [Table A7.1]

A7.1 The nitrogen cycle

Understanding the potential efficacy of changing management on regulating the N cycle requires knowledge of N cycling processes. Through management, producers modify the quantity of reactive N available and conditions of the soil environment. By changing the substrate quantity and soil biological, chemical, and physical properties, they alter the tendency for and pace of microbial N transformations, plant uptake, chemical conversions, and emissions. It is the ability to impact these processes that create

[1]Engineering technologies used to control N emissions due to fuel combustion and waste management are transferable, well established, and covered in depth in other texts. Therefore, the discussion here focuses solely on agricultural nitrogen management.

opportunities to control the N cascade[2]. Descriptions of the forms of N and the major process of the N cycle can be found in Table A7.2

[Table A7.2]

Actions to regulate N dynamics affect the amount of reactive N in the environment through one of six mechanisms: conservation, substitution, transformation, source limitation, removal, or improved efficiency (INC 2011). Examples are constructing wetlands to intercept NO_3^- in runoff (removal), use of nitrification inhibitors to retard conversion of NH_4 to NO_3^- (transformation), and improving distribution uniformity to increase the efficiency of irrigation (improved efficiency). Applicability of each strategy is subject to the constraints of the production environment (Table A7.3). Often there are multiple approaches available to modify N for a given combination of flow and production environment, with the best strategy emerging from the confluence of a litany of factors including, but not limited to: availability of technology costs, effectiveness, crop or animal species of interest, soil, irrigation system, regulations, climate, labor, and the market. The most appropriate response, therefore, is to select options to optimize among technological capacity, social-private and public goods, and environmental outcomes, subject to the context of the farming activity.

[Table A7.3]

A7.2 Inorganic nitrogen management

Nitrogen management refers to four, not mutually exclusive, decisions regarding the rate, source, timing, and placement of fertilizing materials. The canonical objective of N management, regardless of whether inorganic or organic, is to match the availability and supply of N with crop demand as closely as

[2] For a description of the N cascade, see introduction of Chapter 7.

possible[3] (Cassman et al. 2003; Ladha et al. 2005). Synchronizing supply and demand results in high fertilizer use efficiency and decreases pollution potential (Dobermann 2005). In practice, however, plant availability of inorganic N, assimilation by roots, and gaseous and water-borne emissions are a function of a multitude of biological and chemical processes whose rates vary across space (fields, farms, and landscapes) and time (days, months, years) and are subject to a series of constraints ranging from climate to cultivars to cultural practices. A grower is, thus, faced with balancing complex and variable relationships between biology and technology. The challenge of managing these complex relationships underlies the efficiency, and inefficiency, of N fertilizer use in California.

A7.2.1 Reduce nitrogen application rates[4]

Crop production in California requires the addition of N fertilizer to supplement indigenous soil reserves. Simply put, applying N fertilizer to the soil turbo charges the N cycle. Microbial activity increases and the many N transformations they mediate accelerate. Amplification of the biological processes plus the comparatively greater magnitude of N in the system following fertilizer application catalyzes plant growth but is also responsible for additional emissions risk. It is well established that yields increase along with N application rates until a threshold is reached where N no longer limits production; at which point, productivity plateaus or even declines (Cassman et al. 2002). Constraints and realities of farming—technology, information, economics, and weather—require crop producers to supply more N than a crop assimilates to ensure adequate nutrition and high yield. However, growers often apply N in quantities beyond the rate of N uptake. Because of the surplus N use, reducing the rate of application is an often-cited option to control emissions without compromising yield.

[3] It is important to understand that it is practically impossible to perfectly match soil N supply with plant demand. Growers must add more fertilizer N than the plant takes up to maintain high levels of productivity.
[4] The quantity of fertilizer used is called the “application rate” or “rate”, for short.

Reducing N application rates limits the introduction of new N into the system and should decrease NO_3^- leaching and gaseous emissions of nitrogenous compounds. The relationship between emissions and N rate is typically inverse to that of productivity and N rate. Research on N_2O and NO_3^- losses suggests emissions remain low, only slightly elevated above background levels, until a threshold is reached, near the season maximum amount of N taken up. After the N rate threshold is exceeded, pollution increases exponentially (Venterea et al. 2011; van Groenigen et al. 2010; Millar et al. 2010; Hoben et al. 2011). According to a meta-analysis of 18 studies, once N application rates exceed 11 kg per ha greater than plant uptake, N_2O emissions increase exponentially for marginal additions of N fertilizer (van Groeningen et al. 2010). Similar relationships have been suggested for leaching and N fertilizer applications (Broadbent and Carlton 1978). What this research suggests is that incremental reductions in N applied may have multiplicative effects on emissions, assuming N additions exceed plant uptake in the cropping system. Although the precise inflection point will be determined by edaphic soil, crop, and management factors including; irrigation efficiency, carbon (C) availability, timing and placement of fertilizer applications, identifying a threshold provides a metric for growers and custom fertilizer applicators to target.

Changes in N application rates have the potential to decrease yields. Lower productivity may result from either insufficient quantities of N throughout the year as might occur during ideal growing conditions or if N is unavailable during critical phenological periods. Part of the reason growers apply N at current rates is to hedge against such risks (the “insurance” hypothesis). Nevertheless, widespread over-fertilization has been documented in some California crops (Breschini and Hartz 2002; Hartz et al. 2000; Johnstone et al. 2005). Under these conditions, N applications could be reduced without jeopardizing productivity or economic solvency. For example, Hartz et al. (2007) surveyed 78 fields of iceberg and romaine lettuce and found the average N application rate was 184 kg per ha but ranged between 30 to 440 kg per ha. Current University of California (UC) guidelines suggest an application rate between 196 to 240 kg per ha is sufficient for these crops under most production conditions (Chapter 3). Even though the N rate varied by more than 300 kg per ha, yields were not correlated with N rate suggesting misapplication on many sites. Less is known about the potential for over application of fertilizer N in perennial and field crops. One of the only recent surveys of N management practices in perennials did not ask about common N rates in nut crops (Lopus et al. 2010). Hartley and van Kessel (2003) document N rates in rice production. According to their survey, average application rates are within the range of guidelines. Overall, average producers of 5 of 12 vegetable crops and 4 of 12 perennial crops, but 0 of 5 field crops apply more N than the maximum rate suggested in the UC guidelines suggesting there may be opportunities for reducing fertilizer N rate on many crops (Appendix 3.2 and 3.3). Clearly some crops are systematically fertilized excessively. But even for crops that are generally not, potential rate reductions are plausible simply because of the wide ranges in N application rates among fields and farms.

Reducing rates requires more intensive management. Using an N management program that involves diagnostic testing to guide split N applications was shown to be able to reduce N application rates by 60 to 112 kg per ha (approximately 30% of N applied) by comparison to industry standard fertilization practices in processing tomatoes (Hartz et al. 1994). Although the latter results likely significantly overestimate potential reductions at this time due to recent meteoric increases in tomato yields and N uptake with adoption of micro-irrigation (Hartz and Bottoms 2009), they are illustrative of conceivable capacity to better target N decisions.

For growers to reduce rates, information on crop demand and the technology to supply N are critical inputs to guide growers' decisions on when, where, and how much to apply. The two primary tools California producers currently use to guide fertilizer N rate decisions are soil and tissue tests. Soil tests provide an indication of the mineral N in soil and plant-N availability. Tissue tests, in contrast, indicate the sufficiency or deficiency of N within the plant. Extensive research in vegetable crops has proven the value of soil tests for N decision-making (Hartz et al. 2002; Breschini and Hartz 2002; Hartz et al. 1994). Comparatively, the utility of tissue sampling in perennial crops has been called into question recently (Brown personal communication). Antiquated sampling protocols that do not adequately account for spatial and temporal heterogeneity of soils or crop processes (Rosenstock et al. 2010) and "critical sufficiency values"[5] established for cultivars and conditions unrepresentative of agriculture today make tissue tests a blunt tool, at best. Furthermore, the ability to apply split applications and deliver fertilizer rates varies by cropping system and management and is impacted in fertigated systems by the distribution uniformity/irrigation efficiency of irrigation technology used and its management. In some cases, the size of the field and the economics of repeated management may preclude increased number and better timed/even delivery of nutrients.

A7.2.2 Change inorganic nitrogen fertilizer sources

Individual reactive N species are more or less susceptible to microbial transformations, adhesion to soil clay particles, or chemical conversion. Selection of an N source that promotes or suppresses specific N cycle attributes is thus theoretically possible. Options available to change inorganic N sources include: (i) switching between conventional materials (e.g., from ammonium sulfate to calcium ammonium nitrate) or (ii) switching from conventional synthetic materials to "enhanced efficiency materials" [6].

[5] Critical values refer to the concentration of nutrients within plant tissue. They are experimentally derived and reflect nutrient concentrations at a specific time of the year. See Embleton and Jones (1974) and Lovatt (2001) for examples of those in development and still in use.

[6] Enhanced efficiency fertilizers (EEF) are synthetically derived materials that are engineered to moderate the rate N becomes available to plants and microbes, extending it over a longer period of time (Shaviv and Mikkleson 1993). They achieve this by either building protective shells around solid fertilizer that dissolve—e.g., sulfur coated—or using chemicals that retard microbial action—e.g., nitrification inhibitors. The nature of the material itself and environmental conditions—namely temperature and soil moisture—determine the rate of N release, with N being released more rapidly under hotter, wetter conditions. It is important to note that a wide range of EEF are available in the marketplace—from nitrification inhibitors to polymer coated urea—and their mode of action in the soil is different.

Changing between conventional materials can be an effective strategy to reduce NO_3^- leaching and NH_3 volatilization losses but the effect on N_2O emissions is uncertain but not likely significant if at all. With their negative ionic charge and water solubility, NO_3^--based fertilizers do not adhere to similarly charged clay particles and therefore are not readily retained in the soil matrix. They readily leach below the rootzone with water, especially with uneven distribution of irrigation water or with precipitation (Letey 1994; Hanson et al. 2005). Utilizing NH_4-based fertilizer helps retain N in the soil rootzone longer providing greater opportunity for crop uptake.

More so than leaching, changing fertilizer type can dramatically mitigate NH_3 volatilization. Recall that volatilization is a physiochemical reaction of soluble NH_4 being converted to gaseous phase. Thus, fertilizers that contain NH_4 or hydrolyze easily to this compound (e.g., urea) will have considerably higher emissions, especially when applied to the soil surface. Harrison and Webb (2001) conclude from their review of the literature that emission rates from urea-based fertilizer often exceed 40% of N applied while that from ammonium nitrate are an order of magnitude lower. Limited use of urea and widespread use of mixed ammonia and nitrate fertilizer blends are reasons volatilization from current California cropping systems that use chemical fertilizer accounts for a relatively insignificant N flow. Recent empirical results show that only an average of 3% of N applied is given off as NH_3 under California production condition (Krauter and Blake 2009).

Changing fertilizer type is unlikely to have a significant effect on N_2O flux. After reviewing more than 1000 studies of N_2O production, Stehfest and Bouwman (2006) conclude that rates of N_2O evolution from various fertilizers are practically indistinguishable when accounting for the experimental differences—tillage systems, fertilizer placement, soil C, and pH (Snyder et al. 2009). Few experiments of their dataset represent the intensive arid and semi-arid conditions similar to California agriculture and thus extrapolation of the impacts to these conditions are somewhat speculative. However, the variable and often insignificant results suggest that switching between two conventional fertilizer types holds little promise to mitigate N_2O emissions.

In contrast, switching to enhanced efficiency fertilizers (EEF) from conventional synthetic fertilizers is often widely considered a valuable technological option to address the N challenge (INC 2011; Akiyama et al. 2010; Halvorson et al. 2010). Data suggest EEF are effective at reducing N losses. A recent meta-analysis of the efficacy of EEF to regulate N_2O emissions demonstrates polymer coated and nitrification inhibitors decrease N_2O by 35% and 38%, respectively (Akiyama et al. 2010). But the results of the research on EEF and N_2O may be confounded by experimental design. Evidence suggest that although EEF present lower initial fluxes, N_2O production may extend for longer periods and therefore may show higher total losses (Delgado and Mosier 1996). Research, some of it done in California, has also shown EEF slows downward percolation of NO_3^- under irrigated conditions. Stark et al. (1983) studied the effects of N fertilizer type and irrigation management on NO_3^- movement on a loam soil. Less NO_3^- migrated below rootzone when sulfur coated ureas was used by comparison to conventional fertilizer product. However, water management may swamp any benefits from EEF. Stark et al. (1983) found that excessive irrigation pushed NO_3^- down through the soil profile irrespective of N source.

Utility and likelihood of switching to EEF in California is questionable[7] however, especially in the near term. To begin with, EEF are more expensive. Estimates range from 9% (Snyder et al. 2009) to nearly double (California Nitrogen Assessment (CNA), stakeholder meetings). This additional cost is unwelcome without clear yield increases. EEF in recent California vegetable crops trials raised yields only twice in nine experiments, 22% of the time (Hartz and Smith 2009). In the late 1970s and mid 1980s, it was shown that nitrification inhibitors did increase N recovery in strawberry, cauliflower, and lettuce (Welch et al. 1979, 1985). With today's system, however, it is not clear if EEF will produce comparable

[7] Strawberry is the only current cropping systems where the use of slow release fertilizer is the industry standard (Strand 2008, Reganold et al. 2010).

benefits in California as in other regions for which they are being touted. Benefits of EEF are maximized when periodic and uncontrolled soil moisture decrease control of N, conditions only found during winter in some parts of California agricultural valleys. The more common production conditions - hot, dry, and fertigated - can provide equivalent or greater control of nutrients if managed astutely.

Selecting appropriate fertilizer formulations to minimize emissions risk may be an important mitigation strategy for some losses. But there is no universal 'best' inorganic N source to serve growers needs and protect the environment.

A7.2.3 Modify fertilizer placement and timing

Fertilizer application timing and placement offer the opportunity to manage the location, size, and duration of inorganic soil N pools and thereby influence crop uptake. When fertilizer is positioned in the region of greatest root activity during periods of peak plant demand, plants generally have a competitive advantage over soil microorganisms. Resulting plant uptake reduces the soil mineral N pool, leaving less available for microbial transformations that prime it to be lost from the rootzone.

Improving the timing and placement of fertilizer applications almost universally increases N recovery and often results in greater crop productivity. Scheduling fertilization events to coincide with periods of peak crop demand is critical to better the timing. In avocado, specifically matching fertilization events with key phenological periods of rapid vegetative growth (mid-November and mid-April) increased productivity—total weight and fruit size -- from 30% to 39% over four years (Lovatt 2001). Avoiding using N fertilizer prior to winter is an equally important timing strategy. Fertilizer applied without actively growing plant cover is often lost. In a peach trial, fertilizer recovery increased 18% (58% vs 50%) by simply applying N in spring versus fall (Niederholzer et al. 2001). Even more dramatic results illustrating the need to not apply N in the fall are available from research throughout the Midwestern US (Robertson et al. 2011; Snyder et al. 2009). Knowledge of crop growth patterns underlies the ability to split fertilizer applications to meet crop demand. Each crop species has distinct growth patterns, where nutrient demand is critical to further plant development. But generally, N demand of fruiting crops increases steadily while fruit develop and then declines in a bell shaped pattern over the season. In contrast, non-fruiting crops such as lettuce will increase gradually and require increasing amounts of N throughout the entire production cycle (Hartz et al. 1994). Practical complications stem from the need to ensure sufficient quantities of N when peak N demand occurs, anywhere from a few weeks as in corn (Pang and Letey 2000) to a few months as in pistachio (Rosecrance et al. 1998).

Placement can also have a large impact on crop growth and N recovery. For example, Linquist et al. (2009) compare yields and fertilizer recovery of rice grown relying on surface or subsurface applications. Fields with only subsurface N applied recovered an average of 46% more N (53% vs 38%) and grain yields were higher. But it is important to note that improved timing and placement do not always result in increased productivity. Hutmacher et al. (2004) demonstrate that yields of Acala cotton grown across six farm sites in the San Joaquin Valley were statistically similar regardless whether a single or two applications were used. Resources required for additional application would thus have little value.

The impact of improved timing and placement for controlling N leaching and denitrification is relatively more uncertain. Logically, when plants out-compete microbes and assimilate a greater fraction of the available N, losses of N from leaching and denitrification should be reduced. However, it cannot be assumed that emissions will be reduced with better placement and timing alone. Indeed, evidence is mixed. One study (Hultgreen and Leduc 2003 cited in Snyder) shows lower N_2O emissions from band placement versus broadcast surface applied urea. Yet another demonstrates that band placement with urea results in emissions more than four-fold greater (Engel et al. 2010). Increased emissions from band placement might be attributed to extremely high N concentrations within the small area covered by the band; essentially banding creates a hypersaturated zone. Unfortunately, data on the effects of improved timing and placement on N_2O emissions is not available for California.

Improving the placement and timing of fertilizer N are unlikely to significantly alter N cycles in California croplands. This is in part a consequence of the ambiguity in the predicted response, but more so because the practices are already commonplace (Weinbaum et al. 1992). Growers have been splitting fertilizer N applications for some time. The most recent statewide fertilizer use survey asked more than 800 growers in the late 1990s about their N management in 1986 and 1996 (Dillon et al. 1999). The number of respondents that applied N in a single application decreased by 9.2% and the number of growers that applied three or more applications rose 5.7%. Although current use of these practices is largely not quantified, anecdotal evidence from CNA stakeholder meetings with farmers and UC Cooperative Extension agents suggest that these trends have continued, as research repeatedly demonstrates yield benefits from these practices and this underlies most recommendations (Hartz et al. 1994; Breschini and Hartz 2002; Rosecrance et al. 1998; Lovatt 2001). Blanket statements about the effectiveness of a management practice though miss the idiosyncrasies of California production. There are clearly specific production systems where better timing and placement may be appropriate. Rice may be one exception where better placement would increase N recovery (see discussion above) and strawberry may be one exception where research on the timing of N fertilizer application (currently largely applied approximately 6-weeks prior to planting) may need to be reevaluated, especially in light of changes in management due to restrictions on the use of methyl bromide.

Ensuring N is available at the right place and time to satisfy plant demand while simultaneously minimizing inorganic soil N accumulation is a central tenet of sustainable N management (Roberts et al. 2007). Generally, however, the prospect of either fertilizer timing or placement having a considerable impact in California is limited because capacity to achieve this requires knowledge of (i) crop growth patterns, (ii) ability to predict their responses to changes in weather, and (iii) the technology to precisely deliver N when and where it is needed. Information to satisfy the first requirement is reasonably available for field and vegetable crops. Much less is known about N demand and distribution patterns in tree crops (Rosecrance et al. 1998; Southwick et al. 1990; Christianson et al. 1990). The second criterion is more difficult to meet. While data exist documenting uptake rates for many crops, again trees notwithstanding, the capacity to predict weather events at scales appropriate for grower decision making limits their ability to plan and relegates fertility management to be largely prescriptive (see discussion of diagnostic testing in N rate above for exception). This is amplified because of California's highly variable weather among years, which can cause yields to vary by 50%.

Precision agriculture technology[8] may assist in improving fertilizer placement as well as in-season application timing for some field crops. Rice and cotton have been the focus of some experimentation and adoption with precision agricultural technologies (Roel et al. 2000). Evidence of its application and effectiveness in the field is lacking. However, it is either unavailable (e.g., for horticultural systems) or not well adapted (e.g., able to deliver nutrients at a meaningful scale of spatial variation). An effort is underway to adapt precision agriculture to tree crops; harvesters and irrigation systems are under development (Rosa et al. 2011), but engineering and biological obstacles currently impede their practicality. Potential fertilizer N efficiency gains from precision agriculture, beyond simple diagnostic soil and tissue tests remain distant.

A7.3 Water management

Water regulates biological activity, chemical conversion of N, and physical transport of N in soils. Nitrogen moves into plant roots and tissues with water via diffusion and mass flow. Plants cannot assimilate N from dry soils and thus growth is, at minimum, compromised without the presence of

[8] Precision agriculture refers to a suite of technology-rich geospatial and information decision tools that increase place-based fertilizer N decisions (e.g., GPS, spatially variable fertigation).

sufficient water, and potentially altogether halted. Dry, well-aerated soils favor nitrifying bacteria, can be a source of NO, and tend to accumulate NO_3^- , increasing the risk of leaching and denitrification losses when soils become rewetted. Excessive soil moisture, throughout the entire field or locally, physically dissolves and translocates soil chemicals including N. Saturated conditions also restrict gas diffusion. Soil environments with high water content reduce oxygen concentrations, which stimulate denitrifying bacteria to use NO_3^- in its place. Nitrous oxide production can result; the rate depending on local conditions, such as water filled pore space and the presence of a readily available energy source, e.g., C (Davidson et al. 2000). Due to the significant influence of soil water content on a multitude of soil N cycling processes, any discussion of N management in agriculture must jointly consider water management.

Managing soil moisture content in California is unique by comparison to most other agricultural regions of the US and elsewhere. The Mediterranean climate creates two distinct management periods, a summer growing season characterized by hot day time air temperatures and negligible precipitation and a winter cropping season characterized by cool moist weather with episodic and often intense rain events. The lack of summer precipitation, and the resulting dry soils, means crop production during these periods requires irrigation. Wetting and drying cycles resulting from irrigation generally reduces soil aeration and increases microbial activity, and accelerates the transformation of N. Although irrigation can create conditions conducive to N loss, irrigation by definition controls the quantity and timing of soil moisture, and thus provides opportunities to moderate the N cycle not found in rainfed systems. The prospects to control soil water content during winter cropping periods are limited (see Section 7A.3.2). Large rain events that often occur during fallow and dormant periods between active growing cycles can be acute times of N losses when crop residues decompose and surplus mineral N fertilizer remains from the previous season (Cavero et al. 1999; Jackson 2000; Kallenbach et al. 2010).

A well-designed, -functioning, and -managed irrigation system maintains N in the rootzone longer; increasing plant N uptake potential and reducing leaching losses (Feigin et al. 1982a, b). The positive outcomes are mostly a consequence that water is the dominant factor dictating NO_3^- movement laterally and vertically through the soil profile in irrigated croplands of California. Collecting samples from tile drain effluent from 58 sites growing a range of crops throughout California's agricultural valleys demonstrates that mass emissions of NO_3^- (kg) are most significantly correlated with the amount of water moving beyond the rootzone, even more so than the amount of N used (Letey et al. 1979; Pratt 1984). Subsequent studies implicate poor irrigation efficiency, applying water in excess of beneficial uses (Meyer and Marcum 1998; Feigin et al. 1982; Stark et al. 1983) and low distribution uniformity as culprits (Pang et al. 1997; Allaire-Leung et al. 2001) responsible for increasing drainage and leaching. Conclusions are thus consistent with that outlined in the seminal research of the 1970s (Pratt 1979 and subsequent publications): efficient irrigation is a prerequisite for high productivity, low leaching agricultural systems in California.

The fact that soil water content significantly alters the nature and magnitude of gas emissions is well described (Schlesinger 1999; Davidson et al. 2000). Yet data are limited relating irrigation management and control of gaseous emissions. Presumably better water management (e.g., higher efficiency and uniformity[9]) would decrease emissions due to enhanced control of wetting and drying cycles and dampening the effects of soil spatial heterogeneity similar to its effects on leaching.

[9] Two interrelated metrics are used to describe irrigation system performance: (i) uniformity and (ii) efficiency. Uniformity relates to the evenness of distribution of water applied or infiltrated across the field's extent. No irrigation system can practically apply water at 100% uniformity. Spatial heterogeneity of soils and the length of the furrow affect uniformity. Because the common practice is to irrigate until the entire field receives sufficient water, non-uniform irrigations result in sections receiving significantly excess water. Length of furrows, differences between day and night irrigation set time, long irrigation set times, variable pressure, and clogged drip emitters are a few reasons for poor irrigation performance. Irrigation efficiency refers to the amount of water used for beneficial needs (crop evapotranspiration, leaching salts, frost protection, or cooling) related to the amount applied. The goal is to replace soil water lost through evapotranspiration. But low uniformity and the practicality of current systems including those reasons mentioned above and difficulty in predicting crop needs means that water often has to be applied at rates exceeding demand.

Kallenbach et al. (2010) compared N_2O emissions between furrow irrigated and subsurface drip irrigation in a processing tomato system and found that there were greater N_2O fluxes from the furrow irrigated systems during the rainy season without a cover crop and during the growing season when a leguminous cover crop had been planted the previous winter. These results suggest the higher performing subsurface drip system (38.12 cm of water was applied versus 88.64 cm under furrow) provides mitigative benefits. However, research is needed to clarify the nature of the relationship, especially since many gas emissions represent only a small flux of soil mineral N, e.g., $N_2O \approx 1.4\%$ and $NH_3 \approx 3\%$ of N applied in California (Appendix 4.3, Krauter and Blake 2009).

A7.3.1 Improve irrigation system performance

Irrigation system performance is a function of underlying soil properties, technology, and management (Hanson 1995; Breschini and Hartz 2002). What that means, in practice, is that there are many factors that influence irrigation efficiency and distribution uniformity, some of which producers control and others they do not. Growers have limited capacity to affect soil texture and heterogeneity (Childs et al. 1993; Letey et al. 1979). They, however, do decide when, where, and how much water to apply, subject to the constraints of the irrigation and cropping system designs, water and labor availability, and irrigation district policies. And it cannot be overstated that management decisions can override technical capacity of irrigation systems. Analyzing data from nearly 1000 irrigation systems, Hanson (1995) found that distribution uniformity and irrigation efficiency among irrigation types were similar in practice despite the greater technical potential of pressurized systems. It is likely that management has generally improved to capitalize on the advantage pressurized systems present in the 16 years since these data were presented, but that is not a foregone conclusion (e.g., Breschini and Hartz 2002).

Surface irrigation accounts for more than 50% of the irrigated acreage, although pressurized irrigation systems are increasingly widespread (Orang et al. 2008). Optimizing surface irrigation systems requires improving uniformity of infiltration and using the appropriate set times. The most effective way of increasing uniformity with surface irrigation is reducing the field length. Fields half the length (e.g., 150 vs 300 m) have been shown to increase uniformity 10 – 15 percentage points and decrease subsurface drainage by 50% (Hanson 1989). Such gains result from the shorter water advance times reducing infiltration heterogeneity along the length of the field. Shorter furrows however frequently conflict with practices, including demand for labor, and represent a significant increase in cost for producers. Other options to increase performance with furrow irrigations are surge irrigation (Hanson and Fulton 1994) or use torpedoes to compact soil and allow water to move more quickly down the furrow, with the effectiveness of these practices dependent on soil type (Schwankl and Frate 2004).

Pressurized irrigation systems provide a higher potential technical efficiency over surface applications. With pressured systems, improving irrigation is simple. The system must be designed, engineered, and operated correctly to achieve high performance standards. Switching from surface irrigation to a low volume irrigation system will improve performance, assuming appropriate management. In one study comparing irrigation technologies on lettuce in the Salinas Valley, similar yields were obtained with drip while only using an average of 61% of the water used on furrow over three years (Hanson et al. 1997). Goldhamer and Peterson (1984) found yields of cotton were greater with linear-move sprinklers than with furrow and produced less deep percolation. There is no doubt pressurized irrigation systems can distribute water more effectively if working properly and thus converting croplands to their use has significant potential to affect change of the N cascade.

Decisions about the best strategy to improve irrigation management must consider the entire production envelope. The response is frequently dictated by farming and water economics. For example, in production of lower value crops that primarily rely on surface irrigations, surface irrigation may be the only economically justifiable solution. Cotton is more profitable when using furrow irrigation but this management practice presents greater potential for subsurface drainage (Hanson and Ayars 2002) and thus the tradeoffs between economic viability and groundwater contamination are clear. Similarly, in some areas, parcel size and shape together with land ownership patterns preclude the viability of sprinkler systems on forage crops. Accordingly, water management and the call to improve irrigation efficiency from policy makers and environmental and social advocacy groups will exert significant economic pressures on farmers.

A7.3.2 Modify subsurface drainage

In areas of considerable soil drainage[10], placement of engineered drainage systems is an option to decrease deep percolation of NO_3^-. Drains change hydraulic soil properties creating a hydrologic gradient that moves water toward the drain, essentially creating a vacuum to suck up soil water. Captured leachate in agricultural areas is typically N-rich. Letey et al. (1977) found that median NO_3^- concentration of tile drain effluent was 28 ppm NO_3^--N, almost three times the legal drinking water standard. By capturing leachate, drains prevent deep percolation of N to groundwater.

Drainage presents potential for pollution swapping. Drainage simply transfers N concerns elsewhere. Removal of N from the soil decreases leaching potential, but also decreases denitrification potential (Lund et al. 1974). Nitrogen in drain effluent still needs to be disposed of in an environmentally friendly way. Usually, drainage effluent is transferred off-site and disposed of into surface waters. Nitrogen rich effluent then becomes a source of surface water contamination and can contribute to indirect N_2O emissions. Thus, drainage installation is not a stand-alone remedy for excessive N application. When used in combination with options capable of handling the N-rich wastewaters (e.g., biological denitrification reactors), installing drainage systems becomes an option that will reduce N loading.

[10] Drainage refers to the movement and removal of subsurface water from the crop rootzone. Well drained soils create optimal conditions for crop growth and management. Excess water inhibits root development, contributes to root zone anoxia, promotes disease, and prevents access to fields by machinery for crop maintenance.

A7.4 Alternative soil management

Soil management, in the broadest sense, encompasses virtually every cropping decision a grower takes, from tillage to N fertility management. Alternative soil management refers to a subset of practices to manage soil resources that are less widely adopted including: conservation tillage, organic N amendments, and cover crops. An important unifying characteristic of alternative soil management practices is that they add both C and N to soils either from plant or waste residues.

A7.4.1 Conservation tillage

Tillage[11] causes short and long-term changes in soil nutrient dynamics. Through exposing protected soil organic matter to microbial degradation and oxidation, tillage can lead to the loss of soil nutrients (Reicosky 1997). For C, this means increased decomposition and CO_2 respiration; for N, the result is growth of the soil mineral N pool and associated greater denitrification or leaching potentials. Because of this, some suggest reduce the intensity of tillage to attenuate negative perturbations of agricultural nutrient cycles (Lal 2004; Pacala and Socolow 2004).

Conservation tillage[12] presents its own challenges for managing nutrients. With slow decomposition of organic residues at the soil surface, net N immobilization can occur (Doane et al. 2009). Often this immobilization results in lower yields in the short term if not adequately accounted for in the fertility program (Doane et al. 2009). Microbial nitrification will decrease soil surface pH and

[11] Tillage is the cultivation of land by ploughing, ripping, or turning soil. Tillage's primary functions are to aerate the soil, control weeds, improve water infiltration, and distribute fertilizers throughout the profile (Loomis and Connor 1992). Through tillage, soil structure, bulk density, and porosity as well as hydraulic properties such as water retention, hydraulic conductivity, water infiltration, and percolation generally improve (Balesdent et al. 2000; Wu et al. 1992; Lal 1999; Hubbard et al. 1994). Tillage also can change soil pH, but direction of effects depends on the tillage regime (Blevins et al. 1983). An important consequence of tillage is that it increases carbon loss and soil organic matter decomposition.

[12] There are many reduced tillage systems. The extreme is no-till where soils are not disturbed. Conservation tillage, which is more often practiced in California, relates to any tillage system that maintains at least 30% residue cover throughout the year (Mitchell 2009).

presumably decrease volatilization potential, unless lime is applied. In the surface profile, reducing tillage intensity will increase soil organic C (SOC) in the topsoil (Lal 2004). Evidence of increased SOC from conservation tillage throughout the soil profile is limited, despite widespread claims (Baker et al. 2007). Decaying organic residues form a readily available source of C for soil microorganisms, which can lead to increased rates of denitrification by comparison to conventional tillage (Li et al. 2005; Snyder et al. 2009). Though the effect is inconsistent, it appears to be sensitive to fertilizer placement (Venterea et al. 2011), and may be mitigated if reduced tillage is practiced in the long-term (Six et al. 2004). Inconsistent experimental findings, interacting management factors, and antagonistic pollution potential suggest conservation tillage is an imperfect tool to manage N cycling in California.

Conservation tillage is a technical term, with specific constraints on soil surface coverage, and simply reducing tillage intensity somewhat offers many agronomic and environmental co-benefits such as, dust control, water infiltration, and reduced fossil fuel consumption (Mitchell et al. 2007; Linquist et al. 2008). But its utility for sequestering soil C and mitigating N emissions from California croplands is questionable, especially in the near term. Root density and structure will have a large effect on soil C accumulation and crop growth patterns are sensitive to soil microclimates. Residue cover tends to decrease soil surface temperatures allowing roots to amass closer to the surface than they might otherwise. Comparisons of reduced and conservation tillage based only on surface soil C may therefore inherently bias results (Baker et al. 2007). Long-term observations at three sites demonstrate the potential variability in changes in C stocks. De Gryze et al. (2010) show changes in SOC range from -50% to 100% when comparing conservation with standard tillage. Net greenhouse gas emissions were slightly less from systems using conservation tillage. Kong et al. (2009) compared N_2O emissions from minimum and standard tillage practices and found peak fluxes from minimum tillage using inorganic fertilizer were more than double that from standard tillage. Preliminary results from an ongoing examination of N_2O emissions from tomato-wheat rotations under conventional and conservation tillage suggest reduced tillage emitted 37% less N_2O of the N applied (48% versus 76%) (Kennedy et al. 2012). What can be concluded is that the mitigative impacts of reduced tillage depend on a series of other production factors, which are difficult to predict, and uncertain.

Until recently, California cropping systems were not adapted for conservation tillage. Because reduced tillage requires specialized equipment and California crop typology is so diverse, a lack of appropriate implements impeded its use. Today, it is possible to grow processing tomatoes, cotton, rice, and lettuce under reduced tillage regimes (Madden et al. 2004; Mitchell et al. 2007; Venterea et al. 2005; Linquist et al. 2008; Doane et al. 2009). These four crops are cultivated on more than 600,000 ha, an area equal to roughly 20% of the cultivated irrigated farmland. Yet, the area cropped, while rising rapidly, using conservation tillage, was less than 1% in the mid-2000s (CTIC 2004) suggesting a significant expansion potential. And it seems that potential is being capitalized on. More recent statistics indicate nearly 1 million acres of farmland are under conservation tillage in California (Warnert 2012). Even though only a small fraction of croplands meet the requirements to be considered conservation tillage, expert accounts suggest producers throughout California appear to be reducing tillage intensity, especially in the San Joaquin Valley (D. Munk, personal communication).

Based on the available data for California soils, climate, and crop, we conclude that the value of conservation tillage in mitigating N_2O emissions specifically, or climate change more generally, is still speculative, with some conflicting results. Conservation tillage, however, is multifunctional and consideration of climate regulation in combination with other co-benefits warrants increased consideration of this practice.

A7.4.2 Applying organic wastes

Applying organic waste products—manures, composts, and urban green wastes[13]—changes many features of the soil environment, largely for the better. Most importantly, these amendments add organic matter (SOM) to soils. Increased SOM improves aggregation and aggregate stability, which helps drainage, infiltration, and overall tilth—bulk density, porosity, and hydraulic conductivity (Wander et al. 1994; Rosen and Allan 2007). Microbial biomass and labile pools of soil organic C and N also increase with organic amendments (Drinkwater et al. 1998; Poudel et al. 2001). Reserves of SOC and SOM serve as slow-release sources of nutrients and energy for plants and microbes, with the rate of availability depending on the material's quality: C/N ratio, lignin, and polyphenol content (Palm et al. 2001). Use of organic wastes further promotes healthy and active soil microbial communities, slowing the pace of N turnover, minimizing the size of the soil mineral N pool, and in some cases mitigating N fluxes (Drinkwater et al. 1998; Reganold et al. 2010; Burger et al. 2005; Kramer and Gleixner 2006).

Efficient use of organic N in wastes is more complex than managing inorganic N mineral fertilizers. The first challenge is variability in the materials themselves. Organic amendments vary significantly in their N, and C, content. Differences are significant both between types of organic wastes (e.g., beef steer manure versus urban green waste) and within wastes derived from the same type of source (e.g., dairy manure). Of 31 samples of solid organic amendments intended for agricultural use in California, Hartz et al. (2000) found total N ranged between 10 to 47 g per kg among materials and the amount of organic N within the same material category ranged between 16 and 192% for materials with at least 3 samples. Large variation in N composition can be traced to source stock (e.g., animal diets or biomass) and conditions during processing. Without chemical analysis of waste prior to application, nutrient application rate cannot be estimated.

[13]This discussion centers on manures and compost because of their overwhelming dominance of use (416 Gg of manure-N generated by animal production each year alone, nearly 2/5 of the N applied to croplands each year (Chapter 4). In 2007, 258,122 ha of California cropland received manure (USDA Census of Agriculture 2007). Similar concerns are applicable to biosolids.

The second and related challenge has to do with the mineralization rate of N in organic wastes. As mentioned previously, mineralization occurs at variable rates subject to residue quality, environmental conditions (e.g., temperature and moisture), and management (e.g., tillage). These factors interact sufficiently to make SOM become plant available on time scales ranging from days to years, with accurate prediction of release rates requiring advance computation and nontrivial data (e.g., Crohn 2006). In an incubation experiment using California soils, between 4 and 35% of manure and composts were mineralized over the course of 10 months (Pratt and Castellanos 1981). Growing seasons are often shorter in length and thus these results likely overestimate mineralization under typical production conditions. In four months, only an average of 11% of N was released for manures, 6% from composts containing manures, and 2% from composts composed of urban wastes (Hartz et al. 2000). To account for slow release, users of organic N end up having to apply rates well in excess of plant N demand, at least until soils reach an equilibrium where rates of mineralization equal N additions (Pratt 1979; Pang and Letey 2000). Although here we illustrate the issues with solid materials, similar concerns complicate the use of liquid manure, common practice in Central Valley dairies (Feng et al. 2005). More homogenous, faster releasing materials are available (e.g., seabird guano, blood meal, and fish powder); however, cost limits their use in commercials settings (Hartz and Johnstone 2006).

Will using only organic N compromise productivity? This issue is very much debated. Some studies show yields are lower than conventional (e.g., Reganold et al. 2010; Jackson et al. 2004) when equivalent amounts of N are applied in part, presumably because much of the N contained within organic sources is not immediately plant available (Rosen and Allen 2007). Others suggest yield differentials are rarely apparent, however (Badgley et al. 2007; Drinkwater et al. 1998; Reganold et al. 2001). The most recent meta-analysis suggests yields of cropping systems using organic versus inorganic materials were between 9 and 35% lower (Seufert et al. 2012), though many factors unrelated to fertilizer type may affect the productivity of the systems. Research results form California annual cropping systems demonstrate comparable yields can be achieved with intensive management. Over five years, yields of an organic rotation were similar to those from a conventional 2 year tomato-corn rotation (71 Mg per ha), both of which were slightly below average statewide yields over the same time frame (77 Mg per ha) (Poudel et al. 2002). Results such as these coupled with average yield ratios comparing organic and conventional (see Box 7.1) suggest that production N systems using organic N are less productive.

But is using organic N amendment more environmentally friendly than using conventional inorganic N sources? Conflicting results permeate the literature. Because applying organic wastes adds C and builds SOM, N tends to remain in the soil for a longer period. Drinkwater et al. (1998) suggests the use of organic waste decreases leaching by nearly 50%. One report demonstrates that by stimulating the active denitrifier community, there was increased N_2 emissions in organic plots which leached 4.4 - 5.6 times less NO_3^- than conventional plots (Kramer et al. 2006). Wang et al. (2008) show that 77% less NO_3^- was leached from a rotation of cantaloupe and lettuce on a sandy soil using organic-N than one using synthetic fertilizer. It has been shown that N_2O fluxes peak at greater levels in conventionally managed than organic systems (Burger et al. 2005; Kong et al. 2009). Simulations of N mineralization from poultry manure, corn uptake, and NO_3^- leaching show that rates would have to exceed 600 kg of organic-N per ha to meet crop requirements; at this rate nearly 300 kg N per ha would be leached (Pang and Letey 2000). Applying the same model to common liquid manure management practices (e.g., furrow irrigation with less than 80% uniformity), leaching rates approach or exceed 200 kg N per ha per year when N is applied at 1.4x plant uptake (Feng et al. 2005). Data that account for the difference in levels of N input and differences in levels of production suggest similar degrees of NO_3^- leaching per unit applied and output from organic N and inorganic (Kirchmann and Bergstrom 2001; Kirchmann et al. 2002). There is also little evidence that direct emissions of N_2O from manures and composts differ significantly from synthetic fertilizers. Compilation of available data show that emissions from organic sources are approximately similar, if not greater than inorganic sources, 1-2% of N applied (Bouwman et al. 2002; IPCC 2007).

Use of organic wastes in California is constrained by logistical and health concerns. The economics of transporting bulky organic N containing materials limit the distribution of application. Liquid manure is moved at most about 3 or 4 miles from the place of production while solid materials are transported at most 50 miles, but often much less. More recently, concerns have been raised over the transfer of pathogens in manure. If the manure is not composted adequately, it can contain human pathogens (including *E. coli* H0157). Composting of manure emits much of the plant available N as gaseous emissions of N both reducing its fertilizer value and adding to regional air problems.

Integrated fertility or low-input systems that utilize inorganic and organic N sources may achieve both production and environmental goals. Inorganic N fertilizer acts as a quick release supplement to sustain crop growth until organic N mineralizes, more effectively synchronizing soil-crop nutrient cycles (Kramer et al. 2002). Incremental increases in yield and substantial decreases in emission can result (Cavero et al. 1999; Poudel et al. 2001, 2002).

A7.4.3 Biochar

Biochar is produced during the low temperature pyrolysis of organic residues (plant matter, animal waste) to generate renewable energy. The resulting material is then applied to land as a soil amendment. Although the use of biochar amendments to agricultural soils is receiving increased attention as a method for reducing N leakage while sequestering carbon, improving soil fertility, and increasing water retention in soil (Lehmann 2007), few data are available to evaluate its ability to achieve the proposed benefits and even less to evaluate the mechanism by which it achieves the proposed benefits.

Use of biochar is impeded by the large variation in the materials. Materials sold, distributed, and applied under the "biochar" banner may differ significantly in their absorptive capacity and stabilization properties. Differences in materials arise from the wide variety of chemical composition of feedstock and conditions of pyrolysis. Variation further limits the capacity to predict or understand its interactions with soil processes. It remains to be seen if biochar is another "snake-oil" or if it truly has staying power.

A7.5 Landscape approaches

Not every action to control N emissions must take place within field borders. Emissions, by definition, transfer N across boundaries between environmental systems. It is at the points where two ecosystems interface that landscape approaches change flux potential. Practices implemented at the field boundary or strategically distributed across the landscape can capture, recycle, and transform N prior to its release into the wider environment. Currently, most landscape approaches for N management aim to limit NO_3^- movement from the biosphere to the hydrosphere by sequestration and denitrification.

Managing reactive N at the landscape scale offers a prospect for N control but adds concerns as well. When landscape features serve as sinks for N, sustainable reduction must result in long-term storage of N in the burial of plant materials and sediments. Without storage, impacts are delayed, not mitigated. Soil water and N content in the system is high and thus there is a likelihood of denitrification and N_2O evolution. Unmanaged wetlands generally emit only a small quantity of N_2O (Groffman et al. 1998). But it is once systems are overloaded with NO_3^- from agriculture that they become a substantive source of the greenhouse gas. Use may therefore cause pollution swapping to a limited extent, if denitrification conditions cannot be controlled.

Landscape approaches can be divided into two main categories. The first involves the management of natural vegetation at the field's edge or stream bank. The second comprises engineering solutions. While it seems self-evident, it is worth noting here that the effectiveness of any landscape approach, natural or man-made, to regulate N cycling will depend on its positioning and size. A large poorly sited landscape feature, outside an N flow path, will not interact with sufficient N to make a marked difference. Conversely, biological processes may be overwhelmed if the feature's area is insufficient to treat the influent N load. This reality means features often have to be located on prime farmland, creating additional opportunity and operations costs.

A7.5.1 Manage natural vegetation

Vegetative areas at field boundaries, which can range from simple grass buffer strips to complex multi-strata riparian ecosystems, reduce NO_3^- loading to the environment. Grasses, herbaceous perennials, and trees typically intercept NO_3^- as it moves across the soil surface with sediment and runoff or with their roots during subsurface transport. A meta-analysis of vegetative buffers indicates that the median reduction of NO_3^- was 68.3% but actual reductions varied widely, from 2.2-99.9%, (Zhang et al. 2010). Variation in buffer performance can be attributed to its size and topographic positioning. Accordingly, larger buffers sequester more NO_3^-, up to 88% of influent at 30m. Isotopic N experiments indicate actively growing plant cover is important to maintain and increase buffer capacity, with 2/3 greater NO_3^- uptake when vegetative buffers were managed by cutting than unmanaged system (Bedard-Haughn et al. 2004, 2005). Riparian areas at the edge of waterbodies reduce NO_3^- to similar degrees. Data from 89 studies on 45 riparian areas indicates an average 67.5% N removal rate (Mayer et al. 2007). Riparian zones appear to be more effective at removing subsurface NO_3^- than surface runoff suggesting that aggregate effects of soil type, subsurface hydrology, and denitrification potential may have a large influence on their utility as an N management measure.

Dedicating land for vegetative areas can have its downside though. In particular, it removes land from production, with concordant economic consequences. Vegetative areas may place greater demands for labor because of the need to manage the features, be it mowing or biomass harvesting. In some cases, buffers may increase weed or pest establishment. Thus, vegetative buffers present likely tradeoffs with economics, labor, and agricultural chemical use.

A7.5.2 Construct engineered solutions[14]

Human engineered systems, such as constructed wetlands and denitrification reactors, are designed to process N in influent in much the same way as natural features, relying on processes of uptake and/or denitrification. Their ability to reduce N load of effluent and protect water quality is determined by a large number of site-specific factors, such as the timing, magnitude, and concentrations of nutrient load, and hydrologic properties, such as residence time and thus high variability in efficacy should be predicted (Iovanna et al. 2008). Nevertheless, constructed wetlands and denitrification reactors appear to be effective. In California, O'Geen et al. (2007) studied a 1-year old wetland and a 10-year old mature wetland in the San Joaquin Valley. The newly constructed wetland removed an average of 22% of NO_3^- while the more mature wetland removed 45% (O'Geen et al. 2007). Irrigating pasture tends to produce artificially occurring wetlands in drainage basins. Even at low residence times (less than 2 hours), wetlands in these circumstances are capable of reducing NO_3^- loads by 60% and total N by 40% (Knox et al. 2008).

Recently, development and deployment of "denitrification reactors" has been proposed to reduce the N loading from agricultural runoff, as well waste- and stormwater (Collins et al. 2010). A denitrification reactor is essentially a trench with high C infill, such as wood chips. Nitrate-rich waters transit through the C rich substrate slowly enough for denitrification to take place. Management is key to ensure appropriate denitrification conditions are maintained and remains the largest concern. If operated with low residence times, too high N concentrations, or limited C; denitrification reactors may

[14] Many technologies applicable to agriculture were either developed or are also used for treatment of water from wastewater treatment plants and stormwater.

become a source of N_2O. Substrate must be high in carbon and resistant to decomposition so that denitrification is not limited and the material does not have to be replaced often. As with other landscape approaches, the effectiveness of denitrification reactors to reduce the N in the effluent load can vary based on the C material, residence time, and influent N concentrations (Collins et al. 2010; Schipper et al. 2010[15]).

Only a few large-scale bioreactors are in operation in the US, principally distributed at commercial drinking and treatment facilities (Jensen et al. 2012; King et al. 2012). Bioreactors are an effective technology reducing loading at a smaller scale. Robertson and Cherry (1995) show that bioreactors can treat leachate from 60 ppm to 2 -25 ppm NO_3^- , a removal of 74 – 90%. Recently, they have been shown to be effective for treating effluent from onsite wastewater treatment systems (Leverenz et al. 2010). The technology could also be effective for treating agricultural leachate and runoff from tile drains because runoff N is already in the form of NO_3^- and therefore doesn't need to be nitrified prior to denitrification, as in the case in industrial wastewater treatment. Effluent from field drains at local or aggregate at larger scales may prove to be an option worth exploring.

A7.6 Agrobiodiversity

Biodiversity, and agrobiodiversity[16] more specifically, improves N cycling through altering the pace of N turnover, stabilizing soil N within organic matter, extracting a greater fraction of mineral N from the soil, retaining N in the landscape, and reducing the exchange of N between adjoining ecosystems or among land, air, and water (Brussaard et al. 2007; Young-Mathews et al. 2010; Smukler et al. 2010). It achieves all this through virtually every plausible N control mechanism, from efficiency to transformation.

[15] See Ecological Engineering (2010) volume 36, issue 11 for special issue on bioreactors.

[16] Agrobiodiversity refers to domesticated and non-domesticated species that support food provisioning. This clearly includes plants and animals that are consumed but also pollinators and soil biota that are necessary for production.

Managing for diverse agricultural landscapes, therefore, holds some promise for addressing N concerns in California agriculture. However, significant technical and financial obstacles impede diversifying production systems and their surroundings within their current geometry and technological, institutional, and regulatory envelope.

A7.6.1 Plant green manures and trap crops

Cover crops are plants grown for reasons other than to generate income, with altering soil N cycling being one of the most frequent goals. Cover crops can be grown concurrently with a cash crop, as when they are planted between rows in perennial systems, or between annual crops when fields would otherwise be fallow. In either circumstance, cover crops influence N cycling by changing soil physical and chemical properties after they are incorporated into the soil. Effects ranging from rapid N mineralization and availability to near complete inorganic N immobilization are possible, with the consequences being a function of characteristic traits of the cover crops species (biomass, C/N ratio, N fixation) and environmental conditions of production (length of growing season, temperature, soil moisture) (Drinkwater et al. 1998; Hu et al. 1997; Shennan, 1992). Variation in the potential N cycling impacts and the diverse set of cover crop species and cash crop production systems places a premium on thoughtful species selection when using cover crops. When planted for N utility, cover crops serve either of two opposing objectives and it is important to differentiate between them. Leguminous cover crops add new N to the soil (e.g., green manures) while non-leguminous cover crops (e.g., trap crops) capture and recycle N back to the soil surface.

Green manures are grown to increase the soil N pool in support of cash crop nutrient demand (Patrick et al. 2004; Jackson 2000). Incorporation and decomposition of cover crops material provide soil microbial communities energy to mineralize N contained within the green manure. Cover crops with a low C/N ratio (i.e., <20) ensure rapid decomposition and avoid net microbial immobilization of soil N which would have a potentially deleterious effect on cash crop growth (Wyland et al. 1995). The quantity of N made available is determined by the rate of fixation and biomass production, both controlled by inherent species traits, as well as environmental conditions and length of crop cycle. Shennan (1992) reviewed cover crop for California and found that reported rates of fixation ranged from 56 to greater than 200 kg N per ha. Fixation rates at the higher end of that range are of levels to support nutrient demands of most crops. However, as with inorganic N, uptake efficiency of legume N is generally low—averages about 30% (Crews and Peoples 2005). Part of the inefficiency results from rapid mineralization of N after incorporation, which potentially decreases N supply and crop demand synchrony. In a California no-till processing tomato system, Herrero et al. (2001) found that soil mineral N was higher in systems following cover crop incorporation than application of inorganic mineral fertilizer demonstrating the potential for poor synchronization. As previously discussed, nutrient supply and demand asynchrony increases the risk of leaching and gaseous emissions, although higher emissions do not always result. Crews and Peoples (2005) suggest that legume N in irrigated production may decrease N loss in part because of a greater incorporation of legume-N into SOM. By comparison to inorganic N sources, direct N_2O emissions from leguminous N sources are often reported to be lower, approximately ½ on average (Rochette and Janzen 2005).

Non-leguminous cover crops are used as trap crops to capture inorganic N remaining in the soil following cash crop production. This is important because without actively growing plant cover (e.g., in winter fallow and dormant periods) soil N builds up due to mineralization of plant residues and is particularly vulnerable to loss (Jackson et al. 1994). With the EPIC biogeochemical model, research predicts that leaching of NO_3^- in tomato and lettuce systems can exceed 150 kg per ha following the primary summer production season (Cavero et al. 1999; Jackson 2000). Using cover crops over this period consistently and significantly reduces the size of the NO_3^- pool and pollution potential (Jackson et al. 2003). By capturing and sequestering what would have been lost, trap crops minimize the inorganic N pool and present an opportunity to recycle N into the cropping system upon their decomposition. Crop growth patterns and root density and structure determine a species' ability to extract N from the soil. Because of the differences between crops, strategically designing cropping systems and crop rotations is necessary to achieve a high system N efficiency.

Cover crops offer non-N related benefits as well, such as addition of organic matter, disease suppression, erosion control, and maintenance of beneficial insect population and these co-benefits may drive their use (Ingels et al. 1994). Utilization of cover crops to achieve N cycling objectives in California faces many challenges, however. The most frequently cited issues center on the effects of leguminous N on cash crop yields, cost of implementation, competition with cash crop management practices, and depletion of soil moisture. Data suggest the concerns are well founded for some systems. For example, a meta-analysis of research on replacing fallows with leguminous crops found that yields were only an average of 10% less when using legume cover crop to support cash crop growth instead of inorganic fertilizers (Tonitto et al. 2006) suggesting the potential to partially substitute organic N source for inorganic N. However, short time frames between cash crops limiting total biomass production, depletion of soil water reserves by the cover crop, and costs of establishment and incorporation constrain their current use (Jackson et al. 2003) and future potential.

Because of the physiological differences between crops, pairing the appropriate cover crop with the cropping goal is essential to maximize benefit. (Ingels et al. 1994). The growth habit, flowering period, maturity, and reliability of self-reseeding are a few of the characteristics that are important to consider when selecting the right cover crop. Cover crops grown in annual systems, for instance, may need to be fast growing species to maximize biomass production and N uptake during the short windows between cash crops. In perennial systems cover crops that are strong self-reseeders may become invasive weeds competing for light and soil resources. The tradeoffs cover crops create, some associated with N and some with other factors, hinder their application. Ultimately, successful use requires evaluating the benefits and potential concerns of a cover crop species with the demands of the farming system.

A7.6.2 Diversify crop rotations

Impacts of diversifying crop rotations on N cycling will depend on rotation used, the species substituted, and the management of the crops. It is essential to consider entire cropping system N efficiency. For example, safflower is regularly fertilized with 110 to 170 kg N per ha but it has been shown to produce high yields with minimal addition of N fertilizer relying extensively on residual N in rotation with other crops (Kaffka and Kearney 1998; Bassil et al. 2002). Only if the entire rotation is accounted for will diversifying rotation to include sunflower be beneficial. Unfortunately, crops with significant extractive capacity tend to be of low economic value. With the high costs of land and water in California, the inclusion of such crops is often untenable.

One unique case is when using alfalfa in rotations. Alfalfa is a legume that fixes atmospheric N arresting the need for synthetic N inputs. Unless an 'N credit' is given for N released from decaying alfalfa residues when it is plowed under, the subsequent crop may be over-fertilized (Robbins and Carter 1980). As a rule of thumb, an appropriate credit may be between 67 and 90 kg N per ha but data are limited to outline precise guidelines causing producers to ignore the economic saving (from displaced synthetic fertilizer application) and increase potential pollution concerns.

A diverse array of crop rotations is used in annual croplands of California. Some patterns are widespread (e.g., processing tomato-wheat in the San Joaquin Valley; lettuce-lettuce-cole in the Salinas Valley), while others much less so. Ongoing research documenting rotations in Kern County shows that the 10 most commonly observed rotations account for 48% of cropping patterns (MacEwan and Howitt, personal communication). These data illustrate that while clear patterns are discernable, there is a substantial variation. Deviations in planting decisions are consequences of external drivers, such as market, weather conditions, and availability of water. Current conditions are a good example. High commodity prices are leading to a resurgence of cotton production in the San Joaquin Valley after years of decline since 2005, likely displacing area previously converted or planned for other crops.

Diversifying crop rotations in California croplands has only limited potential to address N dynamics under current California cropping conditions. Even in the highly profitable organic vegetable market, on-farm crop diversity quickly decreased from 17 to 3 crops over three years (Smukler et al. 2008). Furthermore, a significant portion of the annual forage in California is associated with the dairy industry which is dependent upon affordable feed. Changing cropping practices and potentially minimizing forage acreage may increase feed prices to unsustainable levels. Economic costs and benefits of farming coupled with little environmental regulation do not incentivize to change rotation patterns and thus there are few examples where change might take place. However, annual crop rotations are responsive to external stimuli and present opportunities to alter N cycling throughout the state on a season-by-season basis. Although cover crops may impart agronomic and environmental N benefits, using cover crops present a number of cropping concerns. Depending on the species and cropping conditions, cover crops can deplete soil moisture, decrease plant available N, increase weed nuisance, harbor pests, and change microclimates, which may lead to frost damage to perennial crops (Ingels et al. 1994). Furthermore, cover crops require increases in management costs that include costs of seed, energy, and labor. These challenges are often cited as reasons for lack of adoption (Jackson et al. 2003)

A7.6.3 Enhance soil biological activity and diversity

Soil animal and microbial diversity is part of the biological resources of agroecosystems and thus managing their activities should be considered as part of the N management portfolio. It is clear that microbial communities control soil N cycling. Soil bacteria determine the pace of N cycling where most N transformation processes are direct results of the activity of these microorganisms including, denitrification, nitrification, immobilization, and fixation. Through these processes, soil fauna affect the rate of N reactions, effectively manipulating the size and duration of soil N pools (Drinkwater et al. 1995). In addition to the effects on chemical composition, soil organisms affect physical composition and structure of soils, which changes gas diffusion and hydraulic properties. At the same time, soil biota is affected by N availability. When soils are low in available N, fungal communities dominate. In contrast, bacterial communities tend to dominate soils with significant quantities of N available.

Management decisions can influence soil biodiversity directly or indirectly. Yet, few approaches aim to directly manipulate soil biodiversity and behavior. Corkidi et al. (2011) demonstrate the potential value of such approaches. The authors analyzed leachate from containers growing three common nursery crops and found that the NO_3^- and NH_4 concentration of that leachate from pots inoculated with arbuscular mycorrizae was up to 80% lower. Alfalfa producers directly enhance soil microorganism as well. Prior to planting a new stand of alfalfa, soils are often inoculated with *Rhizobium* to promote symbiotic N-fixation.

More often, however, soil communities are managed by the indirect means of modifying their environment. Management practices, as discussed above, will each have an effect on the chemical properties of the soil environment, such as pH, oxygen, N, and C availability. Changing conditions has the capacity to change microorganism diversity, and favor or suppress the activity of soil microorganism diversity, with substantial effect on C stabilization and N cycling (Six et al. 2006; Brussaard et al. 2007).

Whilst the functions of soil biodiversity are beginning to come into focus (e.g., Wardle et al. 2004), there are not many mechanisms to translate that knowledge into practical applications for today's current agricultural systems (with at least one exception - use of arbuscular mycorrhizae in plant phosphorus acquisition, Smith et al. 2011). Development and implementation of this approach requires new research into the functional and technical aspects of how it would occur in the field. Thus, it is unlikely to be a significant factor in helping California better manage N use or reduce saturation anytime soon. Active management of microorganisms is the foundation of N treatment in other industries and public health concerns, e.g. wastewater treatment, however. A first step would be to identify the plausible opportunities that could work at the field scale.

A7.7 Genetic improvement

A7.7.1 Improve crop genetic material

Nitrogen use efficiency in plants is a function of the efficiency of uptake (recovery efficiency) and the efficiency of utilization (physiological efficiency). Genetic traits determine a species N demand, ability to recover soil N, and how well it utilizes it once it assimilates it. Not until recently has N use efficiency become a subject of interest for plant breeders. Prior, other desirable traits were the objects of selection (e.g., disease resistance, yield, or product quality). The consequence has been, in some cases, an inadvertent selection against N use efficiency. For example, plant's ability to explore the soil and uptake N is determined by its root system architecture. The root architecture depends on the species but significant intra-specific variation of rooting depth, density, and branching has been documented (de Dorlodot et al. 2007). Commercial lettuce cultivars maximize development of the head, or shoot, at the expense of a vigorous root system. The small root system restricts the plant's ability to excavate N and water (Burns 1991). Producers, in turn, must manage N for a crop that requires N in very significant quantities with a root system less than the size of a football by timing inputs, a near impossible task. Notice of the agricultural N-related resources degradation has prompted new research aimed to genetically maximize N use efficiency (NUE) (Hirel et al. 2007).

Genetic improvement of crop plants may contribute significantly to addressing N concerns in California croplands in the short to medium term, less than 20 years. This is, in part, because information on the processes controlling NUE in plants is still yet fragmentary.. Recently, application of molecular tools has contributed to the more complete understanding of many underlying processes, such as: N transport, enzymatic reaction, and function (Good et al. 2004). Although mechanisms of internal plant N utilization and recycling have been better described recently, rarely has genetic improvement produced greater yields with less N. Genotype by environment interactions are common demonstrating significant plasticity of the trait making experimental selection challenging (Hirel et al. 2007). Phenotypic plasticity underscores the challenge in selecting for high NUE and partly inhibits the translation of results from controlled experiments to field conditions (Hirel and Lemaire 2006). Future gains in crop NUE due to genetic improvement will require experiments that span agronomy, physiology, and molecular genetics.

The principle reason we believe that genetic manipulation can yield results for California soon is simple. The majority of genetic NUE research centers on field crops (rice, wheat, canola, or corn) or model species such as *Arabadopsis* or *Nicotiana*. Lessons learned from these systems may eventually benefit California producers of those commodities; approximately 800,000 ha or 38% of the cropland, which do have a large impact on groundwater NO_3^- contamination. But still greater emphasis examining NUE in vegetables and trees is needed for the effect of genetic improvement to include the bulk of future cropped area.

A7.7.2 Breed animals for high feed conversion efficiency

Feed conversion is the amount of feed required to produce one unit of product where the product can be eggs, meat, wool, or milk. As feed conversion efficiency improves, less feed is required per unit output. This translates into a reduced need for farmland to grow feed inputs as well as reduced nutrient excretion (manure). Genetic improvement provides one way to improve feed conversion on livestock and poultry farms.

Genetic improvement of farm animals has historically improved feed conversion, produced higher yields more rapidly, and resulted in less manure generated. The most significant advances have perhaps come in broiler breeding. Comparison of the Athens-Canadian random bred control (ACRBC), a common breed from the late 1950s, and the Ross 28 broiler, current breed, provides evidence of the potential benefits (Havenstein et al. 2003a,b; Cheema et al. 2003). The Ross 308 broiler on the 2001 feedstuffs was estimated to have reached 1,815 g body weight at 32 d of age, whereas the ACRBC on the 1957 feed would not have reached that body weight until 101 d of age. The shorter age to market resulting from improved feed conversion would require far less feed input (and associated land to grow the feed) to achieve similar product and have markedly less manure output. Comparisons of carcass weights of the Ross 308 on the 2001 diet versus the ACRBC on the 1957 diet showed they were 6.0, 5.9, 5.2, and 4.6 times heavier than the ACRBC at 43, 57, 71, and 85 d of age, respectively. The authors attributed that 85% of the improvement in feed conversion. Improved performance has come at a cost. Concordant to increased growth rates, there has been a decrease in the adaptive immune responses (Cheema et al. 2003). Dairy production has also benefited from genetic improvement of animals. By one estimate, 57% of the increase in milk yield between 1957 and 1997 in the US was the result of better genetics (Cassell 2001). Nation-wide genetic improvement has led to fewer dairy cows, less feed, and less manure while supporting the demand for dairy products (Capper et al. 2008).

The potential for genetic improvement to yield additional benefits for managing N in animal production is not significant in the short term.

A7.8 Animal nutrition and feed management

Protein nutrition influences productivity, profitability, and the efficiency of N use in cattle and poultry production systems. Production of milk, meat, and eggs are correlated with crude protein intake (Kebreab et al. 2001; Bailey at al. 2008; Sterling et al. 2002). It is important to supply protein in sufficient quantities to support growth and development. When diets are formulated for specific protein and amino acid requirements, bioavailability of N and assimilation improve (Powell et al. 2010; Vandehaar and St. Pierre 2006; Huhtanen and Hristov 2009; Nahm 2002). Consequently, an increase in resource use efficiency takes place.

Feed utilization efficiency has multiplicative impacts on N cycling within the animal production unit and croplands. The amount and form of N excretion is influenced by the type and degradability of protein and energy source in the diet. For example, increasing the energy concentration of the diet and using low degradable starch sources such as corn in concentrates could reduce not only the total amount of N in excreta but also the proportion of N in urine (Kebreab et al. 2002), which in turn reduces ammonia emissions.

Feed utilization efficiency also decreases the total demand for animal feeds (assuming livestock production remains constant). Coincidentally, N emissions from feed production and transportation are reduced. At the same time, less N excretion takes place reducing the disposal/recycling burden on land and emissions. Meyer and Robinson (2007) provide an illustration of the benefits of feed management on manure handling. The authors inventoried feed stuffs and feed management at seven dairies in California and found that dairies operated between 16 and 27% N utilization efficiency. That means that for every 1,000 kg of N fed, 840 kg of N are excreted at the least efficient dairy while only 730 kg of N for the most efficient dairies. The consequence is that the less efficient dairies require 15% more land for N application or that the more efficient dairy could milk 15% more cows with the same amount of land assuming the same application rate and efficacy of organic N use. With manure handling practices remaining the same, less N excretion could potentially reduce emissions because most emissions are in part related to the amount of N excreted.

With more than 2.4 million cattle and 350 million birds on feed year-round and up to 2.6 million cattle on supplemental feed in California, feed management presents considerable potential for reducing direct and indirect N emissions due to California's animal feeding operations. But the magnitudes of the benefits are hard to characterize because few data are available to evaluate animal feeding practices in California. Because of this, the discussion here will be restricted to cattle. Castillo et al. (2005) surveyed feed management practices on 51 randomly selected dairy operations in Merced county and found crude protein contents of lactating cows diets averaged 17% $\pm$ 1.19 (SD). This finding suggests that the average operation is not overfeeding N; the National Research Council (2001) recommendation for crude protein consumption in lactating dairy cows is 16.5%. Precision feeding of N is the matching of crude protein with physiological requirements. Castillo et al. (2005) survey demonstrates that the dairies feeding more than one diet had higher N utilization and dairies feeding three and four diets had statistically significantly higher N utilization than those feeding uniformly (Figure A7.1).

[Figure A7.1]

However, feed management rarely accounts for the differential requirements of animals during various points in their lifecycle well. Calves, dry, and lactating cows demand a different amount of crude protein. If fed the same diets, that is only altering dry matter intake, overfeeding of N results causing increased N excretion. Recognition of the variable needs of cattle has led to calls to increase staged or precision feeding (Meyer and Robinson 2007). Most animal operations formulate diets to provide minimum required nutrient concentrations at the least cost. Because protein is among the most expensive ingredients, their use is generally tightly monitored. Despite close attention, N is sometimes fed in larger quantities than required to meet physiological demand. This is especially problematic with low cost by-product feeds, which are often of variable composition (DePeters et al. 2000). An increasingly important concern is the use of distiller's grains as a feed. Distiller's grains are a byproduct of ethanol production and are commonly fed to cattle because of their low cost and high nutrient concentration, which tends to be two to three times as high as unprocessed grains (Belyea et al. 2004). Without reformulation, diets quickly exceed N assimilatory capacity of the animals and excess N is excreted. Hao et al. (2009) shows that NH_4 composition of manure increases with increased consumption of distiller's grain.

Feed management includes the use of dietary additives to enhance production. The additives may be yeasts, enzymes, microbials, ionophores, or proprietary materials. Some additives are well researched, and their mode of action is well defined. Other additives have undergone less rigorous research and little is known of their efficacy in the animal or their subsequent impact on the environment. The most widely researched and publicized supplement is rBST. Some evidence indicates that this hormone decreases the protein requirements for maintenance and lactation by 3.2% and N excretion by 9.1% per kg of milk production (Capper et al. 2008). However, consumers have raised concerns over its use and subsequent transmission into the food supply. Less than 10% of the milk produced in California uses rBST and its future use is expected to continue to decline (D. Meyer, personal communication). Additives and supplements have been important in reducing the environmental impact of poultry production. Gains are the consequence of widespread feeding supplementation. Addition of amino acids and growth promoting substances resulted in reduced N excretion between 5 to 35% in poultry depending on the feeding strategy (Nahm 2002)

When considering feed management/NUE of California animals, it is important to remember that the role of animals in the broader agricultural ecosystems and the impact it has on diet formulation. California cattle and dairy cows, in particular, serve an essential recycling function. A significant fraction of their diets can be derived from consumption of agricultural byproducts, with variable and often less known N concentration. In this way, they concentrate and consolidate N from agricultural industries throughout the state (DePeters et al. 2000). Without them, a significant amount of N would have to be handled, processed, and disposed of by other means. Furthermore, ethanol production creates access to cheap protein (N) source, distiller's grains. Use of this feedstuff complicates diet formulation due to the near double N content of unprocessed grains increasing excretion and emissions (Hao et al. 2009, Chapter 7).

A7.9 Manure management

Manure management typically refers to the practices used to handle animal waste following excretion. In fact, planning for manure nutrient recycling and disposal should begin prior to excretion, with protein management. But here, we restrict the discussion to the methods for handling manure N itself and discuss it within the context of manure management trains—collection, storage, treatment, and land application[17]. Understanding the process underlying the individual component practices is important; however, manure handling requires sets of practices to conserve manure N for land application and thus, in practice, a whole farm approach is necessary if emissions are to be controlled (Castillo 2009; Powell et al. 2010). It is precisely because of this reason that practices that do not necessarily change N characteristics but do enable greater management capacity of manure N, such as liquid-solid separation, are discussed.

A7.9.1 Collect manure more frequently

Manure collection in animal feeding operations aggregate N for storage, treatment, and later application to crop fields. Collecting manure more frequently after it is deposited in barns and open lots will almost certainly decrease N emissions, although data are generally insufficient to quantify the extent. Reductions result from moving the fecal and urinary N from a location with an environment amenable for NH_3 volatilization to one where chemical and physical processes are more easily

[17] This discussion draws heavily on the recent stakeholder process, "An Assessment of Technologies for Management and Treatment of Dairy Manure in California's San Joaquin Valley" and we recommend this publication as further reading for those interested in these issues (TFASP, 2005). Additional discussion on land application of manures can be found in the section on using organic wastes.

manipulated to create less hospitable conditions. Frequent flushing in freestall barns transfers the highly volatile urinary N into anaerobic conditions (lagoons) where pond pH and depth determine volatilization rates (Mukhtar et al. 2012). Since dairy operators flush freestalls with recycled lagoon water (rich in NH_4), increased flushing frequency may cause a marginal amount of additional volatilization. The increase is likely negligible and far outweighed by removing the manure more rapidly from the barn surface. Frequent removal of manure helps control emissions from solid manure too. Corrals, open lots, and poultry houses are vulnerable to volatile, and somewhat susceptible to leaching, losses because of the high rates of N excretion, concentrated spatial distribution of urine and feces, and constant mixing of the soil surface by animal movement. (Chang et al. 1973; Hristov et al. 2011; Xin et al. 2011). Frequent removal to longer-term storage and treatment processes (i.e., composting or dying) decreases the emissions from housing areas; however, the larger N load transported into other components means there is an elevated risk of emissions from these farm components (Rotz 2004).

Economic, operational, and regulatory considerations constrain the frequency of manure collection in California. Manure is bulky and heavy. Moving it, even over short distances, represents a significant undertaking. More regular collection will increase demand for labor, fuel, and machine time decreasing net profits. Even if the costs were not limiting, infrastructure restricts the rate of manure collection at many animal feeding operations. Storage and treatment facilities (e.g., lagoons, solid-liquid separators, drying pads) have a finite capacity and often operate near their limits. Structural expansion may be necessary to accommodate additional volume due to greater collection regimes. Economic and operational concerns aside, current and impending regulations for N and other pollutants dictate collection practices that may be complementary or antagonistic for N control. For example, dairy farmers in the Central Valley are already required to collect manure one to four times daily to control volatile organic compounds (VOC) (Stackhouse et al. 2011). The effect of more frequent manure collection on NH_3 volatilization is unknown, but the potential tradeoffs or synergies illustrates the need to consider multiple pollutants jointly.

In spite of the potential downstream emissions pressure and the functional challenges, more frequent collection would likely have net benefits for environmental N pollution. At this time, it is impossible to know the magnitude of the impact for the environment or for farming practices and economics.

A7.9.2 Nitrification inhibitors

Use of nitrification and urease inhibitors to control gas emissions has received increased attention recently (see discussion on enhanced efficiency fertilizers above). The chemical compounds that arrest or retard N transformations in soil have been tested on feedlots and in poultry houses. In both situations, urease inhibitors have proven effective to reduce NH_3 emissions. Parker et al. (2005) applied it in beef feedlots and documented 49% to 69% reductions in NH_3 depending on the rate of application. But the relative efficacy is temporary, lasting only 7 to 14 days in one study (Singh et al. 2009). Nitrification inhibitors can also reduce N_2O emissions from both fertilizers and manure (Akiyama et al. 2010; Dittert et al. 2001). Akiyama et al. (2010) report that nitrification inhibitors reduce N_2O emissions from N fertilizer by an average of 38% across a wide range of inhibitor chemicals, N sources, and land use types. Likewise 3,4-dimethylpryazole phosphate reduced N_2O following manure slurry applications by 32% (Dittert et al 2001). Use of nitrification inhibitors in manure management systems of California is extremely limited, likely due to cost and climate. However, there is no research on when and where and how they might be effective for California producers.

A7.9.3 Separate solids from liquids

Solid-liquid separation systems are designed to divide manure by the phase of the material. The purpose is to segregate the manure into more homogenous components, in both form and constituency. Handling and treatment of individual fractions can then be specifically tailored for its composition and characteristics more easily. Liquids can be transferred more readily through the system without clogging pumps and pipes. Solids can be scraped, composted, applied as bedding, and potentially manifested off-site. Because the form of the N in the solid and liquid fractions of manure differs, with solids containing mostly organic N which is bound to C and more stable in the environment and liquids containing mostly urea and NH_4 which is highly reactive and vulnerable to volatilization, operators can take advantage of nutrient value and control future N dynamics more readily. In short, separation enhances manageability.

Multiple factors affect division of the solid from the liquid fraction. Inherent system properties—such as flow rate, characteristics of manure, particle size and nutrient load—influence the relative distribution of N in effluent and solids (Zhang and Westerman 1997). Meyer et al. (2004) evaluated the efficiency of a “weeping-wall” separation system in California and found no significant reduction in the N between the influent and effluent; the N remained in the wastewater. A recent study on a Texas dairy using a two-chamber gravity separation system shows a minor reduction of 10% less N in wastewater effluent (Mukhtar et al. 2011). Mechanical separators, by comparison, separate a greater fraction of the N into solids. Data suggests mechanical separators separate as much as 51% of total Kejdal N into solids, but particle size governs the actual efficacy (Zhang and Westerman 1997). As one might expect, mechanical separators are less capable of transferring N contained in smaller particles. Addition of various chemicals to wastewater enhances solid and liquid separation. Synthetic polymers (flocculants) react with fine particulate to coagulate which then settle over time. Common flocculants are often related to polyacrylamide (PAM) which has also been used in irrigated cropland to reduce runoff of sediments and nutrients (Barvenik 1994). Experiments have demonstrated their effectiveness for aggregating N into the solid manure (Hannah and Stern 1985). Zhang et al. (1998) show that adding ferric chloride and a polymer to dairy manure in California can remove 67 to 69% of N from liquid.

Sedimentation basins and mechanical separation systems are common practice on California dairies (Meyer et al. 1997). More than 63% of dairies used some form of manure separation technology in 2007 (Meyer et al. 2011). Manure separation with sedimentation basins, mechanical separators, flocculants, or a combination of the practices provides greater control over manure N. At production scale, separation creates burdensome requirements for labor and equipment. Refining and cleaning the equipment and the basins requires intensive management, with the management intensity being correlated with technology sophistication. However, current levels of adoption suggest utilization is practically feasible for operators. More detailed information will be needed to optimize their utilization and understand their benefits for N cycling.

A7.9.4 Compost manure solids and other organic materials

Composting—the anaerobic digestion of wastes—stabilizes N contained within organic wastes by transferring it into soil organic matter, where it less available to soil microorganism and hence vulnerable to loss. Although often ignored, even under ideal composting conditions a fraction of the N in the compost is released as NH_3 and N_2O during biological immobilization and through chemical reactions and thus composting can contribute to atmospheric and climate concerns (Ahn et al. 2011). The fate of N during waste composting is subject to the physical and chemical composition of the compost pile: aeration, C/N ratio, moisture, pile structure, pH, and temperature. Through modification of these variables, facility operators can control the rate of digestion. Differential management changes the physical properties of the pile and by extension, N emissions. Evidence suggests that N_2O emissions are nearly double in turned windrows than in static piles, 2% versus 1% of N (Ahn et al. 2011). Increased emissions are possibly the result of redistribution of N throughout the pile and greater gas diffusion. The multitude of driving factors and control environment suggest there are likely opportunities to conserve N in composts by changing management.

Composting represents an important component of California's N cycle. It is one of the fundamental steps prior to recycling nutrients in organic wastes to land. Manures and urban green wastes are already widely composted throughout California, with the vast majority (77%) of composting facilitates using turned windrows (TFASP 2005). Despite the uniformity of method, individual composters manage the piles to different degrees. That suggests improved compost pile management may provide an opportunity to mitigate N emissions.

Appendix 7B: Supporting material: Explanation of calculations and evaluating uncertainty

B7.0 Introduction

The changes to California's nitrogen cascade expected from adopting the strategic actions in Chapter 7 are summarized in Table 7.6. Here, we explain the calculations underlying these values.

Estimates are generally calculated as the difference between the baseline N flows established by *A California nitrogen mass balance for 2005* (Chapter 4), and technically feasible relative changes set by research. Whenever possible, estimates of the expected changes with improved practices rely on data and emissions factors derived from California-specific peer-reviewed, grey literature, and novel data compilations completed as part of this assessment. A few calculations, however, require emissions data for which California-specific studies were unavailable. Under those instances, we applied the most widely accepted values. A list of the emissions factors used and their sources can be found in Table B7.1.

[Table B7.1]

B7.1 Agricultural nitrogen use efficiency

B7.1.1 Crop production[18]

Measures of nitrogen use efficiency (NUE) are ratios of the amount of N assimilated to the amount applied. Assuming output remains constant, increased NUE will result in less N fertilizer applied. We compiled data from published and unpublished research results to estimate N use efficiency by partial

[18] Raising NUE on croplands affects indirect emissions from fertilizer production and transport. But it also has concordant impacts on N_2O emissions and NO_3 leaching and thus this section also describes methods used to calculate reduction potentials for "Nitrate leaching from croplands" (Section 8.2.2.2) and "Greenhouse gas emissions from fertilizer use" (Section 8.2.2.3).

nutrient balance (PNB) for the 22 most economically important California crops[19] (Table B7.2). After eliminating the zero-N and excessive N treatments common in N rate trials, we used the median values for yield and N application rate as reasonable benchmarks as the potential PNB with improved practice. We ignored the low and high N rate treatments because of their potential to bias the median value. Data on fertilizer application rates by crop was taken from Rosenstock et al. (2012) to create a weighted average of N use for each crop group and then used the USDA acreage of aggregated crop groups to estimate the total potential change. Avoided emissions due to the reduced N fertilizer use are discussed in relevant sections.

[Table B7.2]

B7.1.2 Animal production

The capacity to improve feed N efficiency in California animal production was based on two studies of feeding practices in dairies. Surveys of six and fifty-one dairy operators have separately been conducted, one in each of the two major dairy producing areas of the state (Tulare and Modesto), respectively. Their results are consistent with each other, average milk N utilization efficiency equals 23% and efficiency ranges between 27% to 30% for more efficient producers (Meyer and Robinson 2007; Castillo et al. 2005). The difference between average and more efficient producers suggests a conservative estimate of the potential to raise milk N utilization efficiency in dairy production is four percentage points.

[19] PNB is used in lieu of other measures of NUE because of the need to compare statewide average data to research results that may or may not have been specifically designed to test NUE. PNB does not discriminate between the source of N, be it from soil mineralization or fertilizer. Thus, the values derived from literature may be an overestimate if soils were fertile or underestimate if they were not. Regardless, a basic premise of sustainable nutrient management is to balance nutrient exports with applications.

We assumed that changing N utilization efficiency will not affect milk yield and milk N concentration is reasonably constant. Therefore we can calculate the change in efficiency on feed N demand by the following

TMN = FN * FE,

where, TMN equals total milk N, FN equals the feed N, and FE equals the feed N efficiency. By this equation and the assumptions about milk N concentration and milk N yield, a four percentage point increase in N utilization efficiency would reduce feed N demand to 85% of current levels.

We then calculated the potential changes in the N cascade from reduced feed demand. We assumed dairy cows eat a diet consisting of 50% legume and 50% grain. Only non-leguminous feed crops were assumed to be produced with fertilizer. Crops produced with fertilizer were produced with an NUE of 45% to calculate fertilizer applied[20]. Leaching and gaseous emissions were calculated based on N applied for crops receiving fertilizer and a ratio of 30 kg N per ha leached for 360 kg fixed N[21].

We assumed the data and methods developed for dairy cows are applicable for all animal production systems and hence total feed N demand in California. Extending the results from dairy production systems to all animals is reasonable for two reasons. One, feed requirements of dairy cows dominate total feed N in the state, accounting for 81% of total demand. Two, and perhaps more importantly, the target change in milk N utilization efficiency was low, four percentage points, and likely represents a lower bound for other animal production systems in the state. For example, Nahm (2005) suggests that a greater than 60% N utilization efficiency is achievable in poultry, yet California mass

[20] NUE based on California specific data described in Chapter 3. Average NUE values in California are generally similar to those found in other regions (Cassman et al. 2002).

[21] 30 kg N leached is based on average leaching values in alfalfa measured in tile drains of California by Letey et al. (1979). 360 kg per ha is average production on alfalfa (Putnam et al. 2005).

balance calculations and research of similar systems from other locations suggest N utilization efficiency is less than 40% (Chapter 4; Neijat et al. 2011).

B7.2 Ammonia volatilization from manure

With increased feed N efficiency, N excretion will decrease. We applied a simple linear equation developed in a California feed study to estimate the changes in N excretion with changes in efficiency. The impact of decreased feed intake on excretion was calculated following Castillo et al. (2005):

N excretion = 0.9*N intake – 89

A certain degree of volatilization from manure is inevitable, but manure management practices have a large impact on the quantity released (Rotz 2004). The Committee of Experts of Dairy Manure Management[22] (Chang et al. 2005) estimate emissions in the Central Valley by three different methods. By their measures, total volatile emissions from the production area and land application combined are likely to be between 25% and 50% of total excreted N[23]. The emission rate from any single operation may occur throughout this range and it is reasonable to assume that either the extremely high and low emission rates occur less frequently. Therefore, the distribution of volatilization rates across the near 2000 diaries in California can be approximated by a normal distribution with mean 37.5 and standard deviation 6.75. That means, that emissions rates for 95% of the operators will fall within 25% to 50%, with most operations emitting around 37.5% of excreted N as NH_3. It also means that producers

[22] Volatilization is correlated with N excretion (James et al. 1999; Oenema and Tamminga 2005). Estimates are again based on dairy production because it is responsible for the vast majority of manure N and by extension manure derived atmospheric NH_3. Emissions per unit of product for cattle on feed and poultry are likely to be higher and lower, respectively because of efficiencies and excretion.

[23] The findings suggest that a California dairy emits 20% to 40% of excreted N as NH_3 from the production unit itself. Field emissions of 10% to 20% of NH_4-N or 5% to 10% excreted manure N, assuming 50% of the N in manure is in this form.

operating in the top quartile of production emit approximately ≤ 33%. Since 25% of the operators are already operating above this level, we presume that it is technologically feasible to shift the total distribution in two ways. One, improve management that shifts the mean emission rates from 37.5% to 33%. Two, narrow the range of emissions by reducing the standard deviation to 3.5. The latter shift means that 95% of producers will volatilize between 26% and 40% of excreted N (Figure B7.1). Narrowing the distribution in this way assumes that there is a theoretical limit for potential best practice (approximately 25%) and excessive emitters have the greatest potential to improve (roughly 20% decrease) (Figure B7.1).

[Figure B7.1]

We then created a simulation program, coded in the statistical program R, that estimated manure N volatilization. First, we randomly sampled each distribution one time for each dairy in the state to obtain the projected emissions rates for each dairy. Second, we multiplied each rate by the amount of manure N produced at an average dairy[24]. Third, we subtracted the N volatilization from the improved from the original practice. We repeated this program 5000 times to obtain a mean and range of potential values of NH_3 emissions reductions.

B7.3 Nitrate leaching from croplands

Nitrate leaching reductions equal the sum total of avoided NO_3 leaching losses from improved management of inorganic and organic N sources on croplands.

$$NO_3\text{-}N = [N_F * \Delta PNB_{INORG} * EF_{CA\ NO3\ INORG}] + [N_{MAN} * \Delta PNB_{org} * EF_{CA\ NO3\ ORG}]$$

where,

[24] For simplicity, we assumed that dairies all had equal number of cows

NO_3-N = annual amount of avoided NO_3-N losses from croplands

N_F = estimated inorganic fertilizer application on croplands

ΔPNB_{INORG} = change in the ratio of N in crop material exported from the field to the amount of N applied from inorganic sources, expressed as a decimal

$EF_{CA\ NO3\ INORG}$ = NO_3 leaching emissions factor for California with inorganic fertilizer

N_{MAN} = amount of organic manure applied to croplands

ΔPNB_{org} = change in the ratio of N in crop material exported from the field to the amount of N applied from organic sources, expressed as a decimal

$EF_{CA\ NO3\ ORG}$ = emissions factor derived from research with organic sources in California, median of solid and liquid manure assume a 50-50 split in handling

NO_3 = NO_3-N * (62/14)

Inorganic fertilizers: Avoided leaching losses were estimated as the amount not leached due to increased NUE (see above) multiplied by the California-specific emissions factor developed as part of the California Nitrogen Assessment, 34% of N applied, a slightly higher amount than suggested by the IPCC, 30% of N applied.

NO_{3INORG} = ΔNUE * fertilizer N * $EF_{CA\ LEACHING}$ * Molecular conversion

Organic fertilizers: we assume organic fertilizers (animal manures, composts) are currently being applied at an average of 60.5% PNB or 1.65x plant uptake. It is important to note that 1.65x uptake may represent unrealistic goals for agricultural systems using organic N fertilizers; however, this value represents the maximum bound set by the Central Valley Regional Water Quality Control Board (CRWQCB 2010) and thus presents a reasonable baseline for this theoretical discussion/analysis. Emission reductions result from decreasing applications to 1.4x plant uptake or 71% PNB, as follows in the equation:

$NO3_{INORG}$ = Manure N * ΔPNB * $EF_{CA\ LEACHING}$

B7.4 Greenhouse gas emissions from fertilizer use

Nitrous oxide reductions equal the sum total of avoided N_2O losses from improved inorganic fertilizer management on croplands.

N = [N_F * ΔPNB_{INORG} * $EF_{CA\ N2O}$] * Molecular conversion

where,

N_2O = annual amount of avoided NO_3-N losses from croplands

N_F = estimated inorganic fertilizer application on croplands

ΔPNB_{INORG} = change in relative amount of N uptake from inorganic sources, expressed as a decimal

$EF_{CA\ N2O}$ = NO_3 leaching emissions factor for California with inorganic fertilizer

B7.5 Nitrogen oxide emissions from fuel combustion

As part of the Statewide Implementation Plan (SIP) to achieve ozone and $PM_{2.5}$ attainment, CARB developed estimates of potential NO_x reductions. We report their estimates for 2014 for the San Joaquin

Valley and South Coast and 2018 for the Sacramento Valley as measures of potential emissions reductions within the current technology and policy envelope (Table B7.3).

[Table B7.3]

B7.6 Wastewater management

B7.6.1 Wastewater treatment plants

Approximately 90% of wastewater is processed at centralized wastewater treatment plants. Currently, our best estimate is that 50% undergoes N treatment (Chapter 3)[25]. Therefore, we assume that 50% of the total wastewater N load passing through wastewater treatment plants (161.1 Gg N) is treated and is denitrified at a rate of 97%[26] already (78.1 Gg N). A reasonable near-term goal may be a 10% increase in treatment to 60% of influent. From that assumption, an additional 16.1 Gg would be treated, equating to 15.6 removed from wastewater and denitrified to N_2 and 0.5 Gg N released as N_2O. We ignore the indirect emissions from denitrification that occurs in N rich ocean environments (Seitzinger et al. 2006).

B7.6.2 Onsite wastewater treatment systems

Few onsite wastewater treatment systems (OWTS) in use today directly treat for N. Without treatment; it is reasonable to expect minimal, say 4% N attenuation. By comparison, current OWTS designed to remove N achieve 40% N removal rates, at least—but often much higher (Oakley et al. 2010). We calculated the effects of switching to improved OWTS via the following:

$$N_{OWTS} = N_{C\text{-}OWTS} * (1 - R_{b.OWTS})$$

[25] It is not possible with the available data to know what extent facilities equipped with N removal capacity utilize it.

[26] 97% efficacy is used to account for fraction of N_2O produced. One review suggests emissions rates between 1 and 5%, we used the median of 3%.

and

$$\Delta N_{OWTS} = (N_{C\text{-}OWTS} * (1\text{-}R_{i.OWTS})) - (N_{C\text{-}OWTS} * (1 - R_{b.OWTS}))$$

where,

N_{OWTS} = N loading from OWTS

$N_{C\text{-}OWTS}$ = Current N loading (17.9 Gg N, 10% of wastewater N)

$R_{b.OWTS}$ = Removal rate of current systems[27]

$R_{i.OWTS}$ = Removal rate with improved technology

Only a fraction of the systems in use—poorly sited or mismanaged—need to be retrofitted or replaced because of their prospect to degrade natural resources. Currently, the total number of systems needing reconditioning is unknown. We therefore calculated changes in emissions for a range of reconditioning (20%, 40%, 80%) and removal efficacy (40%, 60%, 80%). This provides a quantitative range of the potential emissions reduction that might be expected (Table B7.4).

[Table B7.4]

[27] Note $R_{b.OWTS}$ is assumed to be zero in Chapter 4. Here we assume a limited amount of natural environmental attenuation.

References for this appendix are found in the reference list for Chapter 7.

Box A7.1 Why a qualitative, not quantitative, assessment [Navigate back to text]

The California Nitrogen Assessment takes a qualitative and not quantitative approach to its assessment of individual agricultural management practices and technologies capacity to regulate the N cascade. A qualitative assessment was justified for two reasons. Firstly, California production conditions are unique, both in climate and management. Site characteristics significantly influence the fate of N and the efficacy of any practice. Extrapolation from research from other areas is not necessarily appropriate. With the limited research under California conditions, and even smaller evidence pool when considering the dramatic changes in production in the last 20 years, it is more reasonable to evaluate the potential effectiveness of practices from a theoretical perspective than empirical. The second and perhaps more important reason is that management practices and technologies are not distinct. Interactions among practices make it challenging to quantitatively isolate the effects of a given change in management. Reductionist research can help with this. However, farmers implement practices and technologies in bundles. Multiple factors may be changed simultaneously and have synergistic or antagonistic effects on N flows. And therefore estimates of the impact of a single change are meaningless, in practice.

Figure A7.1 Nitrogen utilization efficiency of 51 dairies in Modesto. Source: Castillo et al. 2005.

[Navigate back to text]

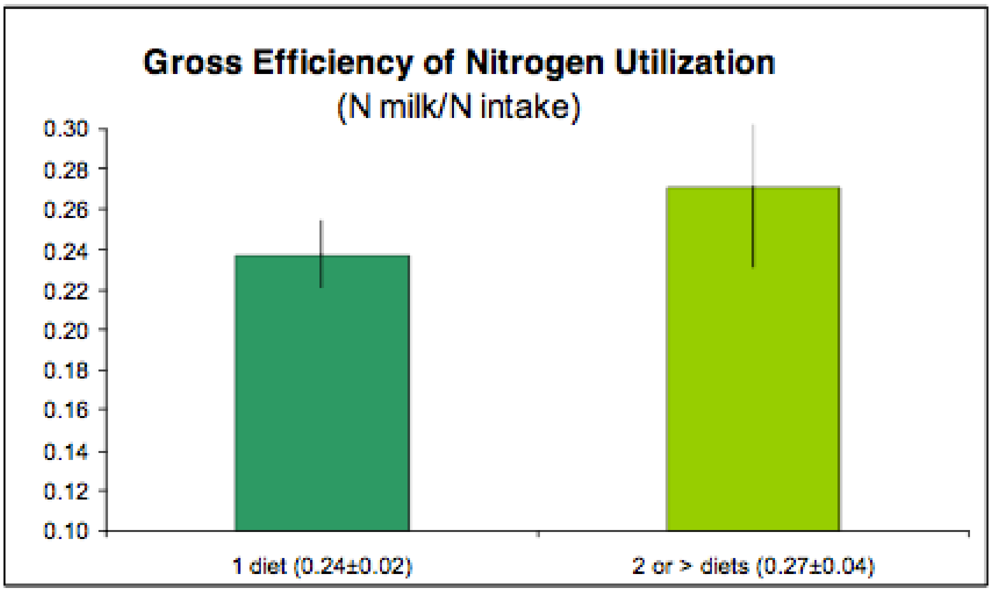

Figure B7.1. Distributions used to calculate potential reduction of NH_3 volatilization from manure handling. Current practice based on dairy production in the Central Valley. [Navigate back to text]

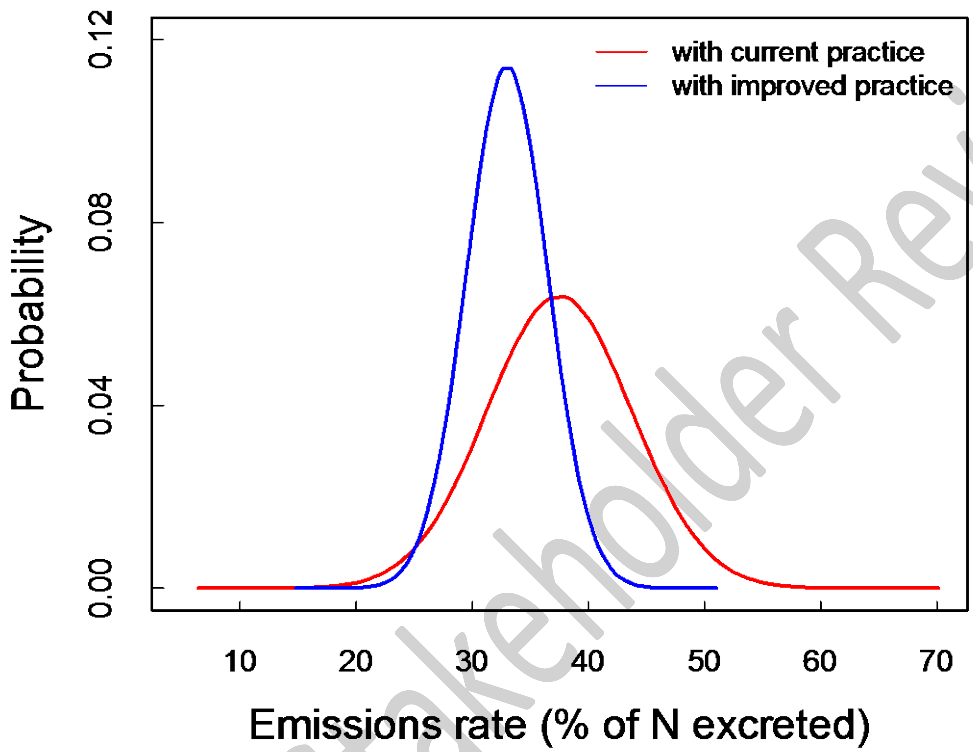

Figure B7.2. R Code to simulate estimated reduction in NH_3 volatilization from manure with improved management practices.

```
ndairies = 2165 #sets default number of dairies in state
Nbase = 404 #sets basline of manure produced Gg N #(Chapter 4)
redN = 0.9 * 418 - 89 #reduces the amount of manure produced based on improved feeding practices
        (Castillo et al. 2006)

#Statistical distributions of manure (see Figure A8.?
cur = rnorm(2000000, 37.5, 6.25) #creates distribution of baseline emissions based on CoC (2005)
imp = rnorm(2000000, 33, 3.5) #creates theoretical distribution of improved practice

#Function for calculating manure production
NH3Manure = function(cur=cur, imp=imp, N=Nbase){
        cur.ran = sample(cur, ndairies)*0.01
        imp.ran = sample(imp, ndairies)*0.01
        totalcurrent = sum(cur.ran*(N/ndairies))
        totalimproved = sum(imp.ran*(N/ndairies))
        list(totalcurrent, totalimproved)}

out1 = replicate(10000, NH3Manure(cur=cur, imp=imp, N=redN))#Function repeated 10000 times
current = unlist(out1[1,])
improved = unlist(out1[2,])

#Function to create summary statistics
summary = function(data){
        x = mean(data)
        y = sd(data)
        z = range(data)
        list(x, y, z)}

#Evaluating summary statistics for current and improved
out2c = summary(current)
out2i = summary(improved)

#Organizes results
outFinal = round(cbind(unlist(out2c), unlist(out2i)), 2)
rownames(outFinal) = c("mean", "sd", "minimum", "maximum")
colnames(outFinal) = c("current", "improved")
```

```
out.range = c((outFinal[3,1]-outFinal[4,2]), (outFinal[4,1]- outFinal[3,2]))

#Plots distributions of current and improved
par(mfrow=c(1,1),mai=c(1,1,1,1), cex.lab=1.5, cex.axis=1.1)
plot(range(5, 70), range(0,0.12), type="n", xlab="Emissions rate (% of N excreted)", ylab="Probability",
        lwd=2, axes=FALSE)
lines(density(cur), col="red", lwd=2)
lines(density(imp), col="blue", lwd=2)
axis(1, c(10, 20, 30, 40, 50, 60, 70), tck=.015)
axis(2, c(0.00, 0.02, 0.04, 0.06, 0.08, 0.10, 0.12), tck=.015)
box(bty="o", lwd=1.5)
legend("topright", legend=c("with current practice", "with improved practice"), lty=1, col=c("red",
"blue"), bty="n")
```

Table A7.1. Resources describing technical options to control the nitrogen cycle from agricultural and non-agricultural sources. [Navigate back to text]

Source activity	References
Fuel combustion	US EPA (1999), Pereira and Amiridis (1995), Skalska et al. (2010)
Wastewater treatment plants	EPA (2008), Metcalf and Eddy (2003)
Onsite wastewater management	Leverenz and Tchobanoglous (2007)
Agriculture	Dzurella et al. (2012), Eagle et al. (2012), Hristov et al. (2011), (2004)

Table A7.2. Major nitrogen cycling processes. [Navigate back to text]

Process	Description	Controlling factors
Mineralization	Conversion of organic N in soil, crop residues or manure into inorganic forms	Temperature, water content
Nitrification	Two step conversion of NH_4 to NO_3 via NO_2	Temperature (< 50 degrees nearly stops), water content, oxygen
Immobilization	Conversion of inorganic N to organic N. Occurs when microorganism decompose materials with high C/N ratio. Decreases plant available N.	Carbon
Volatilization	Release of NH_3 in gaseous form to the atmosphere.	pH, temperature, wind speed
Denitrification	Bacteria convert NO_3 to N_2 gas. Use NO_3 instead of oxygen in metabolic processes in low oxygen environment	Oxygen, temperature, water filled pore space, carbon
Leaching	Downward percolation of NO_3 through soil profile. Physical event where soluble NO_3 moves by mass flow with drainage water	Soil water content, hydraulic conductivity, soil texture

Table A7.3. Strategies to control the release of N into the environment. Source: Adapted from INC (2011). [Navigate back to text]

Control strategy	Advantages	Limitations	Current applications
Improved practice and conservation	Decreases one or more emissions	Education costs, slow adoption, may increase other emission pathways	Tightly coupled water and nitrogen management in cropping systems
Product substitution	Decreases demand for N	Technological concerns, social acceptability	Use of biosolids and urban green wastes on croplands
Transformation	Reduces emissions	May increase other N emissions	Use of biological nitrification/denitification at wastewater treatment plant (tertiary treatment)
Source limitation	Reduces emissions	Requires large changes in societal behavior	Use of carpooling and high occupancy vehicle lanes
Removal	Reduces impacts	Costly, dealing with byproduct of removal is problematic	Treatment of NO_3 contaminated drinking water, selective catalytic reduction in stationary fuel combustion sources
Improved efficiency	Increased output per unit of N, may reduce need if output remains constant	Usually entails significant costs to implement	Feed management in dairy systems

Table B7.1. Emissions factors and agricultural sources used in calculations. [Navigate back to text]

	Emissions factors and sources			
Emission	CA-specific	Source	Global	Source
NH_3 from manure[1]	35% [20, 50]	Chang et al. (2005)		
N_2O from manure			2.0%	IPCC (2007
NH_3 from fertilizer	3.2% ± 2.4	Krauter and Blake (2009)		
N_2O from fertilizer	1.4%	This assessment	1.0%	IPCC (2007)
N_2O from leguminous crops			1.0%	IPCC (2007)
NO_3^- leaching from alfalfa	8.2%	Letey et al. (1979), Putnam et al. (2005)		
NO_3^- leaching from croplands	34%	This assessment	30%	IPCC (2007)

[1]Includes emissions from animal production unit and field operations

Table B7.2. Current and improved partial nutrient balance (PNB) for major California crops. PNB is the ratio of N in crop material exported from field to the amount of N fertilizer applied.

[Navigate back to text]

Crop	Current PNB (%)[#]	Improved PNB (%)[#]			Sources for improved PNB
		Low	High	Median	
Cotton	61	40	93	66.5	Fritschi et al., (2005)
Potato	55	27	91	59	Meyer and Marcum, (1998)
Rice	75	42	117	79.5	Linquist et al., (2007)
Wheat	56	68	104	86	Ehdaie and Waines (2001)
Almond	49	70	90	80	Brown et al., (unpublished data)
Avocado	19	31	45	38	Lovatt et al. (2001)
Grapes, raisin	45	54	70	62	Peacock et al. (1991)
Grapes, wine	56	36	93	64.5	Smart, (pers. comm.), Christensen et al. (1994)
Lemons	51	52	-	52	Embleton et al. (1981)
Nectarines	22	32	103	67.5	Weinbaum et al. (1992)
Oranges	39	44	-	44	Embleton et al. (1974), Ali and Lovatt (1994)
Peaches, freestone	25	17	63	40	Saenz et al. (1997), Johnson et al. (2001)
Pistachio	56	72	-	72	Rosenstock et al., (2010)
Plums, dried	54	14	54	34	Southwick et al., (1996)
Walnut	52	41	151	96	Richardson and Meyer, (1990)
Broccoli	46	35	42	38.5	LeStrange et al. (1995,1996)
Carrots	27	62	75	68.5	Allaire-Leung et al. (2001)
Celery	36	41	71	56	Hartz et al. (2000)
Lettuce	34	51	-	51	Hartz et al. (2000)
Peppers, bell	18	24	36	30	Hartz et al., (1993)
Strawberry	34	54	55	54.5	Bendixen et al. (1998)
Tomatoes, fresh market	61	61	84	72.5	Hartz et al. (1994)
Tomatoes, processing	64	89	108	98.5	Hartz and Bottoms (2009)
Corn[1]	61	-	-	-	No data available for improved PNB

[1] At the time of writing, there is a prominent lack of available studies showing improved PNB for corn and cereal forages under changing water and nutrient applications.

Table B7.3. Estimated reductions in NOx and PM2.5 from the implementation of proposed 2007 measures by CARB (Tonnes year^{-1}). Estimates for the South Coast and San Joaquin Valley are for 2014 and for the Sacramento Valley are for 2018. [Navigate back to text]

	NO_x				**Direct $PM_{2.5}$**		
Source activity	**South Coast**	**San Joaquin Valley**	**Sacramento Valley**	**Total**	**South Coast**	**San Joaquin Valley**	**Total**
Passenger vehicles[1]	662	0	0	662	-	0	0
Heavy-Duty Trucks[2]	19,766	21,720	3,145	44,631	-	1,424	1,424
Goods Movement Sources[3]	9,635	0	99	9,734	861	-	861
Off-Road Equipment[4]	3,476	1,225	629	5,331	861	265	1,126
Total Projected Emission Reductions	33,540	22,945	3,874	60,358	2,880	1,689	**4,569**

[1]Smog check improvements

[2]Cleaner in-use heavy duty truck

[3]Ship auxillary engine cold ironing & clean technology; Cleaner main ship engines and fuel; Clean up existing harbor craft

[4]Cleaner in-use off-road equipment (>25 hp)

Table B7.4. Estimated reductions in N from improved OWTS management (Gg N). Estimates were done across a range of removal efficiencies and retrofit scenarios to account for variation in system management and the fact that not all OWTS present an environmental risk and need to be replaced. By comparison, raising treatment at WWTP 10% reduces Nr by 15.9 Gg yr^{-1}. [Navigate back to text]

	Retrofit (% of existing systems)		
Removal efficiency (%)	**20**	**40**	**80**
40	1.3	2.6	5.2
60	2.0	4.0	8.0
80	2.7	5.4	10.9