CHAPTER SEVEN

Responses: Technologies and Practices

Appendix 7.1 Technical options to control the nitrogen cascade in California agriculture

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7.1.0 Technical Options to Control the Nitrogen Cascade In California Agriculture

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; 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 7.1.1). 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 7.1.1.1.

Box 7.1.1 Why A Qualitative, Not Quantitative, Assessment

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. First, 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 an empirical one. 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. Therefore estimates of the impact of a single change are meaningless, in practice.

[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 N management.

Table 7.1.1.1. Resources Describing Technical Options to Control the Nitrogen Cycle from Agricultural and Non-Agricultural Sources

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

7.1.1 The Nitrogen Cycle

Understanding the potential efficacy of different management interventions in regulating the nitrogen (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 opportunities to control the N cascade[2]. Descriptions of the forms of N and the major processes of the N cycle can be found in Table 7.1.1.2.

Table 7.1.1.2 Major Nitrogen Cycling Processes

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 microorganisms 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	Oxygen, temperature, water-filled pore space,

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

	environment	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

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 (EPA SAB, 2011). Examples include constructing wetlands to intercept NO_3^- in runoff (removal), using nitrification inhibitors to retard conversion of NH_4 to NO_3^- (transformation), and improving distribution uniformity to increase the efficiency of irrigation and avoid saturating some parts of the field and thereby reducing oxygen availability (improved efficiency). Applicability of each strategy is subject to the constraints of the production environment (Table 7.1.1.3). 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 optimization of several factors including, but not limited to: availability of technology, cost, effectiveness, relevance to crop or animal species of interest, soil, irrigation system, regulations, climate, labor, and the market.

Table 7.1.1.3. Strategies to Control the Release of N Into the Environment. Source: Adapted from EPA SAB 2011.

Control Strategy	Advantages	Limitations	Current Examples
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/denitrification 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 by-product 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

7.1.2 Inorganic Nitrogen Management

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

7.1.2.1 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 that are mediated by microbes accelerate.

[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.

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 uncertainties inherent to farming—due to technology, information, economics, and weather limitations—often induce crop producers to supply more N than a crop assimilates to ensure adequate nutrition and high yield. Because of the surplus N use that often results, reducing the rate of application is an often-cited option to control emissions without compromising yield.

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 (Hoben et al., 2011; Millar et al., 2010; van Groenigen et al., 2010; Venterea 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 Groenigen 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, and 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 under ideal growing conditions that induce rapid crop growth and development, or unavailability of N during critical phenological periods. Part of the reason growers apply N at higher than needed 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 that 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 that 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 a N management program that involves diagnostic testing to guide split N applications was shown 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., 1994a). Although the 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 the 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 amount of mineral N in soil and availability to plants. 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 (Breschini and Hartz, 2002; Hartz et al., 1994b). 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]

[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.

established for cultivars and conditions unrepresentative of agriculture today limits the applicability of tissue tests in many situations. Furthermore, the ability to apply split applications and deliver precise fertilizer rates varies by cropping system and management, and in fertigation systems is affected 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 of applications and better timed/even delivery of nutrients.

7.1.2.2. 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 a N source that promotes or suppresses specific N cycle attributes is thus theoretically possible. Options available to change inorganic N sources include: (1) switching between conventional materials (e.g., from urea or anhydrous ammonia to ammonium sulfate, or (2) switching from conventional synthetic materials to "enhanced efficiency materials." [6]

Changing between conventional materials can be an effective strategy to reduce NH_3 volatilization losses, and N_2O emissions. 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 rates 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).

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[6] Enhanced efficiency fertilizers (EEF) are synthetically derived materials that are engineered to moderate the rate at which N becomes available to plants and microbes, extending it over a longer period of time (Shaviv and Mikkelsen, 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 fertilizer type can also have an effect on N_2O flux. After reviewing more than 1000 studies worldwide of N_2O production, Stehfest and Bouwman (2006) conclude that rates of N_2O evolution from anhydrous ammonia are significantly higher than from other fertilizer types, even when accounting for differences in the experimental conditions across studies, such as tillage systems, fertilizer placement, soil C, and pH (Snyder et al., 2009). Burger and Venterea (2011) reviewed only side-by-side comparison trials from North America (none included California conditions), and likewise found that in five out of six studies, anhydrous ammonia was the most likely to generate significantly higher N_2O emissions, compared to urea-based fertilizers. In addition, two recent California studies in corn and wheat found that use of ammonia fertilizers (in anhydrous or aqua forms) generated from 40 to 60% more N2O emissions compared to urea-based or sulfate fertilizers (Burger and Waterhouse, 2016; Zhu-Barker et al., 2015).

Switching to enhanced efficiency fertilizers (EEF) from conventional synthetic fertilizers is often widely considered a valuable technological option to address the N challenge (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 that polymer coated fertilizers 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. Some evidence suggests 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) or similar total annual losses (Parkin and Hatfield, 2010) when compared to fertilizer application without nitrification inhibitors.

Nitrate leaching potential may also be reduced with the use of EEFs. With its negative ionic charge and water solubility, NO_3^- does not adhere to similarly charged clay particles and therefore are not readily retained in the soil matrix. Nitrate readily leaches below the root zone with water, especially when the soil profile is near saturation, which can occur with uneven distribution of irrigation water or with precipitation (Hanson et al., 2005; Letey, 1994). While utilizing NH_4-based fertilizer instead of NO_3-based fertilizer may help to retain N in the soil root zone a little bit longer, providing greater opportunity for crop uptake, NH_4^+ is usually quickly nitrified in agricultural systems, often within one to three weeks (Robertson, 1997). However, research, some of it done in California, has shown that EEFs slow 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 the root zone when sulfur coated urea was used compared 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], especially in the near term. To begin with, EEF are more expensive, with prices estimated to range from 9% (Snyder et al., 2009) to nearly double (California Nitrogen Assessment (CNA), stakeholder meetings) the prices of conventional synthetic fertilizers. This additional cost is often unwelcome to growers 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., 1985, 1979). Under current farming conditions, however, it is not clear if EEF will produce comparable benefits in California as in other regions where they are being promoted. 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.

7.1.2.3 Modify Fertilizer Placement and Timing

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 root zone.

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). 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, such as nitrification and denitrification, that prime it to

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

be lost from the root zone through leaching or atmospheric emissions. The capacity to achieve this synchronicity requires (1) knowledge of crop growth patterns and timing of nitrogen uptake, (2) ability to predict crop growth responses to changes in weather, and (3) 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, and is becoming increasingly available for tree crops (Saa et al., 2014; Muhammad et al., 2015). The second requirement is more difficult to meet for crops grown during the rainy season because of California's highly variable weather from one year to the next, which can cause yields to vary by 50%. For irrigated crops, though, variability of precipitation does not play a large role. For example, fertigated systems are well suited to match N supply with crop N demand.

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 improve uptake and N use efficiency. For example, 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 avoid applying N in the fall are available from research throughout the Midwestern United States (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 fruits 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., 1994a). 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). It is important to note that improved application timing does not always result in increased productivity. Hutmacher et al. (2004) demonstrated that yields of Acala cotton grown across six farm sites in the San Joaquin Valley were statistically similar regardless of whether one or two applications were used. Resources required for additional application would thus have little value.

Fertilizer 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, with the authors hypothesizing that surface-applied N was more susceptible to nitrification-denitrification losses, compared to sub-surface applied NH_3.

Placement can also affect N_2O emissions. Several laboratory and field studies from locations outside of California have shown that concentrating fertilizer N, for example by applying it in bands, tends to produce greater N_2O emissions than dispersing the fertilizer N, for example by broadcasting or disking (Engel et al., 2010; Tenuta and Beauchamp, 2000). These findings were confirmed by a Central Valley field study in corn production, that showed applying urea ammonium nitrate as two bands reduced cumulative N_2O emissions by almost 70%, and applying one band to the shoulder of the planting bed reduced emissions by almost 60%, compared to applying one band directly on the bed (Burger and Waterhouse, 2016). One study (Hultgreen and Leduc, 2003 cited in Snyder et al., 2009) shows lower N_2O emissions from band placement versus broadcast surface applied urea. 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. This is especially true for highly concentrated alkaline-forming N fertilizer materials, such as anhydrous ammonia, that have been shown to result in a build-up of nitrite, which then becomes the substrate for N_2O production (Maharjan and Venterea, 2013).

The potential for improved placement and timing of fertilizer N to significantly alter the current N fluxes from croplands on a statewide basis, however, ultimately depends on the extent of adoption of practices that result in greater N uptake efficiency, reduced NO_3^- leaching, and lower N_2O production potential. Some evidence suggests that some of these practices are already commonplace. For example, 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% (down to approximately 30% of respondents) and the number of growers that applied three or more applications rose 5.7%. Current use of these practices is largely not quantified. However, research has repeatedly demonstrated yield benefits from these practices and this aspect underlies most recommendations (Hartz et al., 1994b; Breschini and Hartz, 2002; Rosecrance et al., 1998; Lovatt, 2001).

There are clearly specific production systems where more attention to better timing and placement may be warranted. Rice may be one case where better placement would increase N recovery (see discussion above) and strawberry may be another case 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.

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 with and adoption of precision agricultural technologies (Roel et al., 2000). Evidence of its application and effectiveness in the field is lacking. For many California cropping systems, technology is either unavailable (e. g., for horticultural systems) or not well adapted (e.g., not able to deliver nutrients at a meaningful scale of spatial variation). An effort is underway to adapt precision agriculture to tree crops; and harvesters and irrigation systems are under development (Rosa et al., 2011). Potential future fertilizer N efficiency gains from precision agriculture, beyond simple diagnostic soil and tissue tests, remain uncertain.

7.1.3 Water Management

Water regulates biological activity, chemical conversion of nitrogen (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 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 of which depends 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.

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

Managing soil moisture content in California is unique by comparison to most other agricultural regions of the United States 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 reduce soil aeration, increase microbial activity, and accelerate 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 7.1.3.2). 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 root zone longer, increasing plant N uptake potential and reducing leaching losses (Feigin et al., 1982a, 1982b). The positive outcomes are mostly a consequence of the fact that water is the dominant factor dictating NO_3^- movement laterally and vertically through the soil profile in the 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 root zone, 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 (Feigin et al., 1982a, 1982b; Meyer and Marcum, 1998; 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 that relate irrigation management with control of gaseous emissions are limited. 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. 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 (Krauter and Blake, 2009).

7.1.3.1 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 which 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

[9] Two interrelated metrics are used to describe irrigation system performance: uniformity and 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 which exceed demand.

generally improved to capitalize on the advantage pressurized systems present in the 16 years since these data were collected, 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 by 10 -15% and to decrease subsurface drainage by 50% (Hanson, 1989). Such gains result from the shorter water advance times which reduce 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 to surge irrigate (Hanson and Fulton, 1994) or to use torpedoes to compact soil and allow water to move more quickly down the furrow; the effectiveness of these practices depends 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, 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. Change in these systems may require policies specifically designed to address these challenges.

7.1.3.2 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). N 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. N-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.

7.1.4 Alternative Soil Management

Soil management, in the broadest sense, encompasses virtually every cropping decision a grower makes, from tillage to nitrogen (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.

7.1.4.1 Conservation Tillage

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[10] Drainage refers to the movement and removal of subsurface water from the crop root zone. 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.

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 that the intensity of tillage be reduced 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 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

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[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 can also 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 C 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).

al., 2007; B. A. 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 measurements of 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, 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; B. 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 precise 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 crops, 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.

7.1.4.2 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 (L. 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.

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

[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, 2009). Similar concerns are applicable to biosolids.

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, a 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 commercial settings (Hartz and Johnstone, 2006).

Will using only organic N compromise productivity? This issue is very much debated (see Appendix 7.3). Some studies show yields are lower in organic than conventional systems (e.g., (Reganold et al., 2010; Jackson et al., 2004) when equivalent amounts of N are applied, presumably because much of the N contained within organic sources is not immediately plant available (Rosen and Allan, 2007). Others suggest yield differentials are rarely apparent (Badgley et al., 2007; Drinkwater et al., 1998; Reganold et al., 2001), or in some circumstances organic systems have even been shown to exceed the average yield of corresponding conventional systems in the same region (Bowles et al., 2015). 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 from 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). Taken all together, it is generally accepted that present-day production systems using only organic N sources are less productive than those using inorganic sources, but notable exceptions exist in some crops and management systems.

But is using organic N amendments 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 that the use of organic waste decreases leaching by nearly 50%.

One report demonstrates that by stimulating the active denitrifier community, N_2 emissions increased 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 in organic systems (Burger et al., 2005; Kong et al., 2009). In addition, Bowles et al. (2015) demonstrate that several organic processing tomato farms in Yolo County, California are able to achieve tight plant-soil N cycling, resulting in very low soil NO_3^- pools and low potential of N loss, even while achieving crop yields equal to or exceeding the overall county-wide average. The authors attributed this phenomenon to a combination of efficient N management, high soil microbial activity, and rapid plant N uptake.

On the other hand, 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 Bergström, 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 to, if not greater than, those from inorganic sources, 1-2% of N applied (Bouwman et al., 2002a, 2002b; 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 emissions can result (Cavero et al., 1999; Poudel et al., 2002, 2001).

7.1.4.3 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), little data are available to evaluate its ability to achieve the proposed benefits and even less to evaluate the mechanism by which it may do so.

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.

7.1.5 Landscape Approaches

Not every action to control nitrogen (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 a 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.

7.1.5.1 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 30 m. 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 is taken up in unmanaged systems (Bedard-Haughn et al., 2005, 2004). Riparian areas at the edge of waterbodies reduce NO_3^- to similar degrees. Data from 89 studies on 45 riparian areas indicate 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 a 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 on 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 (although, conversely, they may also provide habitat and food sources for pollinators and other beneficial organisms). Thus, vegetative buffers may present tradeoffs with economics, labor, and agricultural chemical use.

7.1.5.2 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 expected (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 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 United States, principally distributed at commercial drinking and treatment facilities (Jensen et al., 2012). Bioreactors are an effective technology reducing loading at a smaller scale. Robertson and Cherry (1995) show

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

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

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 does not need to be nitrified prior to denitrification, as is the case in industrial wastewater treatment. Effluent from field drains at a local scale or aggregated at larger scales may prove to be an option worth exploring.

7.1.6 Agrobiodiversity

Biodiversity, and agrobiodiversity[16] more specifically, improves nitrogen (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; Smukler et al., 2010; Young-Mathews et al., 2010). It achieves all this through virtually every plausible N control mechanism, from efficiency to transformation. 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.

7.1.6.1 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) (L. E. Drinkwater et al., 1998; Hu et al.,

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[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.

1997; Carol 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 one 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 (Jackson, 2000; Patrick et al., 2004). 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 crops 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 at levels sufficient to meet the 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 synchrony with crop demand. 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 in systems using 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). Despite the potential drawbacks, 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). These results suggest the potential to partially substitute organic N sourced from cover crops for inorganic N.

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. Most frequently cited challenges include short time frames between cash crops limiting total cover crop biomass production, depletion of soil water reserves by the cover crop, and costs of establishment and incorporation (Jackson et al., 2003). In addition, cover crops in orchards and vineyards can change the microclimate, which may lead to frost damage to perennial crops (Ingels et al., 1994).

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. Ultimately, successful use of cover crops requires evaluating the benefits and potential concerns of a cover crop species within the context of a specific farming system.

7.1.6.2 Diversify Crop Rotation

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, 1999; Bassil et al., 2002). Diversifying rotation to include safflower will only be beneficial if the entire rotation is accounted for. 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). Recent research has shown that an appropriate credit ranges from approximately 67 kgs N/ha to 145 kgs N/ha depending upon soil type, age and status of alfalfa stand, weed intrusion, degree of foliage plow-down, time of year, and time elapsing between plow-down and the subsequent crop planting (Putnam and Lin, 2016) .

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 are much less common. 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, as well as internal drivers such as relative costs of production. 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. On the other hand, low commodity prices for milk require that dairies produce low-cost forage crops to minimize feed expenses, a situation which may limit the diversity of forage crops they can choose from.

7.1.6.3 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. 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 mycorrhizae was up to 80% lower. Alfalfa producers directly enhance soil microorganisms 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, 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 be implemented in the field. Thus, it is unlikely to be a significant factor in helping California better manage N use or reduce saturation in the immediate future. However, active management of microorganisms is the foundation of N treatment in other sectors (e.g., wastewater treatment). A first step would be to identify the plausible opportunities that could work at the field scale.

7.1.7 Genetic Improvement

7.1.7.1 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' nitrogen (N) demand, ability to recover soil N, and how well it utilizes N once assimilated. Not until recently has N use efficiency become a subject of interest for plant breeders. Previously, 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, a plant's ability to explore the soil and take up 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 smaller 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.. 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, which demonstrates 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.

Nonetheless, the principle reason we believe that genetic manipulation can yield results for California soon is that 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.

7.1.7.2 Breed Animals for High Feed Conversion Efficiency

Feed conversion is the amount of feed required to produce one unit of product (i.e. 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, a current breed, provides evidence of the potential benefits (Havenstein et al., 2003a, 2003b; 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 85% of the improvement to better feed conversion. However, 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 United States 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.

7.1.8 Animal Nutrition and Feed Management

Protein nutrition influences productivity, profitability, and the efficiency of nitrogen (N) use in cattle and poultry production systems. Production of milk, meat, and eggs are correlated with crude protein intake (Bailey et al., 2008; Kebreab et al., 2001; 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 feedstuffs and feed management at seven dairies in California and found that dairies operated at between 16 and 27% N utilization efficiency. That means that for every 1,000 kg of N fed, the least efficient dairy excreted 840 kg of N, while the most efficient dairy excreted only 730 kg of N. 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. Even if manure handling practices remain 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 cow diets averaged 17% ± 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. This survey by Castillo et al. (2005) 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 7.1.1).

However, feed management rarely accounts for the differential requirements of animals during various points in their life cycle well. Calves, dry, and lactating cows demand a different amount of crude protein. If fed the same diets, only altering dry matter intake, overfeeding of N results in increased N excretion. Recognition of the variable needs of cattle has led to calls to

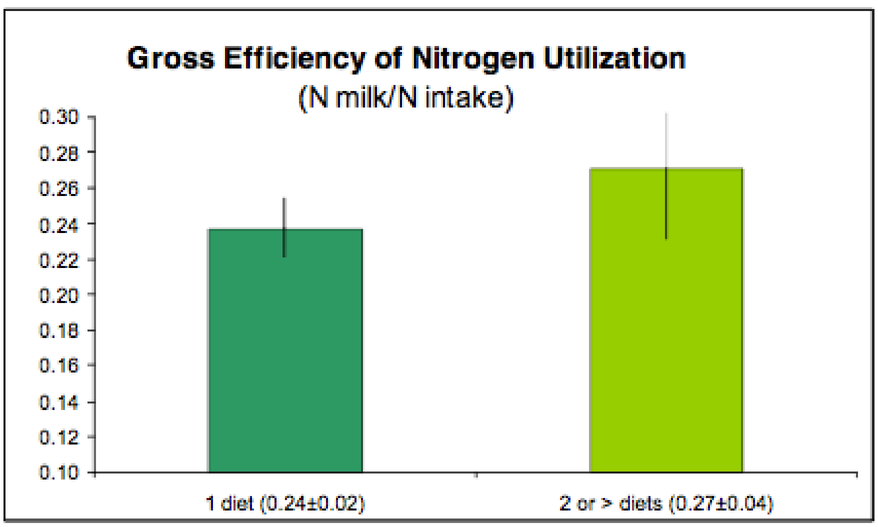

FIGURE 7.1.1 Nitrogen Utilization Efficiency of 51 Dairies in Modesto. Source: Castillo et al. 2005.

increase staged or precision feeding (Meyer and Robinson, 2007). Most animal operations formulate diets to provide minimum required nutrient concentrations at the lowest cost. Because protein is among the most expensive ingredients, their use is generally tightly monitored. Despite close attention, N is sometimes fed in quantities larger than is 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 by-product 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. Given the relatively low cost of distiller's grains, however, reformulating diets by substituting other materials could potentially raise the overall costs of feed.

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 in NUE 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 the role of animals in the broader agricultural system of the state and how crop diversity affects 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 by-products, 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 a cheap protein (N) source, distiller's grains. Use of this feedstuff complicates diet formulation due to the near double N content compared to unprocessed grains, increasing excretion and emissions (Hao et al., 2009, Chapter 7).

7.1.9 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 nitrogen (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.

7.1.9.1 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 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., 2009). 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).

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[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.

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 illustrate 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.

7.1.9.2 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, where and how they might be effective for California producers.

7.1.9.3 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 suggest that 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) coagulate fine particulates, 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 additional 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.

7.1.9.4 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 is 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 the controlled 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 facilities 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.

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