Chapter 7: Responses: Technologies and practices

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What is this chapter about?

Management practices, and their underlying technologies, together with land use decisions, have a dramatic influence on the total amount and ultimate fate of nitrogen (N) in the environment. Based on the California nitrogen mass balance, nine critical areas for intervention in the nitrogen cascade were identified. This chapter reviews these critical control points and evaluates related mitigative strategies and technological options to reduce emissions of nitrogen. This chapter also evaluates the potential for synergies and tradeoffs that may occur from adopting these strategies, as well as the support of current and impending policies for their implementation.

Stakeholder questions

The California Nitrogen Assessment engaged with industry groups, policy makers, non-profit organizations, farmers, farm advisors, scientists, and government agencies. This outreach generated more than 100 nitrogen-related questions which were then synthesized into five overarching research areas to guide the assessment (Figure 1.4). Stakeholder generated questions addressed in this chapter include:

- **From a systems perspective, where are the control points for better management of N?**
- **Are there tradeoffs between reduced N application and other cropping considerations? Will deviating from current N applications affect product quality, increase pest pressure, etc?**
- **Are there current management practices that would increase N use efficiency and reduce N pollution?**

Main messages

Today, countless technologies and practices are available to optimize reactive nitrogen (N) use and change the way Californians interact with the nitrogen cascade. Knowledge and tools to limit the introduction of new reactive N into the cascade; mitigate the exchange of N among the bio-, hydro-, and atmo-spheres; and adapt to the increasingly N -rich environment are already widely available for agriculture, transportation, industry, water treatment, and waste processing. With current technology, we estimate that strategic actions could reduce the amount of reactive N in the environment significantly.

Limiting the introduction of new reactive N—through improving agricultural, industrial, and transportation N efficiency—is the most certain way to create win-win outcomes. Increasing efficiency would decrease the amount of N per unit activity (potentially decreasing costs) and decrease emissions. Fortunately, practices are available to increase fertilizer and feed N use efficiency for virtually every agricultural commodity. Our conservative estimate suggests gains in efficiency could result in an estimated 36 Gg less fertilizer N use yr^{-1} and 82 Gg less feed N demand yr^{-1} without compromising productivity. By comparison to agricultural practices, the efficacy of engineering solutions to increase efficiency is well established.

Because a single source category is generally responsible for the majority (>50%) of each N transfer among environmental systems, priorities to mitigate N emissions are clear. These include: manure management (to reduce ammonia (NH_3) to air), soil management (to reduce nitrate (NO_3^-) to groundwater), fertilizer management (to reduce nitrous oxide (N_2O) to air), fuel combustion (to reduce nitrogen oxide (NO_x) to air), and wastewater treatment (to reduce ammonium (NH_4) to surface water).

Though these activities are the most culpable, a diverse number of additional actions also contribute to these transfers and it will take a systemic perspective to reign in N emissions. Further, because reactive N is intrinsically mobile in the environment, a narrow focus on a specific mitigative action will have the tendency to cause secondary emissions, thereby simply transferring the burden oftentimes with more harmful environmental and human health outcomes.

Reactive N is already changing California's air, water, soils, and climate, and dynamics of the N cascade dictate that further degradation will continue to occur for some time. Moving forward, Californians will have to adapt systems and behavior to the new state of resources to maintain productivity, minimize exposure, and relieve further pressure on the environment. Adaptation will be especially important as populations grow further and concentrations of reactive N in the environment increase. The extent of personal disruption will vary depending on the issue, with fixes like applying insect repellant more often to reduce risk of contracting West Nile Virus being simple and low-cost while such fixes as spending more time indoors on high ozone (O_3) days potentially more costly. There is already a need to treat drinking water to the regulated level of safe (45 mg per L as NO_3^- or 10 mg/L NO_3—N) in many parts of the state, with this need only projected to increase in the future. Agriculture is one industry that must be proactive in its planning. Ozone, groundwater NO_3^-, and increased deposition may all fuel changes in productivity and management. Knowledge of how California's environmental systems will inevitably change and planning for such changes will help future adaptation.

A comprehensive and integrated network of monitoring sites is required to understand and address the multi-media impacts of reactive N **in the environment.** California, by comparison to many other regions of the US, is ahead in having this capacity. Monitoring sites and programs operated by state and federal agencies including the California Air Resource Board, State Water Quality Control Board, and the

Environmental Protection Agency provide an increasing clear picture of N impacts (e.g., O_3, NO_3^-). However, incoherence and inaccessibility of data prevent improved and continuous assessment. A statewide effort is needed to integrate the diverse air, water, climate, and source activity data collections. Comprehensive integration, transparent protocols, and honest evaluation of uncertainty are key characteristics of such an integrated platform.

7.0 Introduction: Critical control points of California's nitrogen cascade

Californian activities mobilize more than one Tg of N each year (see Chapter 4). In the environment, it is transformed through physical, biological, and chemical processes enabling it to move back and forth repeatedly among the hydro-, bio-, and atmo-spheres, where it affects human health and the environment, in both positive and negative ways. That continuous multi-media cycling is referred to as the "Nitrogen Cascade" (Galloway et al. 2003). At certain points in the N cascade, human actions or environmental conditions can modify N transformations or transfers between environmental systems, accentuating or attenuating its impacts. Because of their strategic importance in regulating the N cascade, these points are collectively referred to as "critical control points" (Table 7.1). Critical control points are activities, not specific technologies. Selection of the appropriate technology to accomplish the activities will be subject to constraints on prices, land, labor, and the N intensity of the activity.

[Table 7.1]

Critical control points of the N cascade have been identified at national (US), continental (Europe), and global scales (Galloway et al. 2008; Oenema et al. 2011; INC 2011). These assessments indicate that a few key actions targeted at the critical control points could significantly alter the relationship humans have with the N cycle, for the better. Estimates suggest that increasing fertilizer N use efficiency; treating wastewater; reducing emissions from fuel combustion; and improving manure management would reduce the amount of reactive N released into the environment by 25% to 30%, assuming reasonable and achievable targets (Galloway et al. 2008; INC 2011). The conclusions beget the question: Is technology sufficient to achieve similar or even greater control of California's N cascade, without compromising benefits of N in California?

Based on California's N mass balance, we identified nine critical control points to manage its N cascade (Figure 7.1). Four of these act on the demand for new reactive N and therefore alter multiple emissions pathways simultaneously. Three of these four control points affect the total amount of N required for food production through changes in agricultural N use efficiency, consumer food choices, and amount of food wasted. The fourth control point acts to reduce fossil fuel burning by improving transportation and energy sector efficiency. The remaining five control points target specific transfers of N between environmental systems, including NH_3 volatilization from manure, NO_3^- leaching from croplands, greenhouse gas (GHG) emissions from fertilizer use, NO_x emissions from fuel combustion, and wastewater management. In addition, we present adaptive responses to the nearly inevitable future N -rich environment, including treating for NO_3^- in groundwater used for drinking, and designing N-smart agricultural systems. When reasonable, we provide a first approximation of the mitigative potential attainable with implementation. Additionally, we discuss the potential for synergies and tradeoffs that may occur from adopting these strategic actions as well as the support current and impending policies—both N-focused and beyond—have for implementation. The chapter concludes by arguing that adoption of an integrated practice and policy response is the only reasonable approach forward[1]. Whereas full integration of N management would need to account for countless concerns (e.g., stakeholder groups, scales, source categories), development of agreements and institutions to cross boundaries among N

[1] Two appendices support chapter 7. Appendix 7A reviews specific agricultural practices and technologies that alter N cycling on farms and ranches. Appendix 7B outlines the calculations that support the estimated decreases in N emissions.

sources, species, and impacts could initially provide support, signals, and incentives to align California on a more sustainable N trajectory.

[Figure 7.1]

7.1 Limit the introduction of new reactive nitrogen

The most certain way to reduce the introduction of reactive N into the environment is to limit its production, use, and release. Food production, fuel combustion, and feed importation represent the three primary sources of new N inputs into California's N cascade (see Chapter 4). Due to their significant leverage, there is an immediate need and an opportunity to moderate the N cascade through proactive management of these activities.

7.1.1 Agricultural nitrogen use efficiency

Inefficient agricultural N use increases total N demand, because less of the N applied achieves its intended purpose of producing a harvestable product. Unassimilated N represents a waste of resources used to fix atmospheric N and causes indirect emissions beyond rootzone and field boundaries, with the threat increasing exponentially with excessive use (van Groenigen et al. 2010; Broadbent and Rauschkolb 1977). Because of inherent, and to a certain extent unavoidable systemic technical inefficiencies[2], producers must use fertilizer and feed N in excess of plant and animal demand, respectively, if they want to ensure adequate nutrition, although clearly some systems and some individual operators are more efficient than others (Chapter 3; Breschini and Hartz 2002; Lopus et al.

[2] It is necessary to differentiate between technical and economic efficiency. Technical efficiency refers to the capacity of the system to utilize the resource. Economic efficiency refers to the point when the marginal costs become greater than the marginal returns. The two are rarely equal, especially in agricultural systems.

2010). More judicious use of N would enable producers to cut down on the excesses while maintaining productivity and benefiting farmers' bottom line[3] and the environment both (Hartz et al. 1994).

There are many practices and technologies to manage N in agriculture, oftentimes with research verifying their effectiveness, even when only considering the California-relevant production conditions (Table 7.2). Advanced irrigation systems, crop growth and development models, reduced tillage systems, enhanced efficiency fertilizers, precision feeding, staged feeding, hormones, breeding, and animal husbandry are only a few of the available approaches that have been tested. Today, producers can select from a diverse menu of options to fine-tune N use in their systems (Nahm 2002; Hristov et al. 2011; Ndegwa et al. 2008; Box 7.1). Production decisions, however, are subject to multiple constraints – land, water, economic costs and returns, regulations, technology, etc. Persistent low efficacy of N use reflects the multidimensionality of farming and the historic relatively low importance of careful management of N. Until recently, fertilizer and feed N was relatively cheap production insurance and little attention was paid to the environmental externalities and social costs resulting from N pollution (VandeHaar and St-Pierre 2006; Meyer 2000). At present, control of N pollution is a major driver of production decisions for only a few systems in California (e.g., dairy). For N use efficiency to increase, consideration of N emissions will have to be integrated into operational decision making more often. As stated, technologies are already available to support such improved N practice; however, refinement and innovation will still be needed to adapt systems to the constantly changing policy and production environment.

[Table 7.2] [Box 7.1]

[3] However, fertilizer N costs are but a small portion of total operating costs (<5%) for many crops. In such cases, higher profits derived from lower input costs may be counterbalanced if N becomes yield limiting during the years of optimal production or if implementation costs of improved practices add to operating expense (Medellin-Azuara et al. 2011; Jackson et al. 2003; Hutmacher et al. 2004). Because of the many interacting factors that determine yield, revenue, and profit it is difficult to conclude a priori that increasing technical N use efficiency would yield economic benefits for the farmer. Indeed there are many plausible scenarios when it would not. Many farmers continue to operate at the economic efficient levels, which often mean N use rates are higher than they would be at the technically most efficient levels.

For nearly all cropping systems, N use efficiency is consistently higher in plot and field-scale research trials than the documented statewide average, frequently considerably so (See Tables 3.1 and Table B7.2). These data suggest that it is possible to increase agronomic N use efficiency significantly.[4] Assuming yields do not change, raising N use efficiency even half[5] as much as this amount could decrease inorganic N fertilizer demand (and application) by 36 Gg N year^{-1}. As a result, it is reasonable to expect at least proportional reductions in emissions (8 percentage points). Because emissions increase exponentially after N application rates exceed crop uptake, this may even be a conservative estimate (sections 7.2, 7.3, Appendix 7B). If this were to occur, it seems that only a small fraction of this reduction would be translated into reduced gas emissions or runoff losses because of the relatively small proportion of N applied lost directly through these pathways, thus much of the reduction would likely be translated into reduced NO_3^- loading to groundwater. The fact that recorded statewide average N use efficiencies are almost universally less, across crops, than efficiencies achieved in research trials, suggests that neither technology nor scientific information are primary impediments to N efficient California croplands[6]. Future efforts to increase N use efficiency will have to extend beyond the development of new technological innovations to include socio-economic drivers of technology adoption and use (e.g., Jackson et al. 2003).

[4] Nitrogen use efficiency, here, is assessed as a partial nutrient balance (PNB), which is the ratio of N in crop material exported from the field to the amount of applied N (IPNI 2014). Calculations based on Tables 3.1 and B7.2 suggest a potential increase in NUE of 16-percentage points, based on an area-weighted average for 33 crops. Potential increases vary significantly among crops, with some being far less.

[5] NUE in research trials is always greater than that obtained in the field production, sometimes considerably so, because of technical inefficiencies. To account for this, we have suggested that technical potential of increasing NUE are half of the calculated differences. This is likely a conservative estimate but represents a starting point for discussions.

[6] Results must be interpreted with caution. Estimating NUE by partial nutrient balance (PNB) is unable to distinguish between soil and fertilizer N in the plant. Indigenous soil N contributes variable quantities of N depending on the fertility of the soil potentially confounding the comparison. Research sites may perform better due to underlying soil fertility. Regardless, in virtually every crop examined, statewide average partial nutrient balances were lower than recent research using feasible production practices, sometimes by quite significant amounts, irrespective of crop type.

It appears feed N utilization efficiency in California animal production systems can also be improved, at least incrementally. Because data are sparse, we conservatively conclude that the increase could be at least four percentage points. Even such modest increase would have significant consequences for feed N demand and management of manure N. Assuming that product yield and N concentration remain constant, feed demand would decrease to 85% of current levels (equivalent to an 82 Gg N decrease). At the same time, emissions reductions from avoided fertilizer use and biological N fixation in feed production and the manure N burden would be reduced proportionately.

Increasing agricultural N use efficiency has the potential to create win-win outcomes for the producer and the environment. More shrewd N management may add to labor and material costs for producers, however. Some studies suggest that incremental improvement may be achieved with little added investment (Medellin-Azuara et al. 2013; Schaap et al. 2008). And it is likely that the total investment would be less than the potential resource degradation and health costs caused by N overuse. Therefore, agricultural N use efficiency appears to be the cornerstone of any strategy to slow the N cascade.

7.1.2 Consumer food choices

US and even global food consumption habits dictate the type, quantity, and methods of agricultural production in California. Via the market, consumers send signals that shape farmers' decisions on both what and how to produce. Because foodstuffs differ in their N content and in the amount of N required to produce them, consumer preferences for specific commodities can have a large influence on local, statewide, national, and global N cycling. Increases in consumption of more N intensive foods result in higher fertilizer demand, while decreases in consumption of such foods can decrease the overall need for fertilizers, thereby decreasing the amount of new N entering California agriculture.

Animal products are the least efficient foods in terms of amount of N required to produce each unit of final food N (or protein) consumed, due to the basic biological inefficiencies that occur when animals that have consumed plants are in turn consumed by humans. These inefficiencies are due to the fact that the majority of the N used to produce feed crops – estimates indicate that it can be over 90% (Galloway and Cowling 2002) - is lost to physiological maintenance, manure, and other avenues in the animals that consume those crops, with only a small amount making it all the way to the consumer's plate (see Box 5.1 for more detailed estimates on the percent of feed N that is eventually consumed as meat products). For this reason, consumer demand for animal products, in particular animal protein, is one of the most important factors affecting the introduction of new N into the cascade. Three distinct sets of consumer choices with regards to animal products would yield considerable benefits in terms of reducing inputs of new N. First, consumers could limit their choices to those animal products that are physiologically more N efficient (e.g., require less N per unit of final food product produced), such as poultry (Pelletier 2008). Second, consumers could choose foods from livestock that are raised using lower inputs of new synthetic N, such as livestock finished on unfertilized rangeland rather than in confined facilities requiring fertilized feed crops. The drawbacks of this option might include limitation in available rangeland (likely only an issue in the case of very widespread consumer adoption of this option), higher production costs leading to higher food prices, and potentially higher greenhouse gas emissions, especially methane, from range-fed cattle compared to feedlot cattle. This last drawback is speculative, however, when examined on a whole systems basis, with different studies showing very different results. When compared with beef cattle raised on highly managed pastures, those finished in feedlots resulted in lower system-wide emissions (Pelletier et al. 2010), while some studies of dairy systems (Rotz et al. 2009; O'Brien et al. 2012) found that the pasture-based systems resulted in lower overall GHG emissions per unit of fat- and energy-corrected milk. On the other hand, Arsenault et al.

(2009) found no major differences in emissions between pasture-based and confined dairy systems. To date, similar comparisons have not been examined for non-ruminant livestock, such as chicken.

The third consumer option is to lower animal protein intake to levels consistent with required daily intake. Average US consumers, and by likely extension Californians, consume more than double their recommended levels of annual protein intake, 63% of which comes from animal products (USDA 2010). Moreover, dietary patterns that include less processed and red meat, and more plant foods, are generally accepted in the medical literature as being associated with decreased risk of cancer, cardiovascular disease, and other diseases and mortality risk factors (Kushi et al. 2012), providing a health incentive for this choice.

Lowering animal protein consumption would not likely reduce N loss proportionally, however. Often diets low in animal protein contain greater proportions of fruits, vegetables and nuts; many of which require high N inputs and are grown in California. In contrast, slightly over one-third of the N fed to California livestock comes from feed crops not grown in California, but imported from other states (see Chapter 4). Thus decreasing animal protein intake may lead to tradeoffs, especially pertinent to the California agricultural landscape. (It should be noted, however, that reliance on imported feed does not really eliminate the N impacts, it only exports them out of California.) Nevertheless, because the quantity and quality (e.g., more proteins, fruits, and vegetables) of food demand scale with population growth and affluence (Dawson and Tiffin 1998), both of which are projected to increase measurably in the future, the importance of shaping diets towards low resource intensity foods for the future is clear (Hall et al. 2009). Because of the many dietary derivations that might occur if consumers changed preferences and the variation in N embodied in products, it is not currently possible to quantify the subsequent changes they would have on the N cascade.

7.1.3 Food waste

Addressing food losses may also play an important role in reducing the N loading in the cascade. Food losses represent a waste of fixed N since the fertilizer and feed N either is not consumed or is discarded into the environment. Nitrogen released from decomposing organic materials in the field or landfill contributes to air and water pollution and climate change. Reducing losses, therefore, shrinks resource demand and decreases pressure on the environment.

Food losses occur across every stage of the supply chain: from production through consumption. Food losses at retail outlets alone have been estimated to reach approximately 27% in the US (Kantor et al. 1997). Food losses for individual highly perishable products—such as ones produced in California—can be even higher. Dairy products and fresh fruits and vegetables accounted for half of retail losses in 1995 (Kantor et al. 1997). Consumer losses for whole and low-fat milk beverages is 45% and estimates for fresh fruits range from 8% (blueberries) to 54% (grapefruits) (Muth et al. 2011). Though the extent of food losses and waste in California has not been quantified, these findings clearly indicate that when farm, retail and home wastes are added together, a nontrivial fraction of agricultural products go uneaten[7].

Not all food loss is suitable for consumption, thus N wastage via this mechanism cannot be reduced to zero. However, clearly there are opportunities to recover food at most stages of the supply chain. Although data are unavailable to estimate exactly how much food goes unharvested, California crop producers often abandon significant fractions of production due to pests, costs, market, or weather constraints. Creation of incentives to harvest less desirable products would increase the quantity of food in the market and potentially have ramifications for N cycling. Recent interest in capturing on-farm food losses has catalyzed charitable "gleaning" crews across the state. Farmers who donate production that would have otherwise gone to waste often receive tax benefits. Gleaning results in greater export of N

[7] Food wastes accounted for 24% of total landfilled waste (by weight) in 2008.

off-site, reducing the soil pool of N and reducing the environmental N burden. But current levels of such harvest are miniscule by comparison to the total amount of loss.

Consumers' waste, rather than retail waste, dominates post-production food waste, comprising 96% by one estimate (Kantor et al. 1997). The consequence is loading of landfills with food waste. In California, food waste accounts for 24% of landfilled materials, despite extensive composting and recycling efforts (Cotton 2010; Brown et al. 2009). A diversity of issues contributes to high consumer food waste, including over-preparation, cooking losses, spoiled leftovers, and faulty packaging. Two mechanisms of behavioral change would have a positive effect. First, reducing the amount of food that enters the waste stream could be achieved through education on storage times, improved packaging, and shifting dietary preferences towards smaller portions. Second, education on composting and disposal would also be beneficial. Finding ways to further increase the amount of diverted waste would reduce the N load in landfills and recycle food-N to the soil. Engineered behavioral solutions are an option. For example, cafeterias that eliminate the use of trays (reducing the customer's ability to carry more than one plate at a time), have documented reductions in food waste (Hackes et al. 1997).

7.1.4 Energy and transportation sector efficiency

Reactive N released from fuel combustion has far-reaching consequences on air quality, human health, and downwind ecosystems. California's hot and dry climate and highly N-limited ecosystems only add to the problem. With the projected increases in population, climate change, and changes in land use, pressures on these resources will continue to intensify. Fuel combustion from transportation, energy production, and industrial processes is the major source of N to the atmosphere (40%), largely in the form of NO_x, NH_3 and N_2O emissions in California. NO_x is the predominant (89%) form of fossil fuel N generated and is almost solely created through the combustion process when high temperatures cause

N_2 to react with O. NO_x (usually in the form of NO from fossil fuel combustion) is a precursor to smog and contributor to particulate matter (PM)(Chapter 5). NH_3, a $PM_{2.5}$ precursor, makes up 9% of N emissions generated by fossil fuel combustion and stems from both stationary and mobile sources as a byproduct not of the combustion process, but of the catalytic process. N_2O comprises less than 3% of fossil fuel combustion emissions, but is a potent greenhouse gas with 298 times the global warming potential (GWP) of carbon dioxide (CO_2) (Chapter 4).

California has long recognized the major impact of fossil fuel combustion on air quality and in response has led the nation in combating emissions, primarily of NO_x. However, secondary air pollutants derived from N emissions (i.e., ozone and $PM_{2.5}$ and PM_{10}) still plague the health of Californians, costing hundreds of millions of dollars annually in health expenses (see Chapter 5; Hall et al. 2008, 2010). Additionally, airborne NO_x deposited downwind on the landscape changes soil stoichiometry, promotes invasive species, and preconditions ecosystems for wildfire; all threatening the persistence of sensitive natural ecosystems (Fenn et al. 2003, 2010; Chapter 5). Because of the significant and on-going concerns associated with N, decreasing emissions further remains a critical goal.

Efforts to minimize nitrogen emissions can be divided into two major categories—decreasing emissions from fuel combustion, and decreasing the overall amount of fuel combusted. Control technologies decrease emissions by transforming nitrogen emissions into nitrogen gas (N_2) or filtering nitrogen-containing particulate matter out of the exhaust before release into the atmosphere. Major steps have been taken to reduce emissions after the tail pipe—between 1999 and 2011, particulate matter in the Los Angeles air basin dropped by 47%, and dropped by 26% in the San Joaquin Valley (presentation by Tom Cackette, CARB). The potential for further improvements in these control technologies is limited (Section 7.2.4).

To see more drastic change, like that proposed in California's plan to reduce greenhouse gas emissions to 1990 levels by 2020 (AB32), it is generally agreed by most that decreasing fuel combustion altogether will be key to major reductions in greenhouse gas emissions and other nitrogen-based pollutants. Alternative fuels and alternative vehicles are promising guides to such improvements, and will be required to achieve deep reduction in N emissions without reducing vehicle demand. Simply stated, decreased fuel combustion will decrease N emissions at the source of combustion (mobile or stationary source). But such improvements are complicated by upstream emissions from power generation. Research to understand how nitrogen emissions are affected upstream is still cursory, as life cycle assessments of emissions generally focus on CO_2 and N_2O, and often do not include other nitrogen species. The nitrogen-relevant factors of these technologies are assessed below, with particular attention paid to upstream emissions that can be decreased by improved efficiency in electrical generation.

7.1.4.1. Fossil fuel use substitution in vehicles

Technologies currently in the market or on the horizon include Hybrid Electric Vehicles (HEVs), Plug-in Hybrid Electric Vehicles (PHEVs), Full Electric Vehicles (EVs), Fuel Cell vehicles (FCVs), Flex-fuel Vehicles (FFVs) (designed to run on gasoline of a blend of up to 85% ethanol), and Compressed Natural Gas (CNG) vehicles. In addition to these alternative designs, the use of ethanol and biodiesel fuel blends is expanding as a carbon-intensity reducing measure. The timeline between research and development of new vehicles and 50-75% market penetration may be as long as 50 years (Ogden and Anderson 2011), and requires policy development to both push for technology improvement and create the infrastructure to support major changes in vehicle fleet, including sufficient charging stations for electric vehicles and hydrogen storage for hydrogen fuel cell vehicles (Ogden and Anderson 2011).

While CO_2 emissions are relatively simple to estimate (as they are directly related to the carbon content of fuel), nitrous oxide is significantly more difficult to calculate and makes estimating the emissions of alternative fuels and vehicles hard to track. N_2O emissions are dependent on fuel combustion temperature, pressure and air-to-fuel ratio. Despite decreases in direct emissions from alternative- fuel vehicles and technologies, additional emissions stem from a variety of upstream processes such as resource extraction, electricity production, fuel transport, and fuel distribution. The time of day vehicles are charged presents a major uncertainty in measuring emissions. If the majority of PHEVs are charged at night, as many studies assume, their emissions will be dependent on the type of electricity used in the marginal electricity—the mix used at the end of the day or at non-peak times. If marginal electricity is derived from renewable sources, emissions will fare better than if marginal electricity comes from coal fired power plants or similar sources. Other variations in emissions can stem from driving patterns (such as length of trip) as well as the size of the vehicle itself (Lipman and Delucchi 2010).

Numerous life cycle assessments have been conducted to assess the various emissions levels from alternative fuel vehicles and the potential reduction that can come from improved fuel sources. Table 7.3 compares several life cycle assessments' estimates of the decrease in emissions from different vehicle types compared to the conventional internal combustion engine.

[Table 7.3]

While some life cycle assessments account solely for carbon dioxide emissions, the GREET model, created by the Argonne National Laboratory, accounts for N_2O emissions as well as other greenhouse gases, and represents cumulative emissions decreases as carbon dioxide equivalent (CO_2e) amounts. While N_2O is included in the GREET model, individual pollutants are generally not described in well-to-wheel vehicle studies.

The GREET model estimates that, with the existing California energy mix, which is largely produced by natural gas and renewable fuel sources, electric vehicles can reduce life cycle GHG emissions compared to conventional internal combustion engine vehicles by about 60%, while fuel cell vehicles using H_2 derived from natural gas can reduce lifecycle emissions by 50% (Lipman and Delucchi

2010). However, if that grid mix has a higher dependence on coal-based electricity generation than the California mix, electric vehicles could result in an overall *increase* in GHG emissions. With an entirely renewable fuel source, electric vehicles and fuel cell vehicles could nearly eliminate GHG emissions (Lipman and Delucchi 2010)

Electric vehicles also reduce NO_x emissions. The American Council for an Energy-Efficient Economy estimates that an all-electric vehicle powered by the average California power mix generates 2.3 lbs. (5 kg) NO_x over the course of a year (12,000 miles)[8], compared to 16-20 lbs. (36-44 kg) NO_x emissions from conventional vehicles. Hybrid vehicle NO_x emissions are estimated at 11 $lbs.yr^{-1}$ (24 kg yr^{-1}), and PHEVs using a California energy mix see a 40% reduction in NO_x emissions from today's hybrid vehicles (those with an average range of 50 mpg). These estimates, however, can be affected by the fuel efficiencies of different vehicles, as well as the time of day vehicles are recharged (unaccounted for in these estimates)(Kliesch and Langer 2006).

Despite these variables, it is generally agreed that the use of renewable energy sources will decrease the life cycle emissions, but that using an electricity mix derived largely from coal fired power plants and other non-renewable sources have the potential to increase GHG emissions. Long-term modeling using the Lifecycle Emissions Model (LEM) shows the greatest potential GHG reductions from hydrologic, nuclear, and biomass energy sources (Lipman and Delucchi 2010). California's grid mix is well-suited to house alternative fuel vehicles, and is moving towards being an even better provider of clean energy. California's 2013 in-state power generation included 60.5% natural gas, 8.9% nuclear power, 10.4% large hydro power, 0.5% coal power, and 19.6% renewable power (California Energy Commission 2014). Statewide use of renewable power (in-state generation and imports from out of state) totaled 18.7% of total electricity use in 2013. Governor Jerry Brown has mandated an increase to

[8] Estimates do not include emissions up-stream from electricity generation, such as mining and material transport.

33% renewable power use by 2020, which will bring significant increases in the efficiency of HEVs, PHEVs, and BEVs.

7.1.4.2. Well-to-Wheels Analysis of Biofuels

Biofuels are frequently discussed as a renewable fuel source and a potentially GHG-neutral alternative to fossil fuels (Chum et al. 2011). Substituting biofuels for gasoline can potentially reduce GHG emissions if one only focuses on the potential of feedstocks to replace fossil fuels and sequester carbon during the plant growth phase (Searchinger et al. 2008). However, when examining soil N_2O emissions induced by fertilizer use, all the upstream emissions for inputs, as well as other indirect effects of biofuel production, it is generally accepted that life cycle GHG emissions for common biofuels, especially corn ethanol, can be higher than those for fossil fuels, especially when considering global land use changes (NRC 2011; Searchinger et al. 2008). For example, Searchinger et al. (2008) found that the diversion of existing cropland into biofuel production triggers rising crop prices which in turn induce farmers around the world to convert hundreds of millions more hectares of forest and grasslands, (i.e., systems that are already providing carbon storage and sequestration potential), into cropland to increase crop production for feed and food. Similarly, assuming a conversion factor of 3-5% from synthetic N fertilizer to nitrous oxide (N_2O) from crop production systems, it is agreed but unproven that the next-generation cellulosic crops, such as perennial grasses and woody plants, are likely to provide substantial positive net benefits in reducing GHG emissions from fuel use (NRC 2011; Adler et al. 2007).

It is suggested but unproven that some of these same alternative biofuel crops in California could help manage environmental problems associated with intensive agricultural production and could contribute to overall agricultural sustainability. For example, switchgrass, one of the perennial cellulosic crops, is very salt-tolerant and therefore useful in agriculturally marginal areas, such as the western San

Joaquin Valley, where high salinity impedes production of other crops (Kaffka 2009). In addition, safflower, another alternative biofuel crop, can play a useful role in crop rotation with more valuable crops (e.g., tomatoes or cotton) in California as it can better utilize water and N fertilizer stored at greater soil depths (Kaffka 2009). Sugarbeets also seem to be a promising option, due to their deep soil N scavenging ability and increasing trend in overall resource-use efficiency in California (Kaffka 2009). These cross-cutting environmental benefits may raise the sustainability profile of alternative biofuel crops, and need to be figured into decisions to support development of these crops in California.

7.1.4.3. Fuel Combustion in Stationary Sources

Like mobile sources, stationary sources on fossil fuel combustion will benefit from increased use of renewable energy. That energy can come from new electricity sources—including wind power, solar, hydro and fuel cell. Improvements in power plant design that incorporate cogeneration or a gas-fired combined cycle can also increase overall efficiency. Both systems are designed to use excess heat created through the electricity generation as steam power. Reductions in NO_x from these designs will depend on the efficiency gain involved in the technology being replaced (Bradley and Jones 2002).

7.1.4.4. Reduction in travel demand

AB32 mandates that emissions levels in California decrease to 1990 levels by 2020. Additionally, California set a goal to drop emissions by 80% of current levels by 2050—a goal often referred to as 80in50 (Yang et al. 2009). Yang et al. (2009) model different strategies by which emissions could be reduced so drastically. Their scenarios, which model reductions only for in-state emissions (travel that originates and terminates within California), show that no single strategy for emission reductions can meet the 80in50 requirements, but that there are multiple strategies that can succeed together. In all three strategies examined, Yang et al. found that light-duty vehicle technologies will need to bring the majority of change, using a combined strategy of fuel efficiency and vehicles and carbon intensity of fuel generation. Biofuel-heavy and electric vehicle-heavy scenarios bring the most significant change to GHG emissions. However, as stated above, heavy reliance on biofuels may have tradeoffs in nitrogen emissions.

A key element to one of Yang et al.'s scenarios is a decrease in travel demand. A reduction in travel demand is one alternative to reduce GHG emissions without changing fuel, mode, or vehicle technology. The scenario suggests that a decrease in travel demand should account for nearly one quarter of emissions decreases (based on Yang et al.'s reference scenario). Achieving such dramatic decreases will require changes in the built environment that allow people to travel more easily without the use of passenger vehicles—including building more densely, increasing access to public transportation and potentially adding costs to driving (higher taxation on gasoline and parking costs). Bringing significant change from these measures will not be easy. Heres-Del-Valle and Niemeier (2011) suggests that decreasing vehicle miles traveled (VMT) by as little as 4% may require residential density increases of up to 29%, or increases in gasoline prices by 27% (Heres-Del-Valle and Niemeier 2011). Other studies show that public responsiveness to increases in gasoline prices is limited, and has reduced over time (Small and Van Dender 2007; Hughes et al. 2006). In addition, without improved public transportation infrastructures, higher gasoline prices may disproportionately affect lower income households who lack access to public transportation or must commute long distances to work. To adequately address emissions from fossil fuel combustion, however, will require a suite of changes not only to the technologies we use to combust fuel, but also in the lifestyles that depend heavily on fossil fuel combustion for transportation.

7.2 Mitigate the movement of reactive nitrogen among environmental systems

Critical control points (Table 7.1) exist in other parts of the N cascade, beyond the introduction of new reactive N. Once N has already been ‘fixed’, by natural or industrial means, or released via fuel combustion; it is still possible to mitigate its impact. Generally, each of the major N transfer pathways is dominated by a single activity. For example, animal manure management and fuel combustion are the primary sources of NH_3 volatilization and NO_x, to the atmosphere, respectively. The overwhelming importance of certain activities for specific N species suggests clear research, outreach, or policy priorities to target these concerns.

7.2.1 Ammonia volatilization from manure

Manure N that results from dairy, beef, egg, and meat bird production contributes the vast majority of NH_3 emissions to California’s atmosphere and impacts air quality and the health of downwind ecosystems (see Chapters 4 and 5). This is particularly true throughout the San Joaquin Valley where manure N produced by confined dairy operations contributes to high atmospheric concentration of NH_3 (Clarisse et al. 2009, 2010; Chen et al. 2007), a building block of particulate matter ($PM_{2.5}$),and biodiversity loss in desert and mountainous regions in Eastern California (Fenn et al. 2008, 2010). Therefore, becoming more N sustainable in California requires reducing NH_3 volatilization from manure.

Fortunately, many tactics already exist to reduce NH_3 emissions from animal manures, including frequent manure collection, anaerobic storage, composting, precision feeding, and use of nitrification inhibitors (Ndegwa et al. 2008; Xin et al. 2011; Appendix 7A; Table 7.4). Unfortunately, relative changes in emissions rates from either the common manure management systems (see Chapter 3) or ‘alternative practices’ are not well understood for the climatic and production conditions characteristic of California animal production systems CARB 2005). For example, what impact does increasing the frequency of manure collection with recycled lagoon water have on NH_3 emissions? On the one hand, more frequent flushing of freestalls transfers reactive urea N to the lagoon where depth and pH restrain volatilization. On the other hand, manure deposited in freestalls is collected with recycled wastewater, spreading urea and NH_4 thinly over the concrete/soiled surface and creating conditions conducive to NH_3 emissions (expansive boundary layer, wind, increased total ammoniacal N). Levels of uncertainty about emissions from open lot dairies or poultry facilities are similar. What are rates of NH_3 emissions from corrals under arid conditions of the Tulare Lake Basin, with minimal manure disturbance, distributed patches of moisture from urine, and high temperatures? Or will changes in proposed layer housing structures affect NH_3? One study from Canada, which has similar poultry production systems as California, has shown that layers housed in larger cages, where birds had more space, had a similar nitrogen utilization efficiency (35%) as layers housed in conventional cages (36%) (Neijat et al. 2011), but it is unclear how specifically NH_3 emissions would be affected by the change in housing.[9] So while there are many possible actions operations might take to control NH_3 already (Rotz 2004), the extent of their applicability to California production systems is suggested but unproven. As a result, predictions of the magnitude of effect or efficacy in general for specific interventions are difficult to estimate.

[Table 7.4]

In spite of the uncertainty in emission rates and the variation among operations, evidence suggests there are opportunities to reduce NH_3 emissions from manure management in California. Dairy production creates 79% of statewide manure N and hence dominates NH_3 production. The University of California Division of Natural Resources Committee of Experts reported estimates of NH_3 losses on a typical dairy in the Central Valley, including NH_3 volatilized from the production unit and during land application. While these estimates contain some uncertainty, the reported range of volatilization is approximately 25% to 50% of excreted manure N, a 100% difference between the least and greatest

[9] As of January 1, 2015, the California Shell Egg Food Safety regulation (3 CCR 1350) requires egg producers to provide a new minimum amount of floor space per egg-laying hen. See CDFA 2013.

producers (Chang et al. 2005). The wide distribution indicates there is substantial room for improvement, especially for the operators with the highest emissions rates. Assuming that extreme rates are not very common (e.g., emissions are normally distributed) and there is a differential in potential improvement because of the wide distribution, we suggest that NH_3 volatilization from manure can be reduced by approximately 4 percentage points on average and in total 10 to 15 Gg N $year^{-1}$ given current manure deposition rates (Appendix 7B).

Reducing NH_3 emissions from animal production units requires a whole-farm approach (Castillo 2009). Manure management involves a series of complex unit processes that link together to collect, process, treat, and store manure, with volatilization taking place throughout (Chapter 3; Castillo 2009). When volatilization decreases at any stage, N is conserved and transferred to the next process increasing the total N pool and the potential for emissions in subsequent stages of treatment and disposal. Reducing NH_3 emissions by changing practices for a single component of a manure management train is meaningless. While N conservation is a laudable goal, it must be recognized this ultimately increases the N utilization burden on animal production systems and potentially requires more land or capital for distribution. There is a need to better develop and build the evidence base for N conservation throughout manure management trains, not only individual practices and to identify the best leverage points to reduce losses. It cannot be accentuated enough that such a reduction would require a significant effort by dairies to distribute and recycle the additionally conserved N. In a positive note, the N conserved would largely be in the urea or NH_4 forms, which has higher fertilizer value because it is relatively plant available by comparison to organic N.

One primary constraint to the mitigation of NH_3 emissions from manure management is the cost of control technologies for the producer. Often the changes required increase the producer's cost of production, be it additional labor, more machine operating time, or monitoring and record keeping. The ability for producers to absorb additional costs of NH_3 management is questionable given the thin profit margins characteristic of recent milk markets, evidenced by the decline in numbers of dairies in the state.

7.2.2 Nitrate leaching from croplands

Because of the long time lag between cause and effect—commonly five to fifty or more years in California—reducing N loading to groundwater from croplands will not decrease groundwater NO_3^- concentrations in the short term, and groundwater nitrate concentrations will continue to increase in some locations irrespective of any remedial actions taken (Harter and Lund 2012; Dubrovsky et al. 2010). Regardless, reducing NO_3^- leaching losses from croplands is an important strategy to minimize future groundwater degradation and protect drinking water resources in the long term.

Recent analyses indicate that intensive food and feed production is responsible for widespread groundwater pollution in California's agricultural valleys (Chapter 4; Harter and Lund 2012). Together with more than twenty field studies (Rosenstock et al. 2014), several watershed scale estimates (Miller and Smith 1976), and stable isotope analysis (Burow et al. 2007, 2008; Fogg et al. 1998), there is strong evidence that historical and contemporary cropping practices clearly place groundwater resources at risk (Figure 7.2). However, it is important to recognize that leaching is an essential part of irrigated crop production in arid and semi-arid climates[10]. Without it, plant-toxic salts tend to accumulate within the rootzone and decrease production (Hanson et al. 2008, 2009). For this reason, continued productivity of many California cropping systems depends on transporting salts below the rootzone, which typically occurs with irrigation or precipitation. In such environments, tradeoffs need to be made between managing the soil salt balance for continued viability of farming operations, on the one hand, and the environmental impacts of NO_3^- leaching, on the other hand.

[Figure 7.2]

[10] In other climates, salts are leached below the rootzone by precipitation.

Although NO_3^- leaching and some groundwater contamination from California crop production is practically inevitable, growers have many options for relieving pressure on the resource (Appendix 7A). A recent review identified over fifty management measures that could help (Dzurella et al. 2012). The fundamental basis of managing leaching is that losses are correlated with N and water inputs (Letey et al. 1979; Addiscott 1996; Figure 7.3). Practices that closely monitor and manage soil water and N status over active cropping and fallow periods are effective at reducing losses (Feigin et al. 1982a, b; Jackson et al. 1994; Poudel et al. 2002; Hartz et al. 2000). Consequently, when N use and irrigation efficiency increase, losses decrease. High N and irrigation efficiency result in a small soil mineral N pool and longer residence times of N in the root zone. The latter has the dual benefit of increasing the potential for uptake as well as increasing the potential for denitrification because of the high degree of biological activity in this region. Often reducing leaching requires additional labor and capital resources, and possibly the adoption of new or advanced technologies (Addiscott 1996). But, optimizing the management of existing practices, such as shortening furrows or optimizing drip irrigation technology, can also be an effective strategy (Jackson et al. 2003; Jackson et al. 1994; Hanson et al. 1997; Breschini and Hartz 2002; Appendix 7A).

[Figure 7.3]

Virtually all modern cropping systems in California pose a NO_3^- leaching risk. But certain systems disproportionately affect groundwater. Differences in leaching potential are related to the soil physical properties, irrigation method, crop cultivated, and soil management practices (Pratt et al. 1984). Though actual leaching rates are location-specific due to the aforementioned factors, certain combinations of technologies, sites, and crop species present greater jeopardy. Researchers at the University of California Riverside led an initiative to create a system to identify NO_3^- leaching risk potential for irrigated crop production in the Western United States. The outcome, called the Nitrate Hazard Index, scores the threat of a cropping system based on soil, crop, and irrigation system characteristics (Wu et al. 2005). Knowledge about the vulnerability of the system can be used to guide management decisions, such as planting deep rooted crops, or removing a field from production altogether. Indeed, using such tools might help mitigate leaching. But it must be remembered, the Nitrate Hazard Index is simply a planning tool; management ultimately determines the leaching rates (Pang et al. 1997; Hanson 1995). Arresting cultivation of highly susceptible sites and managing crop-soil-technology combinations that minimize leaching hazard would further reduce NO_3^- leaching.

Our estimates suggest improved fertilizer, water, and soil management could avert at least 7 Gg N leaching losses each year. Reductions represent the minimum expectation when increasing N use efficiency by 8 percentage points (section 7.1.1). It is entirely plausible that leaching losses would be reduced an even greater extent with improved practice. Surplus soil mineral N is highly susceptible to leaching loss, with potential leaching losses rising exponentially after plant uptake is exceeded. Therefore, reducing the size of the pool by increasing N use efficiency is more likely to have an exponential instead of proportional effect.

But would reducing NO_3^- leaching have negative consequences for farm profits? Practices that reduce leaching are often a deviation from common farm practice and typically entail more intensive management, adding to production costs. Efforts to estimate costs are complicated by the number of operations that must be included and the uncertainty and variability in actual leaching rates for a given field. However, it appears leaching losses could be incrementally reduced without significantly affecting farm profits (Medellin-Azuara et al. 2012; Knapp and Schwabe 2008). Dramatic reductions in leaching may require transformative actions in irrigation, manure, and chemical fertilizer management. These transformations are hindered by numerous barriers on and off the farm, including farm logistical limitations to changing irrigation practices, insufficient development or local adaptation and demonstration of required technologies, insufficient grower education, and land tenure issues. . Costs and benefits to individual farmers, however, need to be appraised simultaneously with the costs borne by society at large due to groundwater contamination (e.g., costs of treatment or buying drinking water) and the benefits accruing from cheaper foodstuffs.

7.2.3 Greenhouse gas emissions from fertilizer use

Use of nitrogenous fertilizers is the primary cause of recent increases in atmospheric concentrations of N_2O globally (Crutzen et al. 2008; Davidson 2009; Wuebbles 2009; Ravishankara et al. 2009). In California, inorganic fertilizer use accounts for about 80% of the total N_2O emissions according to California's most recent greenhouse gas inventory (CARB 2014). When integrated over a 100-year timeframe[11], N_2O emissions amount to approximately 2% of California's total climate forcing emissions[12,13]. The relatively small fraction of annual emissions attributable to fertilizer use does not mean it should be dismissed or ignored despite that other sectors and activities contribute similarly sized portions (CARB 2011). The overwhelming dominance of N fertilizer use on California's N_2O budget calls for a recalibration of agriculture to a low-emission trajectory.

Unfortunately, due to the complexity of mechanisms driving N_2O evolution in soils, there are no agronomic "silver bullets" that universally, or even consistently, reduce N_2O emissions[14] (Appendix 7B). Soil physical and chemical properties (including texture, pH, oxygen and carbon availability, and water holding capacity); management practices (including tillage, irrigation, and fertilizer source and rate,

[11] Comparison of radiative forcing across the three dominant greenhouse gases (carbon dioxide, methane, and nitrous oxide) is done by converting emissions to the metric of carbon dioxide equivalents (CO2-e). Carbon dioxide equivalents are conversion factors to calibrate the radiative forcing of emissions over a 100-year timeframe because of the long-lived nature of N_2O in the atmosphere. Over 100-years, N_2O is 310 times as potent as carbon dioxide (CO_2) and methane (CH_4) is 21 (IPCC 2007).

[12] Total fertilizer N use equals approximately 2% but here we are simply discussing fertilizer use on croplands which Are approximately 90% of total sales.

[13] This figure ignores the substantial CO_2-equivalent emissions that accrue during out-of-state manufacture of the fertilizer, which increase the total GHG impact of fertilizer use by 20 to 150% (see box 5.4).

[14] This statement ignores that one could completely cease N fertilizer applications, either organic or inorganic, and N_2O would surely decline because this action is unrealistic if agriculture is to persist.

etc.); weather (including temperature and precipitation); and biological activity each affect the magnitude of fluxes and total emissions (Mosier et al. 1998; Stehfest and Bouwman 2006). Complex interactions among these factors cause large variance in direct emission rates from the field, with the Intergovernmental Panel on Climate Change estimating an uncertainty range of 0.003 – 0.03 kg N_2O-N per kg of N applied (IPCC 2008). The considerable spatiotemporal variability, within and among fields and farms—even when seemingly similar production conditions are present, complicates emissions predictions and control. A recent study measuring N_2O emissions from processing tomato systems in Yolo County illustrates the issues well. Kallenbach et al. (2010) compare emissions from treatments using subsurface drip and furrow irrigation with and without leguminous cover crops grown during the winter, between cash crops. Nitrous oxide emissions were greater when leguminous cover crops were planted compared to barren fields in the furrow-irrigated plots, as might have been expected because they are an additional source of N. However, subsurface irrigation negated the effect of the green manure and emitted less N_2O in comparison to the other treatments. Similar interactions have been found in studies of tillage (Six et al. 2004; Venterea et al. 2011; Mosier et al. 1998), as well as fertilizer placement, and other fertility management practices—e.g., the 4Rs[15] (Snyder et al. 2009). With highly site-specific responses, the limited number of field measurements in California, and concerns about measurement protocols and interpretation (Data tables), conclusions about the ability of individual or bundles of practices to reduce N_2O production and the consequential magnitude of any reduction for specific locations is largely speculative.

Somewhat more certain is that N_2O emissions correlate with N application rates. Therefore, practices that allow growers to reduce N use will generally induce mitigative benefits. The magnitude of the reductions depends on the nature of the relationship between N_2O and N fertilizer rate, with both

[15] The 4Rs typify the current N fertilizer management paradigm. Judicious fertilizer applications are those that use the **r**ight source, **r**ight amount, at the **r**ight time, in the **r**ight place (see Chapter 8)

linear and exponential functional forms being observed, which is controlled by the site-specific conditions identified previously (Figure 7.4) (McSwiney and Robertson 2005; Eagle et al. 2010. Expectations about the impact of marginal reductions of N use are then subject to assumptions of the relationship. If linear, then incremental change will have a proportional effect regardless of the magnitude of reduction. But if exponential, then decreases in N use can be expected to dramatically reduce emissions—assuming producers fertilize at rates greater than crop uptake. In this assessment, we assume a linear response function when estimating potential emission reductions. The assumption is reasonable when estimating emissions over scales as significant as California because field-to-field variation averages out once aggregated (Figure 7.4). Utilizing the median rate of emissions garnered from California specific studies (1.4% of N applied) and the increase in N use efficiency discussed above (section 7.1.1), we might expect to reduce emissions by 0.53 Gg N $year^{-1}$.

[Figure 7.4]

Field-level emissions responses are more likely described by exponential response functions, which is significant because relatively small reductions in N application may dramatically decrease emissions. That suggests that growers could participate in carbon finance schemes such as the Climate Action Reserve's N fertilizer reduction protocol (e.g., CARB 2011) without major chance of under-fertilizing their crop. In general, development of low N_2O production systems is only beginning in California, even though some of the seminal research on N_2O evolution from cropland soils occurred in California (Ryden et al. 1981). Recent research has aimed to set a baseline of emission rates for a range of systems. More comparative research is needed. With the diversity of cropping systems, uncertainty of the impacts of specific practices, and differential importance to state production, a targeted approach could set priorities for future research. Based simply on estimates of inorganic N fertilizer use, future research to develop low-emissions systems should initially focus on almonds and cotton, lettuce, tomatoes, and wheat (Rosenstock et al. 2012). Indeed, special attention may be paid to almonds, cotton, and lettuce as estimates suggest they are responsible for the largest amount of emissions for their respective crop type: perennials, field crops, and vegetables, respectively (Figure 7.5). Lessons learned from these crops can then be transferrable to other production systems with similar characteristics.

[Figure 7.5]

It is important to note that the discussion here so far has concentrated on direct emissions alone. Indirect emissions, those that occur after N is transported beyond the field boundaries due to initial volatilization, deposition or leaching/runoff, represent another source of N_2O to the atmosphere, though the expected magnitude of the flux is smaller. For example, IPCC default emissions factors for N_2O-N for N leached is 0.0075 with an uncertainty range 0.005-0.025 (IPCC 2008), only about 7.5% of expected direct field emissions.

7.2.4 Nitrogen oxide emissions from fuel combustion

NO_x released into the atmosphere in California from fossil fuel combustion is a major source of N (359 Gg N yr^{-1}) (Chapter 4). The major mobile contributors of NO_x include heavy duty diesel vehicles (28% of NO_x), light duty vehicles (14%), and ships and commercial boats (11%). Stationary sources of NO_x, including manufacturing/industrial sources and residential fuel combustion account for 125% of statewide NO_x (CARB 2013 Almanac (2014)). According to CARB (2007), it is feasible to reduce NO_x emission by more than 60.3 Gg in the South Coast, San Joaquin, and Sacramento Air Basins.

7.2.4.1. Mobile sources of nitrogen emissions: Light-duty vehicles

Little nitrogen exists in fuels for light-duty vehicles; rather, N is derived from the N in the air that serves to combust fuel. Emissions from light-duty vehicles are the result of incomplete combustion (releasing particulate matter) and high combustion temperatures (releasing NO_x). The primary way to reduce emissions from this source, without reducing vehicle activity or fuel switching, has historically been to reduce tail pipe emissions. Since the 1960s, a series of technologies have become available that either increase control of the air: fuel ratio and temperature during combustion or modify gas prior to release, which have had the impact of attenuating emission rates per vehicle mile traveled. Today, fuel injectors are used in all light duty vehicles to control the air: fuel ratio in vehicles, which helps to prevent incomplete combustion (Pulkrabek 2004). Exhaust gas recirculation systems recirculate 5-15% of exhaust back to engine intake, lowering combustion temperatures and decreasing NO_x emissions (Pulkrabek 2004). Exhaust Gas Recirculation was first introduced in 1973 and is common place in passenger vehicles today. Three-way catalytic convertors were added to vehicles beginning in the late 1970s to help lower combustion temperatures and decrease NO_x emissions and have become the standard form of NO_x emission decreases. Catalytic convertors serve to speed the fuel combustion chemical reaction, and in best case scenarios, can convert 95% of NO_x into inert N_2. Catalytic convertors are the most effective technology to reduce NO_x emissions from light duty vehicles, but the technology is not without its tradeoffs. Catalytic convertors are generally designed to decrease NO_x emissions, but may have a secondary impact on increasing N_2O and NH_3 production (Lipman and Delucchi 2010; Kean 2009).

While internal combustion engines do not normally reach the high temperatures required to produce N_2O, catalytic convertors, used to lower NO_x emissions, can create N_2O emissions as a by-product. Cold engine starts produce pulses of N_2O that decrease as engines warm up, and aging catalytic convertors emit more N_2O than younger ones. As hybrid vehicles gain market penetration, increasing N_2O emissions are a concern. As hybrid engines cycle on and off when vehicles start and stop, catalytic convertors can cool off enough to produce N_2O emissions multiple times throughout a vehicle's trip. To date, catalytic convertors are not produced to address both N_2O and other NO_x emissions, and the technology's potential requires significant research and development. Potential amendments include electrically heated catalytic convertors, though the heating may result in a small net energy loss for vehicles (Ogden and Anderson, 2011; Lipman and Delucchi, 2010).

The case is similar for NH_3 emissions from light-duty vehicles. Three-way catalytic convertors employ ammonia in the form of urea to help speed reactions and reduce NO_x to a steady state (N_2). Catalytic convertors can over-reduce NO_x beyond N_2, resulting in NH_3 emissions as part of vehicle exhaust. Because three-way catalytic convertors were not introduced until 1981, older vehicles without them produce almost no ammonia. Newer vehicles with efficient catalytic convertors also produce lower emissions, making the problem most abundant in middle-aged vehicles with aging catalytic convertors (Kean 2009). Other materials can substitute urea to reduce NO_x, and urea injections into catalytic convertors can be measured more precisely (Johnson 2009), but there is likely a tradeoff between lowering ammonia emissions and lowering NO_x emissions using the existing three-way catalytic convertor technology.

7.2.4.2. Mobile Sources of nitrogen emissions: Heavy-duty vehicles, ocean-going vessels and off-road vehicles

In the past, emissions controls used for light-duty vehicles could not apply to heavy-duty diesel trucks. Diesel trucks have historically had poor fuel injection control, resulting in poor control of particulate matter (PM) emissions. But there are promising advances in control technologies to reduce emissions from diesel trucks. Often, the turnover to newer engine models can effectively lower emissions (Dallmann, 2011). Vehicle turnover is slow, but California has mandated upgrades to many heavy-duty vehicles and replacing outdated fleets that, over time, will show significant impact on emissions derived from the goods movement industry. Low-sulfur fuel is now mandated for diesel trucks in California, and trucks are being equipped with better fuel injection systems, exhaust gas recirculation to lower combustion temperatures (reducing NO_x emissions) and diesel particulate filters used to trap particulate matter and burn it off intermittently (US EPA 2008; Pulkrabek, 2004). Diesel particulate filters are required in all new vehicles manufactured, and are a required addition to older engines under CARB's Truck and Bus Regulation (CARB 2014). The regulation also includes a scheduled phase-out of engines manufactured prior to 2010: by the end of 2023, all trucks are expected to meet 2010 engine emission standards and to be equipped with a diesel particulate filter. These technology improvements are anticipated to reduce PM emissions from goods movement by 86% by 2020, and NO_x emissions by up to 68% (CARB 2006). Selective Catalytic Reduction (SCR) is being phased into heavy-duty vehicles (a technology commonly used in stationary sources to reduce NO_x). While heavy duty trucks do not currently emit a significant amount of NH_3 (Kean 2009), the increased use of SCR, which uses urea as a NO_x reducing agent, could contribute to increases in NH_3 emissions (Kean 2009).

Ocean-going vessels (OGVs) contribute to 11% of California's NO_x emissions (CARB 2014), and a negligible amount of N_2O. In 2010 the US EPA and the International Maritime Organization officially designated waters within 200 nautical miles of North American coasts, including California, as an Emission Control Area (ECA). Between 2012 and 2016, OGVs operating within the North America ECA are required to reduce their emissions of NO_x, sulfur oxides (SO_x), and $PM_{2.5}$ through a graduated transition to increasingly lower-sulfur fuels (US EPA 2010). In addition, establishing electrical power for ships to use while docked will decrease emissions further. Ships can also generate their own electrical power through solar panels, fuel cells, or with natural gas engines equipped with SCR technology to control NO_x (CARB 2007). However, the introduction of catalytic convertors on ocean-going vessels will likely add the tradeoff of increased N_2O and possible NH_3 emissions.

Off-road diesel vehicles such as tractors and construction equipment are subject to the same technological needs as heavy-duty trucks in order to improve emissions. Low-cost improvements like adding a Diesel Oxidation Catalyst can cut particulate matter in half, but do not affect NO_x emissions.

Adding SCR technology to diesel engines, which can dramatically reduce NO_x emissions, can be cost prohibitive, ranging from $12,000-$20,000 (EPA 2008). The EPA also emphasizes vehicle replacement, short idling times and replacement of aging fleets as key ways to decrease emissions.

7.2.4.3. Stationary sources of NO_x and N_2O

Stationary sources of fuel combustion, including energy generating power plants and manufacturing, comprise about 8% of California's NO_x inventory (Cal EPA 2009; Chapter 3), and 80% of emissions were derived from only 187 facilities in 2007, so the path to lower NO_x emissions is relatively achievable, though retrofits can be cost prohibitive. NO_x emissions are dependent on a number of factors at industrial facilities including flame temperature, residence time at high temperature, quantity of excess air available for combustion, and nitrogen content of the fuel (Bradley and Jones 2002). There are a number of combustion and post-combustion technologies in place to control NO_x emissions from stationary sources (Table 7.5). Reducing peak temperatures, reducing the gas residence time near the flame or reducing oxygen concentrations by low excess air, staged combustion, over-fired air, and flue gas recirculation in the zone of combustion are already commonplace measures that achieve substantial reductions in NO_x emissions (CARB 2011)[16]. Selective Catalytic Reduction (SCR) and Selective Non-catalytic reduction (SNCR) are both frequently used to reduce NO_x to nitrogen and water using ammonia as a reducing agent, presenting similar tradeoffs as mobile sources. SNCR can reduce NO_x emissions by 60%, while SCR can reduce NO_x emissions by as much as 95% (Bradley and Jones 2002; Table 7.5).

[Table 7.5]

Emissions of N_2O from most industrial sources are extremely low (CARB 2014). N_2O from stationary sources generally originates either as a product of incomplete fuel combustion or as a

[16] A full list of technologies used to reduce NO_x emissions from stationary sources as well as their cost effectiveness is available through CARB (http://www.arb.ca.gov/mandrpts/NOxdoc/NOxdoc.pdf

product of adipic acid (used primarily to make plastics) and nitric acid production (used for fertilizer, plastics and explosives). N_2O originating from adipic acid can largely be reduced by N_2O destruction (incineration) while nitric acid-based N_2O requires catalytic reduction. Nitric Acid facilities generally use the same SCR to control NO_x and N_2O emissions, but the system is designed primarily to control NO_x and is therefore significantly less effective at controlling N_2O (Johnson 2009). A third control system, Non-Selective Catalytic Reduction (NSCR) is very effective at controlling both NO_x and N_2O, but is used by few nitric acid plants because of high energy costs (CCTP 2006). The US Climate Change Technology Program emphasizes the need to improve SNCR technologies and encourage research that focuses on simultaneous reduction of N_2O and NO_x[17].

7.2.5 Wastewater management

Until recently, wastewater was discharged without specific treatment for N to the detriment of California's drinking water, wildlife, climate, and ecosystems (Jassby et al. 2005; Gilbert 2010; CARB 2011; Seitzinger et al. 2006; Boehm and Paytan 2010). Today, about 50% receives treatment to decrease its N load prior to release into soils, freshwater, or coastal regions (Chapter 3). However, traditional notions of wastewater N treatment—removal and discharge—ignore ancillary environmental consequences and the nutritive value of this resource. Wastewater N management could be transformed to expand N removal where appropriate and stimulate recycling when possible.

The first goal of wastewater N management is to ensure it is not contributing to degradation of ecosystem services. The most realistic way to accomplish this in the short term is to reduce the N load of wastewater by expanding advanced treatment. Technologies capable of reducing the N load from 40% to 99% of untreated levels are well established for wastewater treatment plants (WWTP) and onsite wastewater treatment systems (OWTS) (Henze 1991; Henze 2008; Kang et al. 2008). Currently N

[17] Other options for addressing N_2O emissions are available through CARB's Clearinghouse of Non-CO_2 greenhouse gas emissions control technologies. http://www.arb.ca.gov/cc/non-co2-clearinghouse/non-co2-clearinghouse.htm#Nitrous_Oxide

treatments largely utilize processes that reduce the N load by creating conditions to support microbial nitrification (oxidation of NH_4 to NO_3^-) and denitrification (reduction of NO_3^- to N_2 gas). Its effectiveness and relative cost make this the most attractive option (Ahn 2006). However, N removal from wastewater and utilization of nitrification-denitrification has drawbacks. Biological N removal can cause N_2O to be emitted during both nitrification and denitrification (Townsend-Small et al. 2011) at rates from 0.5% to 14.6% of the N in wastewater at WWTPs (Kampschreur et al. 2009). Similar concerns likely affect OWTS using nitrification-denitrification to an even greater extent since their operators have little or no control over critical environment conditions regulating waste digestion (e.g., chemical composition, pH, flow, organic carbon). So while options are available that would further significantly reduce wastewater N load prior to discharge, advanced treatment presents environmental tradeoffs.

We estimate that improved wastewater management could greatly decrease N in effluent from WWTPs and OWTS. A conservative increase in N treatment at WWTPs (10% of influent) would reduce N discharged into the environment by 15.6 Gg N yr^{-1}. And depending on the extent of OWTS retrofits and operations, an additional 1.3 to 10.9 Gg N yr^{-1} could be removed.

More widespread N treatment of wastewater is a promising goal. With worsening nutrient scarcity, increasing energy costs for treatment, and rising awareness of the environmental impacts of N, recognizing wastewater nutrients as a latent resource and recycling them to landscapes will have to become a more prevalent part of the wastewater management portfolio. Source separation of human waste is an emerging strategy to handle N rich waters stemming from toilets. Most of the constituent mass of N in wastewater is in urine (≈70% to 80% of the total) (Metcalf and Eddy 2003). With urine separation technology, N can be recycled back to the landscape more easily, saving energy and recycling nutrients to the soil. Source separation technology, in which urine is removed from the waste stream and reused as a fertilizer, can be expected to reduce N loading to wastewater treatment systems by about 50%.

High costs significantly constrain advanced treatment applications for large-scale facilities and homeowners alike. A synthesis of costs shows that capital costs and operations and maintenance costs attributed to N removal can range from $1.08 - $8.51 per kg N removed and $1.08 - $2.00 per kg N , respectively (Kang et al. 2008). The large range reflects differences in the extent of the retrofit or expansion necessary, the specific technology applied, and the amount of wastewater processed. Economies of scale reduce per unit costs for many of the WWTPs reviewed. Based on a median rate, we estimate that it would cost roughly $214 million in capital expenditures to implement N reduction technologies across untreated wastewater throughout WWTPs in California, plus an additional $69 million annually for operation and maintenance. Relative costs for retrofitting or replacing septic systems are also high. Retrofitting an existing system can be $10,000 to $20,000 each (Viers et al. 2012). Another option is to treat effluent emerging from septic tank via biological nitrification and denitrification treatment. Wood chip bioreactors have been shown to reduce influent nitrate by 74 – 91% (Leverenz et al. 2010), with costs ranging from $10,000 - $20,000 to retrofit existing septic systems.

It is impractical, or at least uneconomical, to contend all California wastewater be treated for N given much of it is dumped untreated into the Pacific Ocean. However, the economics of treatment for WWTPs and homeowners needs to be counterbalanced by acknowledgement of the significant indirect impacts, be they ecosystem regime shifts or N_2O emissions that accompany such actions. A thorough assessment of the sensitivity and vulnerability of receiving ecosystems would help to set priorities for future N reductions.

7.3 Adapt to a nitrogen-rich environment

Reactive N is already affecting California's environment and dynamics of the N cascade dictate that further change will continue to occur for some time. Going forward, Californians will have to adapt their behavior to the new state of air, water, and soil resources to reduce exposure risks, maintain productivity, and relieve pressure on the environment. The health of California's populace and rural economy will depend on foresight, planning, and collective action to address imminent N concerns head-on.

7.3.1. Treatment and alternative sources of drinking water

Poor water quality disproportionately affects the most vulnerable citizens among us. A recent study suggests that the rural poor, mostly those of color, access water with particularly high NO_3^- levels (Balazs et al. 2011). That is largely the consequence of the fact that these populations tend to be served by small water systems drawing water from shallow wells which are located in agricultural regions that have seen large N inputs, receive migrating NO_3^- sooner because they are closer to the soil surface, and are sparsely distributed, thereby limiting treatment options. Environmental justice concerns of drinking water contamination is only recently coming into perspective (Harter and Lund 2012). Significant uncertainties still persist about the extent of the concerns and the best solutions (Honeycutt et al. 2012). Yet the dynamics of the problem (large N load migrating through soil profile, shallow wells, unequal cost of treatment burden, few resources available to adapt) align to suggest that the threat is significant and will only worsen and spread to many additional communities (Harter and Lund 2012). Special attention to the ability of marginalized populations in California to obtain safe drinking water may help avert a health crisis.

Though reducing NO_3^- leaching loss will be instrumental for meeting future drinking water needs, the concentration of NO_3^- in drinking water already exceeds safe levels—the legal maximum contaminant level (MCL, 10 mg/L NO_3^--N)—in many regions and remedial actions are needed to minimize exposure (Figure 7.3). Simply put, drinking water will require treatment for the foreseeable future in some areas because it will take decades before groundwater shows the impact of changes in surficial management practices.

Options to treat drinking water supplies for NO_3^- that are proven effective include both removal and reduction technologies, but they are highly site-specific. Siedel et al. (2011)[18] thoroughly review the major options including ion exchange, reverse osmosis, electrodialysis, and biological and chemical denitrification. Because each has clear advantages and disadvantages, selecting the 'best' option cannot be done *a priori*. Characteristics of the water system and water quality must be taken into account. Decisions about cost, waste disposal, information demands, size of the facility and future needs of the community need to be considered, at minimum. Planning for future needs and local conditions is particularly important because of inherent limitations of treatment systems and the demands they place on the community and/or operators. For example, small water systems often lack technical, managerial and financial capacity to mitigate NO_3^- issues and the available funding may cover initial capital cost but not operations and maintenance. Moreover, the use of some technologies such as anion exchange—one of the most common in NO_3^- treatment—requires salt and results in a brine which needs to be disposed of, which can be a significant cost especially for inland communities. In many cases, avoiding the challenges of treatment by developing new water resources instead may be more feasible. However, the long-term sustainability of non-treatment option needs to be considered as with the migration of NO_3^- into groundwater increases with time, some alternatives such as blending or drilling new wells may be feasible now but may not be in the future. While planning for the future, interim solutions including point-of-use may well be needed to deliver safe drinking water.

Because treating for NO_3^- in drinking water can be quite costly (both in initial capital costs as well as operations and maintenance costs) and technically challenging, options for simply avoiding the

[18] Readers are directed to Seidel et al. (2011) and Jensen et al. (2014) for detailed analyses of NO_3- treatment options for drinking water, including applicability, efficacy, costs, trade-offs, case studies, and many examples from California water systems.

high NO_3^- water altogether, or adjusting to it in other ways, are often explored first. Commonly used options in California are well inactivation, blending high NO_3^- water with water from other wells in which concentrations are lower/consolidation with nearby water systems, and development of alternative sources. New wells are often drilled deeper than older wells, in order to reach older groundwater containing less NO_3^-. This strategy, besides being more expensive, also often creates other challenges. For example, deeper water more often contains high levels of arsenic, which may need to be treated for in order to make the water safe for drinking.

In summary, when considering water treatment options together with non-treatment alternatives, an array of management options are available to provide clean drinking water for Californians. Costs, however, can be high. An assessment of the costs for supplying drinking water to populations serviced by high NO_3^- wells in the Tulare Lake Basin and Salinas Valley indicates 12 to 17 million USD $year^{-1}$ are needed to provide water for only 220,000 people (Honeycutt et al. 2012). More densely populated areas would have a lower per capita cost because of economies of scale, yet low NO_3^- water will not come cheap.

When considering all the options for adapting to NO_3^- - rich groundwater, care must be taken to evaluate the relative advantages and disadvantages among them, considering appropriate initial and ongoing capital, labor, and information demands, time scales, and development scenarios—and not simply relative costs.

7.3.2. Adaptation of agricultural systems

Farmers already adapt to N in California's environment. The most obvious example is when growers modify fertility programs to account for NO_3^- levels in irrigation water, allowing it to supplement or completely replace purchased fertilizer N inputs (e.g., Hutmacher et al. 2004). Less attention is paid to airborne N pollutants, despite the prospects for significant economic consequences. Exposure to elevated ambient concentrations of ground-level ozone (O_3) reduces yields, sometimes by nearly 20%, costing producers millions of dollars in lost revenue each year (Grantz 2003; Mutters and Soret 1998; Kim et al. 1998). But few producers select crops or varieties based on O_3 tolerance. As concentrations of N compounds continue to increase in the environment, adapting to these new levels will become a matter of necessity to maintain the productivity of agricultural production systems.

In addition to environmental changes, N-related regulatory changes will also require agriculture to sharpen its adaptive capacity[19]. Concerns of N in the environment are gaining traction in the public domain and N is taking center stage in ongoing state and federal policy discourse. The US Department of Agriculture (USDA), State Water Resources Control Board (SWRCB), EPA, CARB, and local counterparts (e.g., Regional Water Control Boards) have recently examined N use in agriculture. On top of the relatively long-standing air and water quality rules that include NO_x emissions and surface and groundwater maximum contaminant loads, scoping and implementation for statewide regulations and incentives to limit N_2O and further constrain NO_3^- - emissions are under development (e.g., , the Irrigated Agricultural Lands Waiver, the General Order on Dairy Waste Discharge). Reactive N use for every agricultural commodity, in every part of the state, will likely fall under at least one of their jurisdictions, if enacted. Since most of the regulations and incentives are still being discussed or developed, there is considerable ambiguity about their requirements. This uncertainty concerning regulations coupled with continuous changes in environmental conditions complicate the agricultural production environment.

In some ways, the very characteristics that have made California farms competitive in the global marketplace also may make them more vulnerable to N-induced changes in the environment and policy landscape. Relatively large fields and farms, high infrastructure investments, advanced and specialized

[19] Adaptive capacity is defined as the physical and capital resources and the ability to apply those resources in response to external stimuli.

technology, and specialization in certain commodities[20] create the high efficiency agriculture California is known for worldwide. Efficiency has resulted from intensification and specialization, reducing the diversity of management options. Technical options that help producers maximize efficiency and maintain elasticity will be in high demand.

At the state level, however, the diversity of California's product mix allows for a certain degree of plasticity. There is a wide range of knowledge and experience within the agricultural sector overall, due to its diverse array of production systems. Therefore, opportunities may exist to move quickly to adapt to changes in N by modifying production practices and moving between crops. That ability relies on information that will need to be organized, generated, and distributed in a timely and efficient way, and possibly financial incentives to assist with high upfront costs to change expensive infrastructure.

Enhancing the adaptive capacity of California agriculture to environmental, economic, and policy perturbations related to N will require a novel perspective on the form, function, and purpose of the system. Currently, the thresholds that will determine when California agriculture will beforced to make large and fundamental changes to avoid collapse are largely unknown. A few bioeconomic models predict California agriculture's response to N-rich environments and changing policies. They tentatively suggest that incremental change, such as shifting crop species to adapt to O_3 or changing soil management practices to reduce NO_3^- leaching modestly, is plausible without significant economic loss (Knapp and Schwabe 2008; Kim et al. 1998). For the most part, models are created based on feasible expectation for future environmental and policy conditions. Still, N may force California to face a more transformative moment, one that integrates across N sources, species, and impacts. In such cases, assumptions based on previous conditions would be irrelevant. Expecting the unexpected, although

[20] California's commodity mix limits adaptation because incremental short-term adjustments are difficult, if not impossible to achieve. Perennials and dairy systems are highly specialized, stationary production systems that require large upfront capital expenditures. Though a large variety of commodities are produced, few contribute significantly to total agricultural production.

always intrinsic in agriculture, will need to become the norm. Practices and institutions will need to support transitions, whether incremental or transformative[21].

7.4 Synergies and tradeoffs among nitrogen species

The strategies identified to control the N cascade can have far reaching effects, for target N species, non-target N species and environmental systems. Some actions will cause synergistic responses, reducing multiple N emissions simultaneously while improving the state of additional environmental and health concerns. Oftentimes, however, they will induce tradeoffs, where reduction of one N concern inflames another (Box 7.2). Secondary impacts arise from the ubiquity of N in living things, its presence in day-to-day human activities, and its interaction with the carbon and hydrologic cycles. Understanding the potential positive and negative unintended consequences is essential to evaluating the relevance of any particular N response activity.

[Box 7.2]

Implementation of the strategic actions will most certainly modify N cycling in California. For those that systemically address the N cascade, by reducing the amount of N put into circulation (section 7.1), an across-the-board reduction of emissions can be expected[22]. Economic benefits for the actors may result as well, in the form of fertilizer cost savings, for example. The potential of such strategies to bring about simultaneous, multi-N species emission reductions with concomitant economic gains and other co-benefits, merits particular attention.. However, half of the prescribed activities aim at

[21] Shifts among alternative system states due to adaptation may be incremental, as when the grower slightly modifies practices, or may have to be transformative, as when production of a particular crop changes regions or is eliminated altogether.

[22] Specific technologies will inherently alter the relative rates of N emissions and thus while total N emissions will decrease across N compounds, the benefits will likely be uneven across emissions pathways. Precise proportions will ultimately depend on the production conditions and technology used.

individual N transfers. Their limited scope combined with the intrinsic mobility of reactive N[23] increases the likelihood of unintentional emissions. This so called "pollution swapping" essentially reallocates the environmental and human health burden from one ecosystem service or economic sector to another, with occasionally more harmful consequences than the original pollution. Each mitigative action that focuses narrowly on a single activity and pollutant poses such threats (section 7.2). Some significant tradeoffs and synergies are described below[24,25], though given the nature of the N cascade others are plausible.

Minimization of ammonia volatilization from manure: *NH_3* ***(-),*** *NO_3^-* ***(+), N_2O (+)***

Avoiding NH_3 volatilization by improving manure management benefits downwind ecosystems and will help decrease particulate matter formation in the atmosphere. But by reducing NH_3, the likelihood of NO_3^- leaching and N_2O emissions will increase (Velthof et al. 2009), because the manure retains a greater N load than it would have had otherwise. Assuming the additional N is conveyed throughout the manure management train (e.g., collection, processing and storage facilities), croplands must absorb the additional load. Increased N load requires a larger application area or increases the risk of over-application, if additional land is not available for distribution. Even when manure N is applied judiciously, the increased N load itself will likely lead to higher fluxes of NO_3^- leaching to groundwater and gaseous N_2O emissions because of the greater loading to the soil. Indeed, a fraction of the original NH_3 emitted would have deposited downwind and been lost via these pathways anyway. However, the relative quantity of losses via leaching and denitrification would be less than expected from the increased N

[23] Current regulatory activities have the propensity to increase tradeoffs because of the narrow focus on specific N species for specific media (e.g., NH_3 in air).

[24] Signs refer to direction of flow. + = Increasing, - = decreasing. Colors refer to hazard. Green = positive benefits, red = negative

[25] See Chapter 5 of this report for a discussion of the effects of N on environmental and human health.

loads applied to crop fields directly; deposition of airborne NH_3 represents only approximately 20% of applied N and only 1% of that amount is lost as N_2O versus 2% from the original load of manure (assuming IPCC 2006 default emissions factors). Therefore, California decision makers are left weighing the impacts of NH_3 on natural ecosystems (including the potential for fire, invasive species, and biological diversity) and air quality (including $PM_{2.5}$ production) in the case where no additional effort is made to decrease volatilization, versus increased climate change impacts, ozone depletion, and groundwater degradation in the case where volatilization is actively minimized.

Reduction of nitrate leaching from croplands: *NO_3^-* ***(-), N_2O (+)***

Reducing leaching from croplands, without decreasing N application, requires NO_3^- to be better timed with crop demand or remain in the rootzone longer. Greater residence times—through decreased percolation or extending the release of the soil N pool—provide additional opportunities for plant roots to seek out and assimilate the NO_3^-, converting it eventually into organic molecules. It also provides a chance for microbes to denitrify the NO_3^- to N_2[26], especially in heavy clay soils. The efficacy of denitrifying bacteria to completely transform NO_3^- to N_2 depends on soil conditions (water content, organic carbon availability, pH, and temperature). And in the absence of the appropriate reducing conditions, denitrifying bacteria produce intermediary products of NO and N_2O, instead of the inert and desirable N_2. Wetting and drying cycles consistent with optimal N and water management tend to promote environmental conditions conducive for N_2O evolution. Soil heterogeneity only compounds this problem, making it more difficult to maintain denitrifying conditions and producing hotspots and hot moments of N_2O volatilization. California crop producers (and those that regulate them) must decide between practices that preserve groundwater at the expense of climate change. The tradeoff here is particularly pertinent as it juxtaposes a local with a global concern.

[26] Biological activity and organic C content is typically highest within the rootzone.

Emissions reductions from fuel combustion: NO_x (-), NH_3 (+)

Combustion technologies already effectively limit NO_x emissions from transportation and industry. As discussed, additional gains are plausible, especially at unregulated sources or by improving conversion efficiency of technologies. Certain technologies that use postcombustion catalysts to transform NO_x to N_2, however, have the potential to produce NH_3 instead of N_2. This is common in industrial applications, where "ammonia slip" results from aging catalysts or too little reaction time. Potentially more troublesome because of the relative ubiquity of the source activity, is the increased production of NH_3 from vehicle engines using 3-way catalytic converters. Under today's driving environment (congestion, low speeds), conditions promote less reduction to N_2 and, consequently, NH_3 becomes a larger fraction of tailpipe emissions. What this means is that the relative proportion of oxidized N (NO_x) to reduced N (NH_3) is changing in the atmosphere, with NO_x decreasing and NH_3 increasing. In short, efforts to control NO_x contribute to the increase in NH_3 in the atmosphere.

Transformation of wastewater management: NH_4 (-), *NO_3^-* ***(-), N_2O (+)***

Nitrogen removal from wastewater at WWTP and with OWTS almost exclusively relies on microbial nitrification and denitrification at this time. Fortuitously, the process tends to result in lower concentrations of NH_4 and NO_3^- in wastewater effluent with reduced N loading to the soils, rivers, and ocean environments, assuming discharge patterns remain unchanged. However, a larger amount of the N is released to the atmosphere as N_2O. According to one study of WWTP in Southern California, emissions of N_2O at WWTP utilizing advanced technology to remove N can be three times as high as emissions at facilities that do not use advanced N removal technology (Townsend-Small et al. 2011). A fraction of the emissions occur during nitrification. But most result from incomplete denitrification, as the wetting and drying cycles of N and carbon rich materials present ideal circumstances for microbial activity. Even under the tightly controlled environs, it is challenging to virtually eliminate N_2O. Treatment of wastewater at WWTPs in California serves to protect sensitive aquatic ecosystems for endangered species habitat and recreation or groundwater resources. While essential to avoid degradation, it is important to recognize that this protection is achieved at the expense of negative impacts on climate and the ozone layer.

7.5 Policies that unintentionally distort the nitrogen cascade

Many federal and state policies protect natural resources by limiting reactive N (Chapter 8). For some N species and sources, regulations attempt to moderate N movement and accumulation directly, as is the case with countless air quality rules imposed by the CARB and local air quality control districts or General Order for Dairy Waste Discharge being implemented by the RWCB Region 5. A few of the transfers described in Section 7.2 fit within this category, with the exceptions of N_2O emissions from fertilizer use and NH_3 from manure management, which are currently unregulated. Perhaps equally important to the unregulated sources though, is to understand the potential for policies to incentivize or obstruct Californians ability to manage the N cascade more effectively. In certain cases, current policies, unrelated to N, unintentionally influence N management indirectly through secondary mechanisms. Due to the indirect nature of these mechanisms, they are often not immediately apparent to policy makers and have not yet been thoroughly researched. Therefore, it is impossible to determine the extent or magnitude of their distortions of the N cascade, at this time. However, explicitly calling attention to these policies and their links to the N cascade underlies the development of a systematic approach to addressing it.

Ethanol production

US government policies promote the use of biofuels. Farmers across the mid-Western US have responded by producing corn-ethanol. A byproduct of corn-ethanol distillation is 'distillers grain'. Distiller grains are often used as ruminants feed. One reason they make good fodder is because they are a cost effective source of N, which tends to be relatively expensive per unit from other sources (e.g., alfalfa). Distillers' grain becomes a protein supplement for the animal. Concerns from utilization of distillers grain arise from its high N content which can lead to excessive amounts of N excreted and in manure (Hao et al. 2009). Excessive N excretion creates mobile N and can lead to environmental pollution. On the other hand, N in manure may provide a cheap alternative to inorganic sources if managed appropriately. The difficulty of managing diets including distillers is not unreasonable given it has just been developed recently and there is still much ongoing research on digestibility and solubility. While distillers grain present opportunities to recycle nutrients and to reduce production costs, the large influx of N causes environmental concerns.

7.6 The need for multi-source and multi-media solutions

This chapter focuses on strategic actions that California may take today to balance the N challenges. Unfortunately, many of the currently available and utilized approaches are narrowly focused around specific N source and impacts. Efforts to respond to N challenges must be structured in a way to address multiple components both from technical field perspectives and from environmental perspectives. Actions considering multiple N species simultaneously will support more efficient and effective strategies for N management. Fortunately, this assessment finds that many management practices and technologies are already available. However, continued environmental degradation despite the existence of effective control technologies leads this assessment to conclude that the challenge is only in part technical. Policies to promote adoption are also needed to create positive changes in California's N landscape (see Chapter 8).

[Box 7.3] [Box 7.4]

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Box 7.1 Can California crop production "go organic"? [Navigate back to text]

Organic fertilizers are thought by some to be more environmentally benign than their inorganic counterparts, resulting in a call for a paradigm shift in fertility management. Ignoring the debate that surrounds this assumption (Appendix 7B), we took a basic mass balance approach to consider the questions; can California crop production "go organic"? And if so, what would it take? Conversion would require organic N to be available in sufficient quantities to meet crop demand and sustain productivity and farm profitability. Current evidence raises doubts that either criterion could be met without significant transformation of systems and landscapes.

Food and feed crops exported approximately 347 Gg of non-biologically fixed N from the field in 2005 (see Chapter 4). Because N exports do not typically account for N in non-edible portions that remain in the field (crop residues) and it is impossible to exactly match crop uptake, exported N is only a fraction of the total required for production. For the sake of simplicity, lets assume inorganic N and organic N are used with equal efficiency. That is, exported N is an average of 54% of total N applied (see Chapter 3)[27]. That means 643 Gg N is actually required to meet crop demand at current levels of N export from fields.

Where would this quantity of N come from? Organic systems primarily use manures, composts and leguminous crops to enhance soil N supply. In 2005, manure production was 416 Gg N. If we assume that 30% is lost during processing via volatilization (US EPA 2004), 292 Gg are available, 45% of the total required. Unless the animal population increased or manure was imported into the state, the approximately 351 Gg remaining would have to be derived from planting leguminous crops. Green manures grown in California can be expected to fix atmospheric N at levels equivalent to 52 to 226 kg

[27] There is little evidence to suggest that using organic N sources are more efficient than inorganic N (e.g., Cassman et al. 2003; Crews and Peoples 2005). Further, because only a fraction of the organic N applied becomes plant available during the growing season, growers often apply N well in excess until soils reach equilibrium, where N inputs equal available N (Pratt 1979). Thus, 54% is likely even a conservative estimate.

ha^{-1}, depending on species, environmental conditions, and length of growing period (Shennan 1992). Based on these estimates, legumes would need to be cultivated on 1.6 to 6.9 million ha, or on 32 to 141% of the currently irrigated cropland. Considering the propensity for double cropping, growing two crops successively on the same piece of ground with fallow periods typically less than five months, the uncertainty in required N and fixation rates, and the high cost of transporting bulky manure, the feasibility of using organic N sources to make up the N deficit is questionable within the current agricultural system.

Let us, however, assume equivalent levels of N can be delivered via organic materials as typically applied in conventional systems. Then the question becomes whether organic N-based systems would sustain current levels of productivity and profitability. While some research trials demonstrate crop yields of certified organic systems (which includes many other practices beyond nutrient management) can be similar to those of crops produced by conventional means (Howarth et al. 2002), other analyses suggest otherwise. For example, Seufert et al. (2012). found evidence in a global meta-analysis that organic yields might be more N-limited than conventional yields in many contexts.

However, producing the same quantity of food and fiber is only one possible objective when asking whether California can "go organic" in terms of N sources. Ultimately, profitability of a practice is a large determinant of whether it can be adopted on a large scale, and profitability results from the relationship between production costs and returns. These realities prompt a number of questions, beyond the scope of this current assessment to address. For example, with continuing changes in fuel prices and state agricultural policies, how will the future costs of inorganic fertilizer compare to the costs of implementing more widespread use of organic forms of N, and how will farmers respond to these cost differences? Will conservation payments to farmers be available to help offset the costs and technical challenges of using more organic sources of N?

Finally, how much substitution of organic for synthetic sources of N is even necessary to achieve environmental gains while maintaining crop productivity and farm profitability? For example, research in Michigan suggests that a reduced input system using only 30% of conventional fertilizer input and adding a leguminous cover crop can sustain conventional level grain yields while accruing substantial soil quality improvements (Bhardwaj et al. 2011). Can similar effects be achieved for California crops?

Ultimately, while switching to organic sources of N can make important contributions, the magnitude and complexity of the N challenge mean that no individual practices or systems—be they conventional, organic, low-input, integrated, biodynamic, bio-intensive or whatever else—will solve the problem alone. Organic practices must be one arrow in a quiver of solutions, along with many others. Extended focus or overemphasis on any one solution detracts from the development, refinement, and outreach of the diverse site-specific systems that will be required to make significant inroads in reducing N pollution on a statewide basis.

Box 7.2. Lifecycle accounting and pollution trading: Next generation decision-making [Navigate back to text]

Control technologies have historically been and, for the most part, are still evaluated based on their ability to impact or regulate specific N species from a particular source. Emphasis on individual transfers of N, without systemic consideration of the entire N cascade, can result in exchanging one N pollutant for another (as discussed in Section 7.2). Risks of pollution swapping extend throughout the supply chain and can even induce non-N pollutants. The wider environmental context needs to be considered to determine the value and appropriateness of a control technology. Unintended consequences may results when practice efficacy is defined too narrowly.

To begin with, the N cascade is inextricably linked with the carbon (C) cycle. As a result, fertilizer and food production, transportation and industrial combustion, soil processes, and waste processing and disposal affect both biogeochemical cycles simultaneously. The implication is that, in many cases, the perturbation of one cycle cannot be fully assessed without including effects on the other and implementation of risk reduction strategies can create tradeoffs among emissions of various elements.

A lot has been made of the interaction between C and N in terms of climate change and agriculture, with the value of practices that at first were thought critical to agriculture's response being heavily scrutinized. No-till or minimum tillage is one notable example. Cooling benefits of accumulation of soil C by minimum tillage has been called into question, with some evidence suggesting benefits are off-set by increases in the much more potent N_2O; however, the effects are far from certain (Baker et al. 2007; Six et al. 2004; Butterbach-Bahl et al. 2004). Tillage presents an example of tradeoffs in direct field emissions, but tradeoffs among indirect emissions of greenhouse gases may also occur. Draining rice fields mid-season to control methane emissions has been cited as a possible mitigation option (Eagle

2010)[28]. When soils dry out, oxygen diffuses into the soil allowing the soils to go from anaerobic to aerobic, reducing methane. But the transition of soil water content presumably would create conditions conducive to denitrification. Regardless if direct field emissions of N_2O increase, the added machine time necessary to manage the field—draining and reflooding, increased herbicide applications, etc—would increase CO_2 emissions from fuel combustion. Consideration of the entire suit of emissions associated with changes in production is needed to support notions of mitigative technologies.

The agricultural examples illustrate the need to account for emissions of N and C across the entire life cycle of a production system to differentiate among practices. Much has been made of the value of such assessments, with diverse institutions from private companies (e.g., Tropicana Orange Juice) to international organizations such as the FAO (e.g., Livestock's Long Shadow and its follow-up) utilizing them. However, often the comparisons are rife with controversy. Disagreement stems from where the system boundaries are drawn and the underlying assumptions of the life cycle model. Inconsistencies across life cycle assessments lead to comparisons that are as equivalent as apples and oranges. One of the most high profile examples is from the highly controversial report titled, "Livestock's Long Shadow" (Steinfeld et al. 2006). The report states that the radiative forcing of the global livestock industry is greater than the impact from transportation. The report, however, compared emissions from feed to fork for livestock but only the direct emissions from fuel combustion for transportation, and not all the indirect emissions associated with fuel extraction, processing, and distribution. Thus, concerns have been raised about the appropriateness of the appraisal (Mitloehner et al. 2009). For N, Kendall (personal communication) has found little consistency in the methods used to calculate N_2O emission in life cycle assessments. Therefore, we conclude that there is clear value and need to evaluate practices based on life cycle assessment. At the same time, transparent evaluation for further refinement of the methods will add to their value.

[28] Mid-season drainage is less feasible in California because its tendency to delay harvests, increasing risk of crop damage.

Because of the need of full accounting of greenhouse gas emissions, it is important to note that direct field emissions account for only a fraction of total climate forcing from fertilizer use. So called indirect emissions, those that don't occur from within the field of application boundaries, can be quite significant. Prior to the field application, production and transport of fertilizer generates a small amount of N_2O, but large amounts of carbon dioxide because of the energy demand for N fixation via the Haber Bosch process (See Box 5.4). After application, there are many pathways for N loss. When it moves beyond the field, it is still likely to produce N_2O emissions. In some cases , such as riparian environments, probability of emissions increase as conditions become more conducive (saturated soils). Crutzen et al. (2008) suggests that when up- and downstream effects of agriculture are included in the accounting, emissions factors more accurately reflect 3 – 5% of applied fertilizer is given off as N_2O, more than double the amount of direct emissions.

Box 7.3. Toward a unified monitoring strategy for California's N cascade [Navigate back to text]

A comprehensive monitoring network and information system is needed to understand and shape California's N cascade. The primary function would be to provide information in practical and useable formats on the status of N stocks and flows, ecological and human health impacts, and feedback information to assess the efficacy of policy interventions.

Fortunately, California has the makings of a robust monitoring network already in place. Regulatory agencies operate monitoring stations, with the capacity to detect major N compounds and their derivatives. The most well developed monitoring network is for air quality, with more than 100 monitoring sites operated by CARB and the 13 regional air basins catalog ambient ozone, $PM_{2.5}$, and nitrogen dioxide concentrations. Deposition of N compounds (NH_3, NO_x), however, is less well observed. Less than twenty active monitoring stations, sparsely distributed throughout the state, catalog dry and wet deposition of N species through the EPA Clean Air Status and Trends Network (CASTnet) and the National Atmospheric Deposition Program (NADP). In addition, water quality programs, including ones headed by the US Geological Survey, State Water Resources Control Board, Regional Water Quality Control Boards, and Department of Public Health, and concerned citizen groups, monitor NO_3^- concentrations at wellheads, in freshwater streams and lakes, groundwater, and coastal regions. Monitoring activities of the numerous agencies identified provide a sound basis for assessing conditions and change in N species.

Tracking sources of N is more difficult. This is largely because the majority of N emissions are non-point source by nature. Observing both the extent and intensity level of non-point source activities is almost impossible. Fertilizer use is a prime example. Whilst CDFA collects data on fertilizer sales, it provides little reputable information about when, where, and how much N is used, all factors that decidedly determine the impacts on the environment. Even when the necessary information is collected, it may not be made available publically. The Dairy General Order requires producers to report the N applied by field, but the information resides on hard copies within the board's office and is not public record at this time. By contrast to non-point sources, data are widely available on point sources, including emitters like industry (e.g., food processors) and wastewater treatment plants. Access though is still limited; they too languish in disparate locations and difficult to access forms.

Development of a unified, transparent knowledge management system to integrate information from the monitoring networks would be an important step to developing practical and policy response strategies. State and national programs collect information without synthesizing it. That practice is in stark contrast to the multi-source and -impact nature of the N cascade. Development of mechanisms that allow exchange and synthesis of data will underscore targeted multi-media response strategies. With data more easily assessable to decision-makers, new insights on priorities may be possible. Researchers would benefit too. A comprehensive data management system would provide easy access to historical and current public records. When coupled with an assessment of the N impacts, a comprehensive data system facilitates identification of clear research gaps and areas of concern.

Development of a unified strategy that integrates monitoring and data management would foster novel insights and support decision-making when managing the N cascade.

Box 7.4. Metrics for nitrogen management [Navigate back to text]

Our understanding of the current state and changes in the N cascade relies on measurement of N in the environment. N measurements are typically expressed in terms of mass loading (e.g., kg NO_3 per ha) or concentration of a particular form of N (e.g., ppm NO_3). Data collected quantifying these metrics of N can then be translated into management strategies, policy recommendations, and regulations. Smart N metrics capable of documenting the conditions of California's N cascade (at an appropriate scale and reasonable cost) are therefore central to the development of response strategies.

What forms of N are measured and where they are measured can influence the interpretation of the impacts and influence the response options. For example, field-scale mass balance suggests groundwater recharge from only a few cropping systems in California leach a mass of N that would meet the maximum contaminate load standards of a concentration of 10 mg/L NO_3-N (approximately 35 kg N per ha at average recharge rates) that has been set to ensure safe drinking water (Harter and Lund 2012). However, N in groundwater recharge may be attenuated through denitrification or diluted through increased irrigation or precipitation. Changes in N concentration during its transmission to groundwater suggest that where in the soil profile N is measured is important in understanding its actual impacts on drinking water.

Defining metrics and designing measurement and monitoring programs should be tied to impacts of N on the environment and the delivery of ecosystem services. The nature and magnitude of impacts are dependent upon the sources of N, the media (air, soil, or water), and the chemical forms of N. It is important to note that the relationships between sources and impacts are not one-to-one. Only in some cases does the sources of N largely determine its transmission in certain forms into certain media. In many cases, however, a single source contributes to multiple N concerns simultaneously – directly and indirectly. A balance must be struck between concentrating measurements and attention on primary sources versus on the subsequent cascading effects..

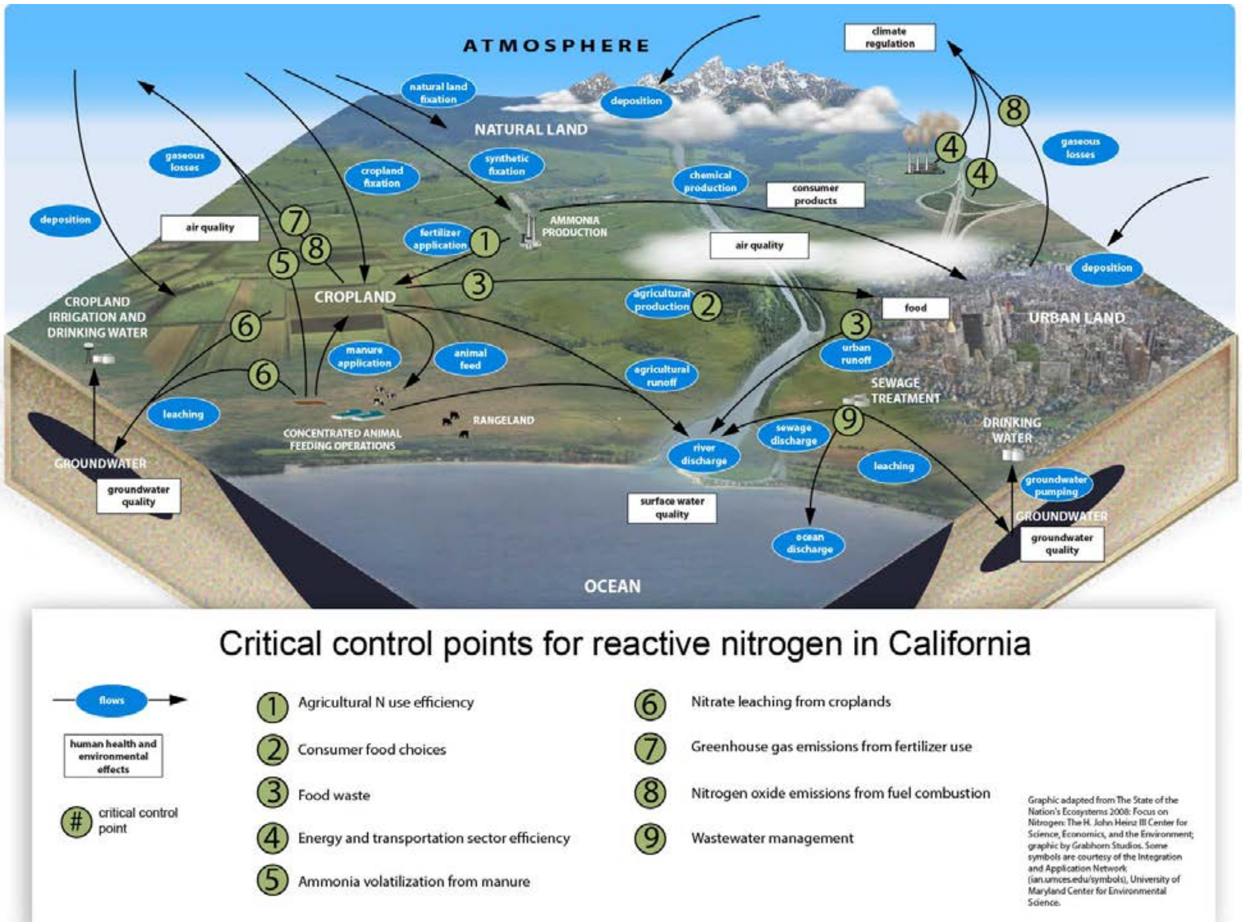
Historically, measurements have informed management and policy to help maintain N impacts below an acceptable threshold of risk. When a contaminant is found to have a direct correlation with environmental or health outcomes, control mechanisms can be put in place to limit the damage. Statewide ozone standards are one example of this approach. CARB and the air basin monitor air quality for ozone concentrations and suggest citizens take precautionary measures when concentrations exceed safe levels. A similar approach – though less frequently – is used as part of the water monitoring programs. Though effective, the concern is that addressing single impacts in isolation ignores the intertwined dynamics of the N cascade. For some cases, a multi-impact management approach may be appropriate in some locations (e.g., Tulare Lake Basin with its poor groundwater quality, high ozone levels, and high N deposition).

Not all metrics address only a single N source or impact (e.g., NO_x concentrations). Collective metrics that aggregate across end points are available for some environmental impacts, with additional ones just coming into use. Perhaps the most well-known collective metric is applied global warming and greenhouse gas emissions. Methane, nitrous oxide, and carbon dioxide emissions can all be expressed in terms of their radiative forcing over a fixed time-frame (100 years) in a common unit, 'carbon dioxide equivalents'. Unifying the metric allows management practices that affect various impact pathways to be compared. Collective metrics are also used to define acidification – e.g., SO_x and NO_x – as H^+ equivalents. Clearly it is possible and potentially advisable to present collective metrics when multiple factors affect a single impact.

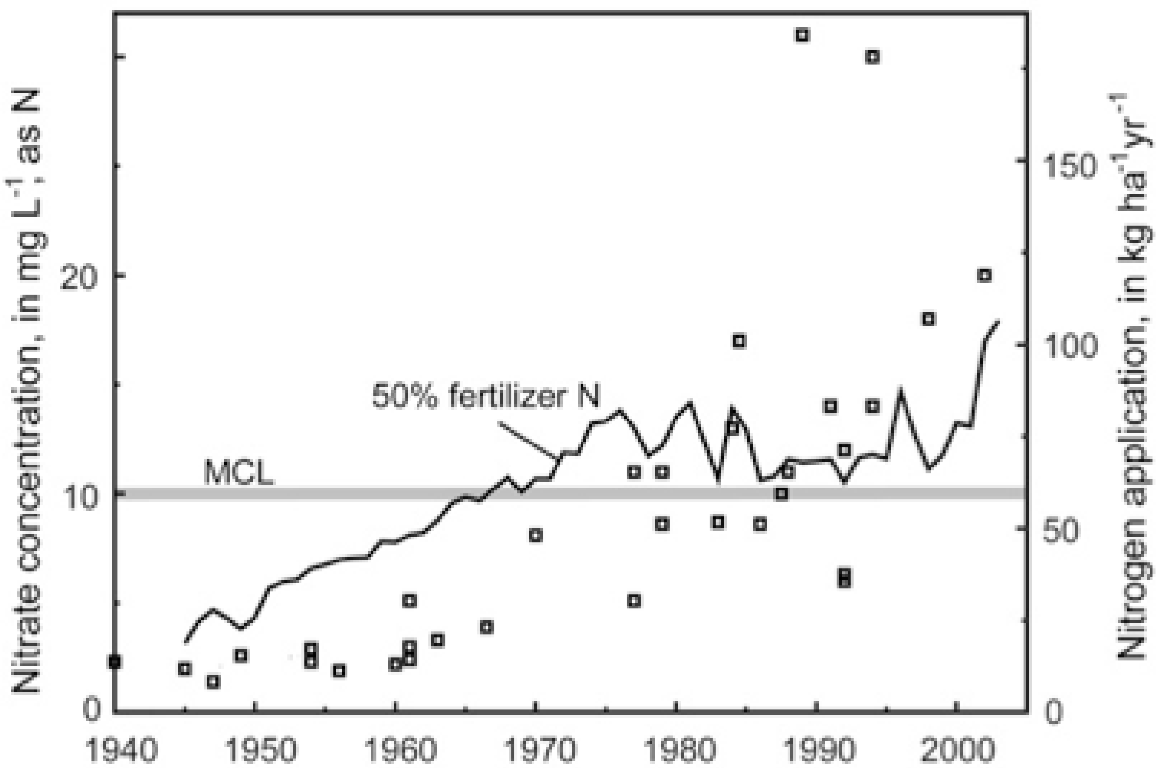
But often, a single source affects multiple impacts in opposite directions, so that tradeoffs exist, for example between food production and climate change. Here, collective metrics may be able to capture the relationships between the impacts. Recently, the global warming intensity (GWi) of cropping systems (yield-scaled global warming potential) has gained traction in agronomic discussions because it scales the emissions by crop yield, acknowledging that some emissions are necessary in highly productive agricultural systems and food production is critical to survival. While the research community has begun to adopt this collective metric; it is yet to be integrated into policy or management approaches. The relatively slow adoption rate illustrates the speed at which a collective metric might be used outside of research. Despite the sluggish transition, GWi presents a good example of the type of innovation that will be needed to address multiple N impacts in a systematic way.

Metrics are fundamental to any N response strategy. California has the infrastructure needed to form the basis of a useful N monitoring program (see Box 7.3). Coupling innovative metrics to the realities of the N cascade is still a challenge. Further, integrating information that can quickly and in near real-time feed back into the management and policy process is the next frontier in addressing N issues in California.

Figure 7.1. Critical control points for reactive nitrogen in California. [Navigate back to text]



2174 **Figure 7.2. Trends in nitrate loading to groundwater from croplands near Fresno, 1940-2005**. Squares
2175 represent concentration of nitrate and groundwater recharge data from wells agricultural areas.
2176 Assuming that 50% of the N fertilizer reached the water table, the solid line represents 50% of N
2177 fertilizer application divided by the area of fertilized cropland. Source: Burow et al. 2008; Burow et al.
2178 2007. [Navigate back to text]



2179

Figure 7.3. Relationship between mass nitrogen leaching (kg ha^{-1}) and nitrogen application rates (kg ha^{-1}). Data compiled by the California Nitrogen Assessment. Outliers of high leaching and N application rates omitted from graph. **[Navigate back to text]**

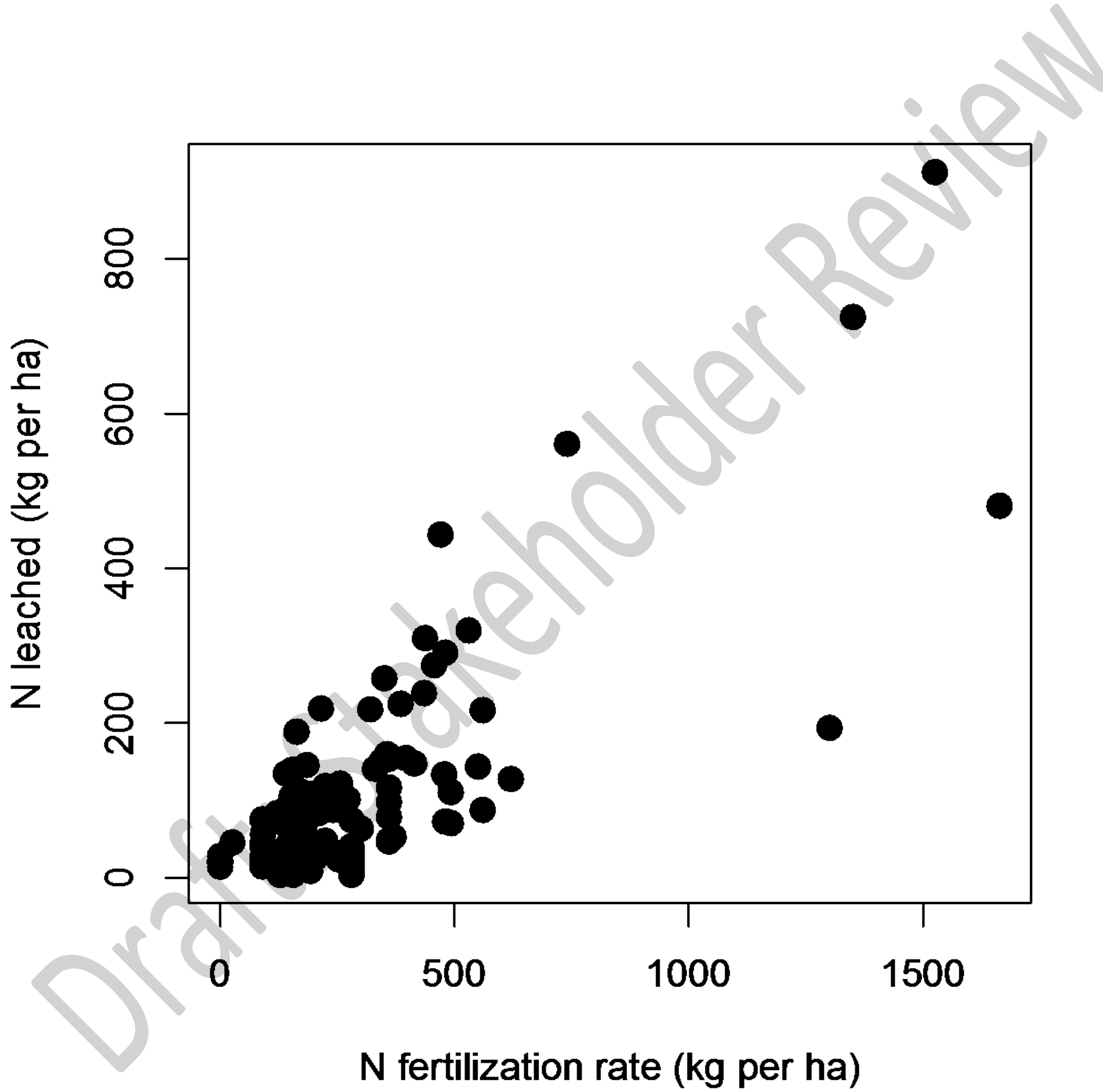
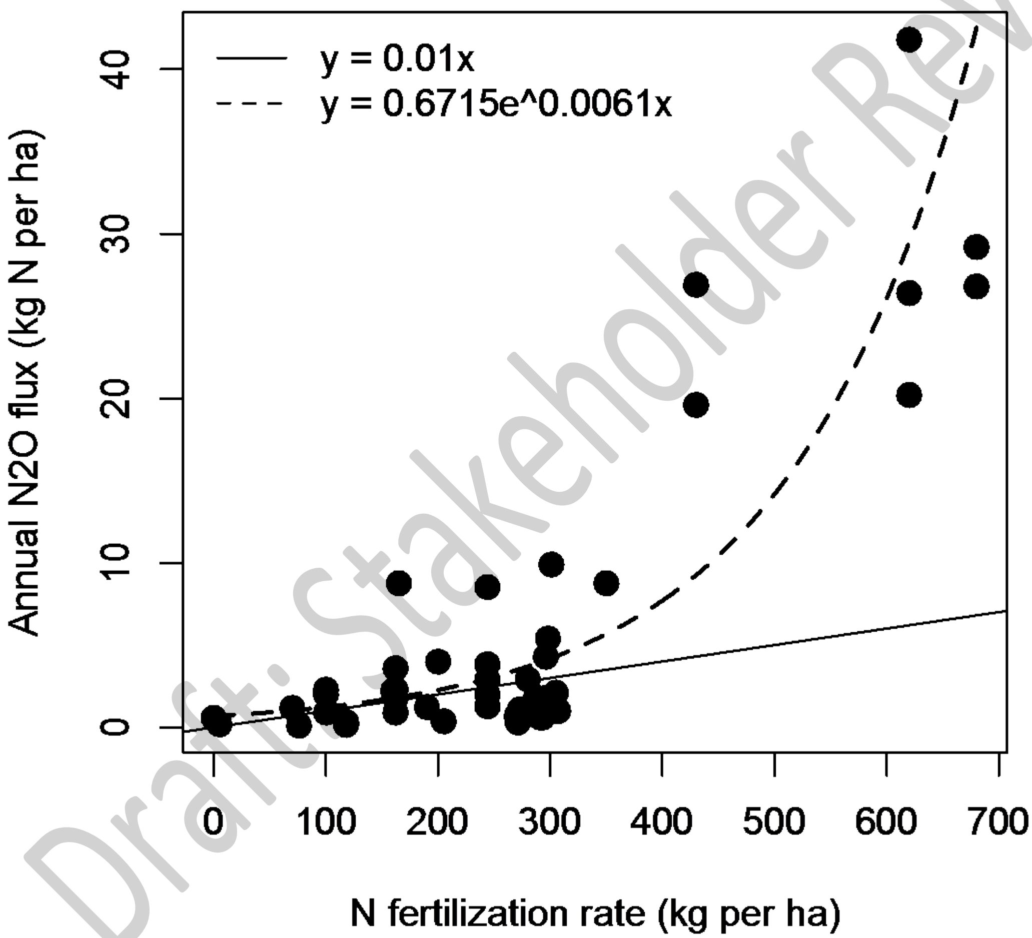
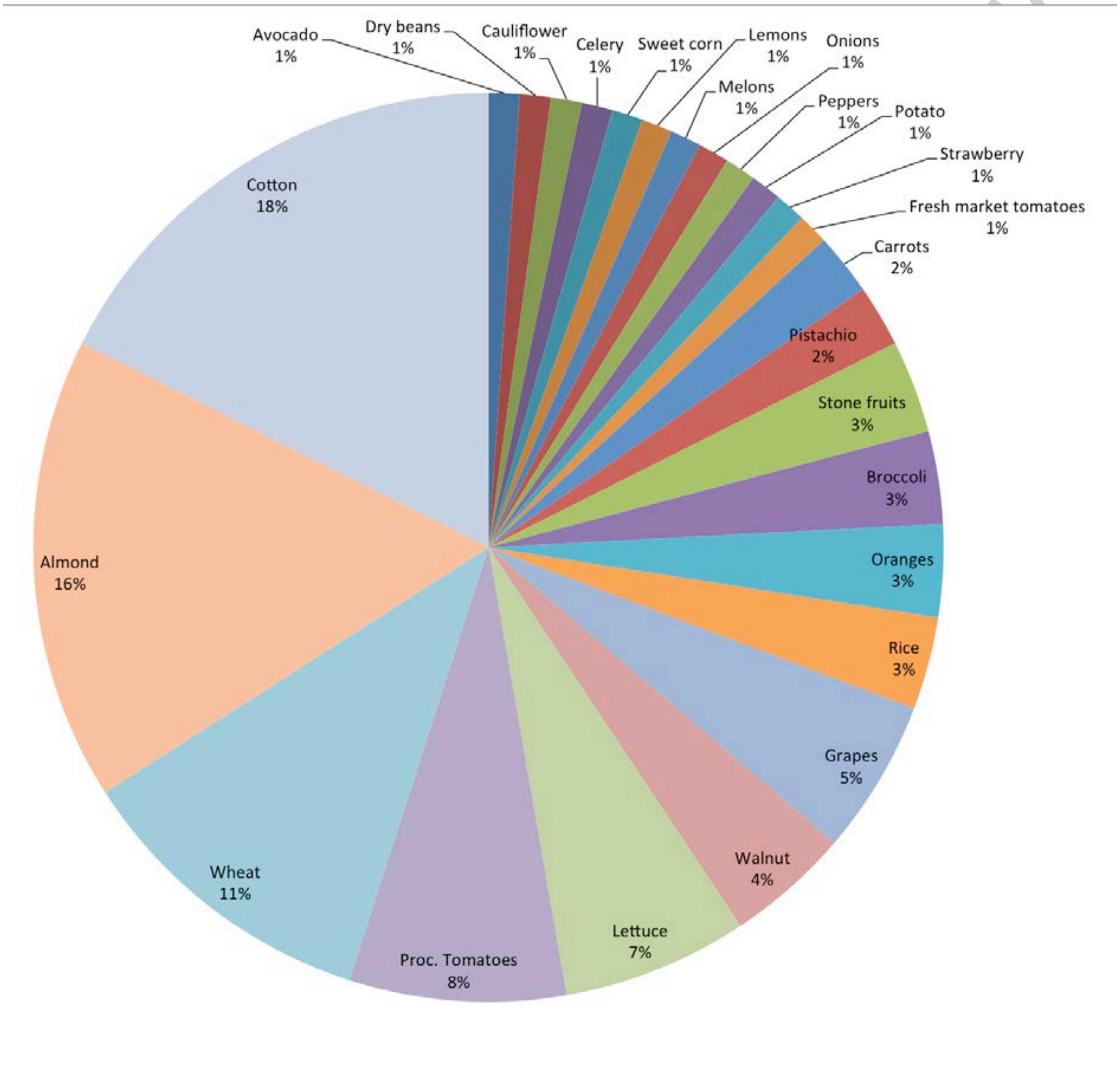


Figure 7.4. Impact of nitrogen application rate on nitrous oxide fluxes from California agricultural soils. Data compiled by the California Nitrogen Assessment and Rosenstock et al. (2012). Calculations account for approximately 76% of annual fertilizer sales. Rice is not included due to the neglible amount of N_2O produced under flooded soil conditions. [Navigate back to text]



2189 **Figure 7.5. Relative contribution of N_2O emissions for 33 crops in California.** Based on California-
2190 specific emissions factor (1.4% of N applied), fertilizer use data developed by the California Nitrogen
2191 Assessment, and USDA Census of Agriculture 2007. The emission factor used for rice is .3% of total N
2192 applied (IPCC 2006). [Navigate back to text]



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2194

Table 7.1. Critical control points for reactive nitrogen in California. [Navigate back to text]

Control points to limit new N inputs
1. Agricultural N use efficiency
2. Consumer food choices
3. Food waste
4. Energy and transportation sector efficiency
Control points to reduce N transfers between systems
5. Ammonia volatilization from manure
6. Nitrate leaching from croplands
7. Greenhouse gas emissions from fertilizer use
8. Nitrogen oxide emissions from fuel combustion
9. Wastewater management

Table 7.2. The mitigative effects of cropland management practices on the fate of N. Source: Literature in Appendix 7A, CNA farm operator discussions, and expert opinion. (*Editorial note: legend continues on next page*) [Navigate back to text]

		Direct Mitigative effects[a]				Confidence[b]			
Cropland management goal	**Yield**	**↑ NH_3**	**↑ N_2O**	**NO_3 ↓**	**NO_3 →**	**Evidence**	**Agreement**	**Applicable system[c]**	**Barriers[d]**
Nutrient management									
Reducing N rate	±	+	+	+	+	***	**	v, tv, e	Δ $\$_i$?
Switching N source	n	+	±	±	±	*	***	all	Δ
Changing N placement and timing	+	+	±	+	+	***	**	Lim.	$i ? r
Water management									
Switching irrigation technology	n		±	±	+	***	***	v, tv, sb	$\$_i$
Increasing soil drainage	+	+	+	-	+	***	***	f	$\$_i$ Δ ?
Soil management									
Conservation tillage	n	-	±	+	+	**	**	f, v	$\$_i$ Δ t
Organic amendments & practices	±	-	±	±	±	**	*		
Diversify crop rotations	n	n	±	+	+	*	**	f, v	$\$_i$
Manage fallow periods	n	-	±	+	+	**	***	f, v	$i $\$_o$
Edge of field	n	n	-	+	+	***	***	f, tv, sb, e	Δ
Agricultural residue	-	+	-	-	+	**	**	f, r	t Δ
Genetic improvement	+	-	±		+	***	***	Lim.	$\$_i$?

[a]Mitigative effects: + = positive effect, - = negative effect, ± = uncertain, n = no effect

[b]Confidence: Relates to the amount of evidence (increasing with more) available to support the relationship between practice and fate of N and the agreement within the scientific literature (* = contrasting results, *** = well established).

[c] Applicable cropping systems: f_r = field crops (receiving manure), f_n = field crops (not receiving manure), r = rice, tv = trees and vines, v = vegetables, sb = small fruit and berries, e = nursery, greenhouse, floriculture, Lim. = limited applicability

[d] Barriers to adoption: t = science and technology, $\$_i$ = cost of implementation, $\$_o$ = opportunity cost, ? = information, Δ = logistics, L = labor, r = regulations

Table 7.3. Estimates of emissions reductions of select alternative fuel vehicles compared to standard vehicles with gasoline internal combustion engines (ICE). Comparisons of CO_2e emissions are based on whole vehicle life cycles, including both manufacture of the vehicle and standard mileage for a lifetime of usage. Comparisons of NO_x emissions are based on annual standard mileage assumptions only, not counting upstream emissions. Hybrid electric vehicles = HEV; plug-in electic vehicles = PHEV; full electic vehicles = EV; fuel-cell vehicles = FCV. [Navigate back to text]

Vehicle type	Pollutant	Grid	% decrease from ICE	Source
HEV	Annual NO_x	CA	41%	Kliesch and Langer 2006
HEV	Life cycle CO_2e	Avg. US	20-25%	Samaras and Meisterling 2007
HEV	Life cycle CO_2e	Low carbon US	30-47%	Samaras and Meisterling 2007
PHEV	Life cycle CO_2e	Avg. US	32%	Samaras and Meisterling 2007
PHEV	Life cycle CO_2e	Low carbon US	51-63%	Samaras and Meisterling 2007
PHEV	Annual NO_x	CA	65%	Kliesch and Langer 2006
EV	Annual NO_x	CA	88%	Kliesch and Langer 2006
EV	Life cycle CO_2e	CA	60%	Lipman and Delucchi 2010
FCV	Life cycle CO_2e	CA	50%	Lipman and Delucchi 2010

Table 7.4. Anticipated effects of dairy manure management technologies. Source: San Joaquin Valley Dairy Manure Technology Feasibility Assessment Panel (2005). See Appendix 7A for detailed discussion of practices. (*Editorial note: table continues on next page*) **[Navigate back to text]**

		Mitigative effects[a]				Confidence[b]			
Animal management goal	**Yield**	**↑ NH_3**	**↑ N_2O or NO_x**	**NO_3 ↓**	**NO_3 →**	**Evidence**	**Agreement**	**Potential system[c]**	**Barriers to adoption[d]**
			Feed management						
Precision feeding	+	+	+	+	+	**	***	d, b, p	Δ $\$_i$?
Supplements & hormones	+	+	+	+	+	**	***	d, b, p	r
			Manure storage and treatment						
Frequent manure collection		+	±	+	+	*	***	d, b, p	$\$_i$
Solid-liquid separation		+	+			***	***	d	$\$_i$ Δ ?
Composting manure solids			>			**	*	d, b, p	$\$_i$ Δ L
Biological additives for wastewater		±	±						$\$_{I,}$ t
Anaerobic digestion of wastewater		+	±			**	***	d	$\$_{I,}$ r
Storage cover for wastewater ponds		+				*	***	d	$\$_i$
			Land application of manure						

Measured applications & flow meters	±	±	+	+	**	***	d	$_i$
Split applications	±	±	+	+	**	**	d	$_i Δ
Incorporation below surface	+	+	-	+	***	***	d, b, p	?
Species improvement								
Genetic improvement +					***	***	p	$_i t ?

[a]Mitigative effects: + = positive effect, - = negative effect, > = minimal impact, ± = uncertain, n= no effect

[c]Potential systems: d = confined dairy, b = beef feedlot, p = poultry, c = grazing cattle

[d] Barriers to adoption: t = science and technology, $_i = cost of implementation, $_o = opportunity cost, ? = information, Δ = logistics, L = labor, r = regulations

2210 **Table 7.5. Removal efficiencies (%) for select primary and secondary technologies**. Sources: US EPA
2211 1999, World Bank 1998. **[Navigate back to text]**

	Fuel		
NO_x reduction technology	**Coal**	**Oil**	**Gas**
Combustion control			
Low-excess air	10-30	10-30	10-30
Staged combustion	20-50	20-50	20-50
Flue gas recirculation		20-50	20-50
Water/steam injection		10-50	
Low-NO_x burners	30-40	30-40	30-40
Postcombustion treatment			
Selective catalytic reduction	60-90	60-90	60-90
Selective noncatalytic reduction		30-70	30-70

2212