Chapter 3: Direct drivers of California’s nitrogen cycle

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Contents

Boxes

Figures

Tables

Appendix

Main messages

Synthetic nitrogen (N) fertilizer sales (and presumably use) in California have increased dramatically since World War II and risen by at least 40% since 1970 but consumption has leveled off in the past 20 years. Croplands receive an estimated 90% of the N fertilizer sold today; the remaining 10% is used to fertilize urban lawns, gardens, and recreational areas. Nitrogen fertilizer application rates (kg per ha) have increased an average of 25% between 1973 and 2005. Data show the majority of California crops recover well below the global average of 50% of applied N with some crops assimilating less than 30%. Data describing the distribution and use of organic N sources (e.g., manures, composts, and leguminous cover crops) are limited but indirect indicators of supply and demand suggest growing use.

Until recently, manure management decisions were made without much regard to their N consequences. The breath of techniques used, the limited information available, and large variability among operations, especially for San Joaquin Valley dairies, makes any conclusion about changes in manure management practices tentative. Land application of manure represents an important recycling of N to the production/food system. Efficient utilization of manure N is complex because it is mostly in the organic fraction. Because as much as 50% of dairy manure and 100% of poultry and beef feedlot manure is handled as a solid, transported offsite and their application are not tracked; the fate of this large pool of N is speculative.

In spite of sizeable increases in the amount of fuel combustion activities that emit nitrogen oxides (NO_x) into the atmosphere, emissions have declined steadily since 1980. Over the past 30 years, the number of stationary NO_x sources went up almost three-fold. Meanwhile, the small vehicle population increased 155% between 1980 and 2005. Total vehicle distance traveled increased 237% over the same time period. Mobile sources continue to be the dominate NO_x source producing 86% of total emissions statewide in 2008, but the relative importance of sources has shifted from the small vehicle fleet to off- and on-road diesel engines. Emissions have been controlled by aggressive technology forcing regulation. Under certain conditions, implementation of advanced NOx control technologies cause inadvertent releases of ammonia.

About 77% of food N will enter wastewater collection systems and about 50% of wastewater is dispersed in the environment without specific treatment for N removal. This includes wastewater treatment plants with limited nitrification, leakage from sewers, and many wastewater infiltration systems. Attempts to control N pollution have led to a steady increase in the level of treatment practiced at municipal wastewater facilities throughout California. In 2008, nearly 50% of wastewater treatment facilities reported performing at least advanced secondary treatment and 20% performed tertiary treatment processes. At last estimate, onsite wastewater systems are used to treat more than 3.5 million people's wastewater and approximately 12,000 new units are installed each year.

Alfalfa yields have increased an average of 53% per ha between 1950 and 2009 statewide suggesting biological nitrogen fixation has become a progressively important input of N to California's land surface. Because N fixation rates are proportionate to productivity, greater production suggests biological nitrogen fixation has become a progressively significant source of N to California's terrestrial biosphere. Data are inadequate to accurately estimate changes in the extent or productivity of other leguminous crop species planted on agricultural lands or species capable of biological nitrogen fixation in natural lands. Biological nitrogen fixation may be offset by higher rates of atmospheric N deposition and preferential uptake of soil N by plants.

Wildfires cause locally acute N air pollution and potentially release soil N reserves into waterways. The area burned by wildfires has increased markedly since 1990, with the top years with the largest area burned in California's history occurring in the last decade (2003, 2007, and 2008). However, there is high annual variation in the extent of wildfire each year evident by a 44-fold difference between the years with the least and greatest (12,400 versus 548,000 ha) area burned since 1970. Greater fuel loads resulting from drought, fire suppression, and invasive species threaten to increase wildfire intensity, a key determinant of N emissions.

Changes in land cover and land use fundamentally alter N cycling in ways only recently becoming appreciated. Land use change can result from a shift in land cover or simply a change in the intensity of use; both have occurred in California. Urban areas grew 37.5% between 1972 and 2000 and now cover 4.2% of total land base. Urbanization has caused agriculture to relocate from some regions. The net effect of urbanization and agricultural relocation/expansion has been a 1% decrease in total agricultural land over the same time frame. This shift in land cover has been accompanied with an intensification of use. Urban intensification has led to higher density urban areas throughout the state, but the rate of change has been variable. In croplands, the mix of crops produced has changed from relatively N extensive to N intensive species. Field crops were still grown on 53% of cropland in 2007 (largely because of the land area dedicated to alfalfa) but this is a significant decrease from 74% in 1970. Simultaneously, the dairy cow population has doubled and the broiler population has tripled in conjunction with higher flock/herd size concentrating N rich feed in California and amplifying manure N handling concerns.

3.0 Introduction: Controls of California's N cycle

This chapter reviews the direct drivers of nitrogen (N) cycling in California. A direct driver is a human action or natural process that "unequivocally influences ecosystems processes" (MA 2005, Ash et al. 2010). The objective of the chapter is to introduce the ways by which direct drivers modify the N cycle and describe historical trends in these activities. Specific attempts are made to highlight key lessons that have hador will change the trajectory of a driver's impact. The chapter does not discuss either the underlying context shaping human actions and natural processes or the relative magnitudes of N flows that result from these activities. Those topics are covered in the preceding chapter (*Underlying Drivers of California's Nitrogen Cycle*) and the following (*California's Mass Balance*), respectively. The eight most important direct drivers of N cycling in California are: (1) fertilizer use and soil management; (2) feed and manure management; (3) fuel combustion; (4) waste management; (5) biological nitrogen fixation; (6) ammonia synthesis for industrial processes; (7) wildfire; and (8) land use change.

3.1 Fertilizer use and soil management

Nitrogen fertilizer use is a direct disturbance of the N cycle. Application of N fertilizer stimulates soil microbial activity and provides a nutrient subsidy to enhance plant productivity. Managed well, plants capture a sizeable fraction of the fertilizer. Often, however, N fertilizer is applied well in excess of plant uptake. Under such circumstances, the biochemical properties of N and common fertility and soil management practices inevitably cause N to be released into the air, soil, and water (Galloway et al. 2003, Harter et al. 2006, Stehfest and Bouwman 2006).

3.1.1 Inorganic N fertilizer use on farms and lawns

Sales of inorganic N fertilizer have increased 12-fold since the materials were introduced after World War II. Inorganic N fertilizers, also known as synthetic or mineral fertilizers, were first derived from Chilean nitrate deposits. However, the invention of the Haber-Bosch process in 1908 radically changed the availability of inorganic N (Erisman et al. 2008). After the Second World War, demand for explosives—another product derived from the Haber-Bosch process and the root motivation for its development—declined, and a rapid increase in the production and distribution of synthetic fertilizer ensued. The consequence has been a large increase in the use of synthetic N fertilizer in the developed world (Galloway et al. 2008). In California, N fertilizer sales (and presumably use) have grown at an average annual rate of 5% between 1946 and 2009 (Figure 3.1). Annual sales grew at their fastest pace prior to 1980. Since that time, sales of N fertilizers have leveled-off. Recent annual sales of more than 600,000 Mg of N fertilizers are not distinctly higher than sales were in 1980.

[Insert: Fig. 3.1]

Statewide sales data only provide a partial picture of N fertilizer use in the state. Farm operators and urban land managers make fertilizer decisions at the field- and household-level subject to local conditions and constraints. It is the fertility decision for a particular parcel of land that determines the intensity, effectiveness, and outcomes of N use. Robust knowledge of synthetic N use at this level is thus paramount to understanding the full picture of N fertilizer use in California, however, in general, insufficient data limit understanding about the use of inorganic N fertilizers at such scales (see Supplemental Data Tables).

The California Nitrogen Assessment supported an effort to document changes in N application rates for 33 important California crops in 1973 and 2005 (Rosenstock et al. in review). Average N use per ha across the 33 crops surveyed increased 25% over this 33-year period (Appendix 3.1). The magnitude and direction of change was crop specific. Application rates for a few crops increased by more than 75%. Yet, for 10 of the 33 crops examined the average rate at which N fertilizers were applied declined. Nitrogen fertilizer use on vegetables and nut crops showed the largest increases. This is particularly important because the area dedicated to these crops increased simultaneously with higher N application and many of these crops recover far less than 50% of N applied (see sections 3.9.2 and 3.2.3, respectively). The consequence has been both a greater amount of N applied per unit of area as well as greater potential loading to the environment. In contrast, N application rates for stone fruits and subtropical fruits generally decreased. Of the 33 crops, only four crops—cotton, almond, rice, and wheat-accounted for 51% of the total N applied. This finding suggests that a relatively small number of cropping systems have a disproportionate affect on N use in California croplands. It is worth noting that the research did not estimate N application rates in the nursery or greenhouse industries because data are unavailable to represent the diversity of species grown. Ornamental horticulture production systems tend to have among the highest application rates; 100 – 7,000 kg per ha (Evans 2007).

Not all the inorganic N fertilizer sold in California is applied to agricultural crops. Property owners and public space managers regularly apply inorganic N to grow and maintain urban green space. In many parts of California, lawns are more widely distributed than agriculture (e.g., southern Coastal California) and thus, urban uses dominate fertilizer inputs (Townsend-Small et al. 2011). However, neither the extent of urban green space nor the intensity of N use in urban areas is well documented in California. Estimates of lawn coverage range between 271,770 and 1.1 million ha, a more than 300% difference between the minimum and maximum (Templeton et al. 2000, Milesi et al. 2005, Green 2007, Wu et al. 2010). The median area (687,500 ha) is 56% larger than the land area used for alfalfa (approximately 440,000 ha in 2008) suggesting lawns are the most widely cultivated commodity in the state. Fertilization rates on lawns are uncertain. Surveys conducted in other parts of the US indicate that the average household lawn receives an average of 100 kg per ha (Law et al. 2004, Osmand and Hardy 2004) and a recent study suggests similar rates may be common in southern California recreational areas (Townsend-Small et al. 2011). Parks, fields, and golf courses, however, typically receive greater amounts of N fertilizer than household lawns. Application rates on golf courses in southern California have been reported to exceed 400 kg per ha (Wu et al. 2007). Industry data suggests turf receives an average of 50 kg N per ha when accounting for the wide range of N use (Scott’s Fertilizer Company, personal communication).

Due to its extensive coverage, the cumulative impact of urban fertilizer use is potentially large. For example, assuming the coverage of lawns is equal to the median estimated area (687,500 ha) and it is fertilized at an average rate, 50 kg per ha (Scott's Fertilizer Company), approximately 34,375 Mg of N would be applied to urban landscapes in California each year. This estimate may even be conservative; calculations made by Liptzin et al. (2011, Chapter 4) suggest 10% of fertilizer sold in the state is applied in urban areas. Even with the more modest estimate, the total N applied to lawns is greater than that applied to some agricultural species (e.g., carrot) demonstrating the relative importance of this land use to N dynamics in the state.

3.1.2 Organic N use on croplands

Crop producers, at times, apply organic N in lieu of or in addition to inorganic N fertilizers. Commonly used organic N materials include manures, composts, waste products, and leguminous plant species (Hartz et al. 1996, Hartz and Johnstone 2006, Gaskell and Smith 2007, Hartz and Bottoms 2010). Organic N materials, for the most part, represent a transfer of N from a different land use. Cows do not produce N, it simply passes through them and is converted from feed into manure. Compost is a collection of N from different waste products (e.g., food waste, manure, and urban green waste). Leguminous plants are the only exception (section 3.5, biological N fixation). Utilization and availability of organic N sources largely depends on its production through tangential activities. Since organic N supplies carbon (C) and N, it is generally thought that organic N sources provide co- benefits supporting soil health (Reganold et al. 2010). Some evidence suggests that organic N sources present a lower pollution potential and are more N benign than inorganic fertilizer (Drinkwater et al. 1998, Poudel et al. 2002). Under some conditions this may indeed be true, but research has shown that, like inorganic N, organic N can be a source of reactive N to the environment (Barton et al. 2001, Kirshmann and Bergstrom 2001, Harter et al. 2006, Heinrich 2009, van der Schans et al. 2009, section 8.1).

Control of organic N applied to croplands is more complex than inorganic N. Crop species utilize inorganic N forms, NH_4 and NO_3. The vast majority of N in organic materials is in the organic form and must mineralize to become plant available. The rate at which mineralization occurs depends on the characteristics of materials (e.g., how it was produced and N content), environmental conditions (e.g., temperature and water), and microbial activity (Hartz et al. 2000). Despite intensive study over many decades, the ability to predict the mineralization rate has proven elusive, especially in commercial production environments (Crohn 2006). Inability to forecast N supply and time N releases with plant demand make managing fertility challenging and can create pollution concerns (Pang and Letey 1997).

The extent that organic N materials are applied as a primary fertilizing material or as a soil amendment (e.g., in lettuce production, Smith et al. 2009) is largely unknown. A survey conducted by Dillon et al. (1999) suggests that their use is common. More than 20% of the 800-some farmers surveyed applied composts or manures in 1986. In the subsequent 10 years, the use of these N sources became 24% more prevalent. When only considering producers that shifted production to new crops, the percentage reporting they applied organic N materials in 1996 vs. 1986 rose to 55% of respondents.

Indirect indicators support the conclusion that organic N is increasingly demanded and available in California. The N fertilizer used by certified organic farms invariably comes from such sources (Smukler et al. 2008, Reganold et al. 2010) and the land dedicated to these systems has grown rapidly in recent years. Between 2000 and 2005, the area of certified organic farms in California increased 31% from 59,421 ha to 77,963 ha (Klonsky and Richter 2007). The most recent USDA Organic Agricultural Census reports that more than 110,000 ha were in certified organic production in 2008, suggesting nearly a doubling in the 8 years between 2000 and 2008 (Klonsky and Richter 2007, USDA 2010). According to the Organic Census, 58% of certified organic farms produced or applied organic compost and 49% applied green or animal manures in 2008 (USDA 2010). Furthermore, the increase in animal and human population has resulted in a greater availability of N-rich manures, composts, and urban wastes destined for land application than ever seen before.

Little information is available to understand where and how organic N sources are used. The two exceptions are for applications of liquid manure associated with dairy production in the San Joaquin Valley and the application of biosolids. In both cases, the State Water Quality Control Board (SWRCB) requires documentation of organic N distribution for regulatory compliance to minimize water quality concerns. By comparison, the distribution and application of solid manure are not tracked. As much as 50% of the dairy manure and 100% of the poultry and beef feedlot manure are exported and applied to land offsite. The fate of embodied N will be determined by the cropping and application practices. A lack of data makes it difficult to quantify this significant transfer of N from production systems to croplands.

3.1.3 Agronomic nitrogen use efficiency (NUE)

Current N application rates use more N than plants require and leave considerable amounts of N in the soil after harvest. Only a small fraction of this surplus is used in the subsequent growing seasons. Inevitably, whether from inorganic or organic sources, it leaks from croplands and escapes into the environment. Both nitrate (NO_3) leaching potential and the rates of nitrous oxide (N_2O) emissions increase nonlinearly with greater surplus N (Broadbent and Rauschkolb 1977, Van Groenigen et al. 2010). Because surplus N is an over application on the part of the grower, it represents an economic loss for the producer. Maximizing N recovery and minimizing surplus presents win-win conditions critical to increasing economic returns and reducing N loading to the environment.

California cropping systems recover only a fraction of the N applied. Globally, it is widely established that the efficiency of fertilizer N applications averages 50% in the first growing season (Tilman et al. 2002) and less than 5% in the second (Fritschi et al. 2005, Ladha et al. 2005). Global research on N use efficiency (NUE), however, is conducted under conditions not characteristic of California cropping systems (e.g., rainfed cereal crops). The California Nitrogen Assessment compiled a database of previous NUE research for California (Table 3.1). Results indicate that fertilizer recovery in California is regularly lower than the global average. Nitrogen use efficiency was below 50% for 69, 67, and 48% of the crops for which data were available depending on the method of measurement (isotopic ^{15}N, zero-N, or PNB, respectively). Lower than average NUE in California is not altogether unexpected. Large annual variation in growing conditions causes plant N demand to vary by as much as 50%. Conventional wisdom suggests many producers apply extra N fertilizer as “insurance” to buffer themselves against the economic risk of unfulfilled yield potential. The strategy is cost effective because fertilizer is less than 5% of operating costs in high-value cropping systems. Lower value field crops generally had among the highest NUE.

[Insert: Table 3.1]

Limited evidence suggests California cropping systems are becoming more technically N efficient, but croplands still pose a pollution risk. Partial nutrient balances (PNB), a ratio of N harvested to N applied, for 33 crops in 1973 and 2005 show an average increase of PNB by 37% over this time frame by comparison to 25% increase in N application rates (Table 3.1 for some of the 2005 results; Rosenstock et al. in review). Similar to N application rates, the trend depends on the crop in question. An area-weighted PNB (which weights the calculation for area under production for each crop) indicates an amount equivalent to 53% of the N applied statewide could be accounted for in crop products and byproducts exported from the field. Assuming the PNB values are representative of California cropland as a whole, this statewide PNB suggests there was a surplus of almost 300,000 Mg of N sold (and presumably applied) in 2008.

N leakage results in N loading to soils, air, and water. Due to regional differences in N use associated with land cover, crop mix, hydrology, and climate, large regional differences in N loading to the environment have occurred as a result of N fertilizer use over time. The clustered spatial distribution of California crop production and settlements would suggest that counties, regions, and watersheds that contain current or previous high densities of N-intensive crops (e.g., Salinas Valley) or urban areas (e.g., Los Angeles) receive the largest inputs from fertilizer N. An analysis characterizing the N loading of water- and air-sheds based on crop production and urban horticulture is needed to fully understand the spatial trends in N fertilizer use, efficiency, and loading.

3.1.4 Soil management in croplands

Every aspect of crop production affects NUE and N cycling. Virtually every decision an operator makes modifies the complex biological and chemical relationships governing plant-soil-atmosphere interactions. For example, the application of carbon rich residues immobilizes N in soils (Bird et al. 2001) or fertilizer placed in closer proximity to plant roots increases uptake (Miller et al. 1981). Very limited information exists to evaluate the current state of, or historical changes in, the entire suite of soil management practices. Trends for major production decisions (Table 3.2), besides fertilizer use, demonstrate the constant flux of nutrient and soil management practices.

[Insert: Table 3.2, Box 3.1]

3.2 Feed and manure management

Animal production is both a N sink and a N source in California. Animals require dietary N and amino acids (building blocks of proteins containing N) for maintenance, growth, and production. Animal physiology limits the conversion of feed N to animal mass resulting in much of the ingested N being excreted in manure (Kebreab et al. 2001, Powell et al. 2010). Up to a biological threshold, the amount and form of manure N can be altered by dietary manipulation. After excretion, manure handling practices determine how much of manure N is conserved and able to be recycled as a fertilizer versus released into the environment.

3.2.1 Diets and nitrogen utilization efficiency

Improvement in analytical techniques and investment in research has allowed formulation of diets to meet animal nutritional needs of crude protein, rumen degradable/nondegradable protein, or specific limiting amino acids (Morrison 1945, NRC 1994, 2001). Diets can be formulated to meet minimum and/or maximum protein and/or amino acid requirements. Since the general objective in formulating diets is to provide the necessary nutrition for the least cost, the minimum constraint is typically used; protein ingredients are usually more expensive to feed. The possible exception to this rule is with the use of inexpensive by-product feeds. By-product feeds, such as distiller's grains, almond hulls, or carrot tops, may or may not increase dietary concentrations of proteins or minerals depending on the use of maximum constraints when formulating diets. When diets are more closely formulated for protein or amino acid requirements, N is used more efficiently (a higher percent of the consumed N is incorporated into animal product).

Partial efficiencies of N use can be calculated during each stage of production as the ratio of N converted to animal product and/or retained to N consumed by the animal (ASAE 2003). Careful attention must be directed to the unit of time involved for each category of animal. For turkeys and broilers, total N use efficiency is equivalent to partial N use efficiency. For all other production animals (i.e., beef, dairy, swine, layers) total N use efficiencies can be calculated over the life of the animal as the sum of lifetime N retained and/or converted to animal product divided by total lifetime N consumed. Partial efficiencies range from 15 to 64% depending on the species and production category (Table 3.4). Average partial efficiency of N conversion to animal product is 14.9% for feedlot steers during the 153-day feeding period, 24.4% for high producing dairy cattle, 63.7% for milk fed calves, 34.0% for grow-finish pigs, and 35.4% for layers. Efficiencies for broilers are near 60%. Ingested N not converted to animal product or used for growth is excreted (Nahm et al. 2002, Hristov et al. 2011).

[Insert: Table 3.4]

Diet has a profound impact on N excretion and loss. An animal's diet determines manure characteristics (e.g., form of N and moisture content), which in turn determine the probability for certain N transformations. Urea and uric acid formation and excretion increases with increased consumption of dietary N, especially when animals consume N above recommended nutritional levels. Urea N voided by cattle and uric acid voided by birds may be quickly hydrolyzed to NH_3 when urease and microbes are present increasing the risk of NH_3 volatilization (VandeHaar and St Pierre et al. 2006, Xin et al. 2011). If physical conditions are favorable, the process from excretion to volatilization takes place rapidly, in a time span ranging from a couple of hours to a couple of days. Decomposition of organic N excreted from cattle occurs at slower rates than hydrolysis of urea and these slower rates of transformation increase the feasibility of manure collection and N conservation within the animal production facility. However, management of organic N is more difficult than urea and NH_3 when applied to land (section 3.1.2). A management conflict, thus, arises between the ability to conserve N within the animal production unit and planning for its end use as a fertilizing material on croplands.

3.2.2 Manure management within a confined animal feeding operation

Manure N is a resource and a potential pollution concern. Within the animal production unit, the goal of manure management from the rancher's point of view is to maintain a clean environment for the animal, reduce nuisance from odors, and improve animal health. From an environmental standpoint, manure management must also conserve the N embodied in the manure until it can be applied to cropland. There are many pathways through which N may be lost in animal housing and manure storage/treatment facilities, and some emissions are inevitable. The primary pathway of loss is volatile emissions of NH_3 into the atmosphere. It is estimated that between 20 and 40% of the N excreted on dairies in the San Joaquin Valley (CoC 2005) and 4 to 70% in poultry houses worldwide (Rotz 2004) is emitted as NH_3. Leaching of NO_3 to groundwater may also be a concern under corrals (Adriano et al. 1971a). Because emissions occur from various components of the animal production unit, N needs to be managed throughout the entire process. It is meaningless to consider management of one practice without placing it within context of the entire transfer from animal to the field. Conservation of N in one management area does not guarantee conversation throughout the system.

Manure management practices are diverse and constrained by the design of the facility. Differences between freestall and open lot dairies in the Central Valley are a good example (Figure 3.2). Manure deposited in freestall barns is collected by flushing water over the concrete surfaces transferring it to a pond (lagoon) to be stored/treated as wastewater. Collection of manure in liquid form can help minimize emissions from housing, but economic considerations limit the distance it can be transported for land application (Coc 2005). In contrast, manure in open lot dairies is deposited on the soil surface where it dries. While manure resides in place, open lots are sources of NH_3 (Cassel et al. 2005). Lots are scraped and manure removed at specified intervals, typically two to four times per year. After collection, solid manure is stacked and stored prior to use (land application or exported offsite). That example illustrates that N flows and critical control points depend on the structure and operation of the facility. Modifications of manure management processes can only be made within context unless wholesale shifts to new facility designs are adopted. Because of inherent infrastructure of the operations, transformative changes are often cost prohibitive.

Until recently, manure management decisions on many California dairies were made independent of N conservation or utilization. Implementation of manure handling practices significantly change N dynamics and, therefore, it is imperative to understand unintended consequences of changes in practices. Three surveys documenting manure management practices in 1988, 1997, and 2004 have been published, but differences in the geographic extent and questions asked among the surveys make comparisons tenuous (? et al. 1988, Meyer et al. 1997, SAREP 2004). Nevertheless, it appears dairy operators are adopting practices that increase ranchers' ability to manage N (Table 3.5). For example, between 1988 and 2002, the percentage of respondents that used settling basins to separate solids from liquids doubled to 66% and those that composted solid manure rose from 6 to 21% statewide. These two manure treatment options provide greater control over manure N by isolating more homogenous manure components and stabilizing N into organic matter, respectively (Panel 2005). As mentioned above, these trends represent only a single component of a complex interdependent system. Many nuances of manure management that alter N dynamics on a dairy facility are not covered in the surveys (e.g., frequency of collection). Both the lack of information and the diversity of manure handling practices limit the ability to evaluate the status of manure N management.

[Insert: Table 3.5]

Even less information is available to evaluate changes in poultry manure handling practices. In contrast to the highly variable dairy management systems, manure management in poultry operations is considered to be more uniform throughout the industry. The common factor in confined poultry production facilities is birds are raised indoors and under roof structures. This minimizes contamination of manure with rainwater and maintains a solid product that is manageable and transportable. The frequency of manure removal can range from once weekly to only twice yearly for California layer production systems (Hinkle and Hickle 1999, Mullens et al. 2001), while manure is generally removed between flocks for broiler and turkey production. Dried material is then sold for animal feed, as a soil amendment, or transported to commercial processing plants for pelletization or composting. Manure characteristics (e.g., moisture content), environmental conditions (e.g., temperature and wind speed), and drying method (e.g., depth of stack) will alter NH_3 emissions in the house and during processing (Xin et al. 2011). Like that of dairy systems, the future of California poultry manure management practices is uncertain. Implementation of newly defined housing systems (Proposition 2) may change manure handling practices and subsequent N dynamics on ranches.

Manure management practices traditionally are in a state of transition as managers seek to improve management to reduce nuisance and comply with environmental regulations. Regulations have caused operators to evaluate and modify practices, which has undoubtedly changed N dynamics, although for the most part inadvertently. With the current regulatory trajectory, many facilities will be faced with adopting new manure management techniques.

3.2.3 Land application of manure

Nitrogen excreted from animals is regularly recycled back into feed and food production systems. On confined dairy systems in the San Joaquin Valley, liquid manure is surface applied to feed crops close to the production unit. Poultry and beef producers do not regularly produce feed for their animals and thus the resource is transferred onto croplands away from the facility. Most often, manure solids are applied to croplands bound for human consumption. Effectively utilizing N in manures (organic N) is a complex task. Land applications of manure are discussed along with other organic N materials in section 3.1.2.

3.2.4 Manure management for grazing animals

Grazing lands can be sources of NH_3 and N_2O to the air and NO_3^- to nearby waterways and groundwater. Manure excreted from grazing animals is not collected or stored. Urine and feces are deposited on the pasture and depending on microbial activity, hoof action, soil type, plant species composition, topography and climate may be incorporated into plant roots, adsorbed to soil particles, lost atmospherically, leached, or managed in runoff (Oenema et al. 2008). Since the manure itself is not managed, pasture management becomes critical. Grazing patterns, stocking density, and pasture productivity will determine the ability for the environment to buffer manure N deposition. Manure management on California grazing lands is important for pasture-based dairies, beef cow calf operations and where free range poultry graze.

3.3 Fuel combustion

Fuel combustion releases reactive N gases into the atmosphere. Nitrogen oxides (NO_x) are produced when N and oxygen react at high temperatures, with conditions conducive to NO_x formation typically occurring during energy production, industrial, and transportation activities. Combustion of some fossil fuels, in particular oil and coal, produce additional “fuel NO_x ” because of trace amounts of N in the fuel itself. Fuel NO_x is less common in California because the fuel mix used contains less N. Technologies used to control NO_x emissions sometimes inadvertently release ammonia (NH_3). For example, three-way catalytic converters installed on passenger vehicles can reduce NO_x to NH_3 instead of the environmentally benign N_2 when the air:fuel ratio is high, a common occurrence during acceleration (Keane et al. 2000, Baum et al. 2001). Ammonia is also used as a reagent to control NO_x emissions from stationary sources, specifically with selective catalytic reduction (SCR) technology. If the SCR system is not optimized (e.g., too much ammonia in the gas stream, temperature is too low, or the catalyst has aged), NH_3 is released directly with flue gas without completing its intended reaction.

3.3.1 Growth and technological change of combustion activities

It is well established that there has been enormous growth in combustion activities in California. In the 20 years between 1987 and 2007, the number of stationary sources producing NO_x, not including dry cleaners and gas stations, increased from 3,391 to 9,311, a 275% increase (CARB 2010). Mobile source activity shows similar sizeable growth rates. The small vehicle population in California increased 155% to over 36.8 million vehicles between 1980 and 2005. Vehicles traveled more than 1.5 billion km each day in 2005, a 237% increase in the total distance traveled compared to 1980 (Figure 3.3). Less substantial but significant rises in the activity of larger mobile emission sources—trucks, buses, aircraft, and trains—have been demonstrated in some parts of the state as well (Reid et al. 2007). Recently, ocean-going vessels have received increased attention because as much as 70% of emissions takes place near port (Corbett et al. 1999). In 2006, there were 10,986 documented port calls in California; of which, 46, 22, and 9% were container ships, tankers, and automobile carriers, respectively (CARB 2008). As source activity has universally increased, sales of fuel have risen sharply. Sales of gasoline increased 66% from 33.9 billion L to 56.1 billion L and sales of diesel increased by 363% from 2.2 million L to 10.1 million L between fiscal years ending in 1970 and 2009 (Board of Equalization 2009).

Activity alone, however, does not determine N gas production. Emissions are a function of the intensity of the activity and the technology being employed. These factors interact in dynamic ways to create emissions. Automobile emissions are a good example. Traffic conditions, the age of the vehicle, and fuel mix significantly affect the total amount and emissions profile (Bishop et al. 2010), not simply the amount of fuel or vehicle distance traveled itself. Age of the vehicle is considered such a significant factor (defines the technology in use and the deterioration of the technology) that it is widely suggested that the oldest 10% of the fleet is responsible for 50% of vehicle emissions (Niedermeyer et al 2006?). It is thus important to consider technological change and adoption in conjunction with source activity to understand the impact fuel combustion has on California's N cycle.

While technological innovation and adoption has helped dampen overall emissions, mobile sources are still responsible for the vast majority of California's NO_x emissions. Eighty-six percent of NO_x in 2008 was derived from mobile sources statewide (CARB 2010). Of these, heavy-duty diesel vehicles, trucks and buses were responsible for 37% of the mobile source emissions (or roughly 31% of the total emissions) (Figure 3.4). Emissions from these sources are now the largest source of NO_x in the state. This represents a departure from previous trends. As little as 16 years ago, NO_x emissions resulted mostly from passenger vehicles. The change in the relative importance of NO_x source can be traced to aggressive technology forcing regulations on passenger vehicles and more lax policy for diesel engines. The California Air Resources Board (CARB) is currently considering rules to regulate emissions from the latter sources. The relative importance of mobile NO_x sources is regionally dependent. That is, industry or energy may contribute a larger or smaller fraction of the total in some regions than in others. As one might expect, mobile sources were responsible for 91% of emissions in the South Coast Air Basin versus just 81% in the San Joaquin Valley in 2005 (CARB 2009). For the latter region, mobile sources accounted for 24% fewer emission in 1975 presumably reflecting population and urban growth in the area. Although mobile source emissions are far greater than other emissions sources across the state, variable contributions of other activities points toward a requirement of regionally tailored approaches to problem.

Technological advances in other areas have decreased NO_x emissions in spite of rising activity levels (Kirschstetter et al. 1999, Yeh et al. 2005). Popp (2010) examines the trends in adoption of NO_x-reducing technology at coal-fired power plants across the US and found that between 1990 and 2002 there was a 375% increase in the adoption of combustion modification technologies, but the use of post combustion technologies lags behind. Power plants in California are not typically coal-fired. However, meeting California energy demand requires import of energy from beyond state boundaries, much of which is produced from coal. In California power plants, greater market penetration of post combustion technologies has occurred. More than 60% of the energy generated with fuel-fired gas turbines in the state apply post-combustion controls (CARB 2004). Engine refinements in vehicles have substantially reduced NO_x from these sources too. New vehicles are required to employ advanced technologies to conform to increasingly stringent regulations. Innovative technologies, such as three oxygen sensors to maintain combustion conditions, have had large effects on emissions (Pokharel et al. 2003).

[Insert: Fig. 3.4]

3.3.2 Dispersal of atmospheric N emissions

Once airborne, NO_x and NH_3 can be transported short and long distances in the atmosphere. Environmental conditions controlling the atmospheric chemistry, transportation, and deposition of N ultimately dictate if, when, and where the N will land (Ying and Kleeman 2009). Nitrogen oxides can be transported from 10s of m to 1000s of km while NH_3, on the other hand, usually deposits after traveling shorter distances. One estimate indicates that 48% of NO_x and 47% of NH_3 produced in Los Angeles landed outside the air-shed (Russell et al. 1993). Transport of airborne N compounds away from the source of emissions make combustion derived N an issue of local and regional concern.

Movement of sources as well as pollutant dispersal means that there is a distinct spatial component to atmospheric N pollution. It is now clear that areas closer to major roadways experience the highest concentration of N-related air pollutants (Durant et al. 2010, Karner et al. 2010). Particulate matter concentrations within 100 m of the I-405 freeway in southern California were almost 3-times higher than those just 200 m further downwind and more than 5-times greater than those 50 m of the source upwind (Zhu et al. 2002). Environments closer to roadways are also larger sinks for N deposition.

Elevated N levels near roadsides are common near both highly traveled highways and near smaller roads and parking lots (Maestre and Pitt 2005, Davidson et al. 2010). The spatial dimension of combustion derived N emissions means that people and ecosystems closer to traffic and industry are exposed to higher levels to damaging N compounds.

3.4 Waste management

Much of the N imported into households, businesses, and municipalities is discarded through residential, industrial, and commercial activities in garbage, refuse, organic materials, and human excretions. The materials are collected, processed, disposed of, or reused as part of the municipal solid waste stream or by publicly owned treatment works (wastewater treatment plants) or onsite treatment systems before disposal. The constituent mass of N in waste products represents a significant latent N pool. Without treatment and appropriate disposal or reuse, the N will eventually contribute to environmental concerns. (Carey and Migliaccio 2009, Kampschreur et al. 2009).

3.4.1 Landfills and organic wastes

Landfills are the primary sink for municipal solid waste. Solid waste contains large amounts of organic materials (and N) making landfills a potential source of N pollution. Degradation of organic materials in landfills primarily creates methane, a potent greenhouse gas (EPA 2011), but small amounts of N_2O also are produced (e.g., equal to 3% of the total global warming potential of one landfill; Rinne et al. 2005). The common practice of covering California landfills with soil to reduce nuisance affect N_2O emission rates; the amount being sensitive to the type of material used (Bogner et al. 2011). Water contamination, in particular groundwater, is a more pressing concern for landfill N than N_2O in California. Reports suggest landfill liners degrade as they age and have the potential to leak NO_3, creating plumes that are difficult to detect with current monitoring approaches (Lee and Jones Lee 1996, Pivato 2011).

California's landfills are filled with organic materials. In 2008, organic materials were estimated to be almost one-third (32.4%) of the overall waste stream (Table 3.6). Of this, food, lumber, and leaves and grass were the 1st, 2nd, and 6th most prevalent materials. Food waste was the largest fraction (15.5%) or in absolute terms, 5,586,552 Mg (CCG 2009). Food waste represented a quarter of total waste discarded from households, a 47% increase in the four years since 2004 (17% versus 25%). Other compostable materials, such as leaves and grass, prunings, branches and stumps, and manures, accounted for 7.2% of waste at landfills. In addition, CalRecycle (2009) estimates that 14.9% of the total waste stream or 5,230,357 Mg is derived from lumber, a 48% increase between 2004 and 2008 despite the precipitous decline in construction activity. These recent studies demonstrate that despite aggressive efforts aimed at waste diversion programs, landfills continue to be a primary receptacle for organic N containing materials.

[Insert: Table 3.6]

Composting and processing of organic waste is a key step to recycle N back to land and reduce environmental N burdens of landfills. Surveys of the composting and processing industries indicate that a significant amount of materials are processed for reuse. Between 2000 and 2008, there was an 81% increase in total products produced at these facilities (Integrated Waste Management Consulting 2007). The way in which materials are applied determines if recycling occurs and greatly influences dynamics. The agricultural industry is the primary sink for organic wastes, accounting for 46% of the total across five regions (Table 3.7). Application of organic wastes helps recycle N into the agricultural system. Distribution and use of these products beyond the processor is relatively unknown. Some 26% of the organic materials are applied at landfills where they are used for beneficial reuse and alternative daily cover. This means they are placed on the surface of refuse to control nuisance (e.g., blowing litter and odor). Using composts in this way does little to alleviate pollution concerns. Part of the determinant of use appears to be location. In Southern California, 50% of the materials are spread as alternative daily cover at landfills. In contrast, nearly half (48%) are recycled to agricultural soils in the Central Valley (Integrated Waste Management Consulting 2007).

[Insert: Table 3.7]

3.4.2 Wastewater treatment and dispersal

Spent water from households and urban areas contain a significant amount of N as a result of the constituent mass of feces, urine, industrial waste, and byproducts of food preparation. Influent N levels vary depending on community size and water conservation. Effluent N levels are determined by treatment, which is largely a function of regulatory requirements for discharge. Treatment may take place in a regional, or centralized, wastewater treatment plant (also known as a publicly owned treatment works) or in onsite wastewater treatment systems.

3.4.2.1 Publicly owned treatment works (POTWs)

Centralized wastewater treatment plants process about 90% of wastewater generated in California. The amount of wastewater created scales with the size of the population with a typical value around 379 L per capita-day, depending on the degree of water conservation. It has also been found that wastewater contains about 14.3 g N per capita-day, however, the nitrogen mass loading is not proportional to the volume of wastewater generation.

When considering the effects of wastewater on N cycling, it is useful to start with collection systems. Wastewater is transported through a system of pipes and pumps to a municipal POTW. Aging infrastructure and seasonally high flow can cause wastewater collection networks to leak through seepage or overflows during transit. During overflow events, N laden waters are released and often reach surface waters. Between 1970 and 2011, there were 11,084 sanitary sewer overflow incidents reported throughout California (CIWQS 2011). Only 10% of the sewage was recovered and 84% or approximately 141 million L reached surface waters (CIWQS 2011).

At the POTW, sewage may undergo physical, chemical, and/or biological treatment. The type and extent of wastewater treatment processes employed has a large effect on nutrient removal and the final N load of the effluent (Table 3.8). For a thorough description of wastewater treatment processes and their effect on N removal see Tchobanoglous et al. (2003). Broadly, the technologies can be grouped into primary, secondary, and tertiary treatment. During primary treatment, a portion of the floating and settleable solids is removed through screening and/or sedimentation in clarifiers. Secondary treatment consists of contact with treatment bacteria for conversion of wastewater organic matter into new bacterial cells and carbon dioxide. The greatest potential to remove N from wastewater occurs during the secondary treatment processes. However, many large wastewater treatment plants perform a limited secondary treatment where insufficient air is provided for nitrification, resulting in high effluent ammonium concentrations. To remove N during secondary treatment, a significant increase in retention time and energy for aeration is needed to accomplish nitrification followed by denitrification in anoxic zones. Thus, the removal of N requires a more intensive secondary treatment process, which may be referred to as an advanced secondary process. To maintain a steady-state secondary process, microbial cells must be removed periodically. These cells, along with the primary solids, are collectively called “sludge” and removed for further processing (see discussion of biosolids below). Tertiary treatment aims to remove any remaining suspended or dissolved materials following secondary treatment using filtration. Tertiary treatment is most often performed to meet regulatory requirements for water reuse projects but does not change N content.

California facilities are treating wastewater to the highest standard in history. Between 1997 and 2008, the percentage of facilities using advanced secondary and tertiary processing increased from 7 – 15% and 18 – 20%, respectively for the facilities reporting (Table 3.8). As described in the 2007-2008 report, nearly 80% of processed wastewater receives at least secondary treatment and there is the potential that 50% of the total flow receives advanced secondary and tertiary treatment. It is important to understand the uncertainty in this statement. Facilities report the levels at which they have the capacity to treat wastewater and the amount of flow they are capable of treating. The proportion of wastewater actually treated to each level is not reported.

[Insert: Table 3.8]

Following processing, wastewater effluent may be reused for various applications or, more commonly, discharged to surface waters or applied to land. For small POTWs, the specific effluent dispersal scheme will depend on the location of the POTW and time of year. However, nearly all-large POTWs discharge to surface waters; including rivers and lakes for inland systems, and to the ocean for coastal cities. By one estimate, 49,227 Mg of solids and 5110 million L of effluent each day are discharged directly into the ocean (Ocean 2010). Most of the ocean discharge is from the Los Angeles (38%) and San Diego (33%) regions. Many coastal wastewater facilities do not remove N prior to ocean discharge. However, inland POTWs are being scrutinized because of the realization, by the public, that wastewater effluent is being discharged into rivers and lakes that are key water supplies for downstream communities; a practice known as “unplanned indirect potable reuse”. It is anticipated that pressure to improve effluent water quality will result in greater implementation of wastewater denitrification systems.

3.4.2.2 Biosolids management

Biosolids consist of primary and secondary solids from centralized POTWs and sludge removed from septic tanks, known as septage. As a result of increasing population, the generation and reuse of biosolids (processed sludge) is also increasing in California. In 1988, it was estimated that 339,450 dry Mg were produced, while in 2009 more than 650,000 dry Mg were generated, a 91% increase over a 20 year period. Most of the biosolids are produced at 10% of the POTWs within Region 4 – Los Angeles - producing nearly 40% of the state total in 1988, 1991, and 1998 (SWRCB 2004, CASA 2009). These reports also suggest the use of biosolids is changing. In 1988, 60% of biosolids were landfilled, while in 2009 more than 61% were applied to land. While the application of biosolids to land is controversial, in part due to the past practice of combining industrial wastes with domestic and commercial sources, it does represent an important opportunity for recycling organic N back to soil systems.

3.4.2.3 Onsite wastewater treatment systems (OWTS)

Developments in remote areas cannot be connected economically to sanitary sewer infrastructure. These facilities utilize OWTS, sometimes referred to as septic systems. The term septic system is used because of the widespread use of the septic tank for low-maintenance primary solids removal. As with primary treatment systems described previously, septic tank effluent contains nearly all of the influent N in the form of ammonium. Historically, a septic tank provided the only treatment prior to land application, usually by subsurface infiltration. However, modern onsite systems can achieve the same level of water quality as centralized facilities. The effluent quality requirements for onsite systems is based on site specific considerations, mostly concerned with leaching and accumulation of nitrate in groundwater.

Between 1970 and 1990, the percentage of California's population using OWTS declined from 12.2% to 9.8% (Census 1970, 1990). Despite this proportional decline, 28% more people (1.09 million) reported using septic systems in 1990 due to population growth. In 2002, it was estimated that approximately 10% of California's population, about 3.5 million people, relied on OWST to treat wastewater and about 12,000 new OWTS are set-up each year (Leverenz et al. 2002).

The effectiveness of the OWTS dispersal system in the treatment and removal of N is dependent on the complex physical, chemical, and biochemical characteristics of the soil (US EPA 2002). The basic model for soil-based N removal from septic tank effluent is adsorption of ammonium on clay particles around the dispersal system, nitrification when unsaturated conditions develop, and denitrification under saturated conditions that occur with the next hydraulic load (e.g., flush of wastewater). Thus, nitrogen removal is compromised under certain circumstances, including sandy soils, high groundwater areas, and in saturated systems.

Because of the lack of control and other challenges associated with incidental N removal in the soil, engineered N removal systems are being required in some areas. These systems utilize the same processes used in centralized treatment systems to convert wastewater nitrogen into an atmospheric gas through nitrification and denitrification. It is anticipated that regulatory objectives to protect the quality of groundwater will result in greater use of OWTS designed for N removal (e.g., SB 885).

3.5 Biological nitrogen fixation

Some plant species have formed mutually beneficial associations with bacteria to help overcome soil N limitation. Most frequently, the symbiosis occurs between the plant family *Fabaceae* (the legumes) and *Rhizobia*, but can also occur in non-leguminous plants such as alder. The plant provides carbohydrates created through photosynthesis to the bacteria in exchange for N fixed from the air. The process is called biological nitrogen fixation (BNF). The rate of fixation depends on the presence of symbiotic organisms, their nutritional status, soil acidity, and soil N levels (Ledgard and Giller 1995). Native and non-native plants in unmanaged and managed landscapes throughout California transfer N from the air to the terrestrial biosphere via this mechanism (Cleveland et al. 1999, Putnam et al. 2006).

The most productive leguminous species in California is alfalfa. The area of cropland dedicated to alfalfa between 1950 and 2007 averaged 432,000 ha and ranged between 368,000 to 484,000, a 32% difference (Figure 3.5). Over that period, average yields increased 53% from 10.5 Mg per ha to 16.1 Mg per ha. The extent and productivity of alfalfa production is relevant because N fixation is proportional to dry matter production (Unkovich et al. 2009). Applying this relationship to 1950 and 2007 figures, alfalfa transfers 44% more N from the air to the land's surface each year on 6% less land (USDA database 2009). Yields of alfalfa are regionally dependent; production increases as one travels south in the state. For example, production was more than 50% greater in the San Joaquin Valley than in the Intermountain Region in 2004 and 2005 (Putnam et al. 2006). Differential yield suggests that the amount of N fixed and the importance of BNF to N cycling will be unique to each region. Other legumes, such as beans, green manures, and clover fix atmospheric N. Their relative impact on the overall N cycle in California is assumed to be minor because of limited use. Due to its extent and productivity, alfalfa is the principal activity driving BNF in California croplands.

[Insert: Fig. 3.5]

Biological N fixation also occurs in the natural and non-agricultural areas of California. Data documenting species coverage is too limited to detail its extent in these areas. We speculate that species capable of fixing N are becoming more widespread for two reasons. First, some invasive species that fix atmospheric N, such as French and Scotch Broom, are becoming more widely distributed (Haubensauk et al. 2004). Second, Caltrans uses vetch, a legume, for bank stabilization and soil improvement along roadsides. Such occurrences suggest potential expansion in the coverage, but the net balance of coverage in N fixing plants is unknown. Another complication in understanding changes in BNF in natural lands is that growth in area does not necessarily equal an increase in BNF. The rate of fixation is sensitive to soil N levels; plants will preferentially take up soil N when it is available. With higher rates of atmospheric deposition, it is plausible that N fixation in many areas is being suppressed, lowering the total amount of N fixed via this mechanism. The capacity for BNF has likely changed, but data are insufficient at present to determine how and to what degree.

3.6 Ammonia synthesis for industrial processes

Fertilizer remains the primary product of the Haber-Bosch process, but many industrial processes use synthetic NH_3 as a substrate. The production of plastics, explosives, dyes, and drugs often includes NH_3 or an NH_3 derivative (Domene and Ayers, 2001). Common products containing NH_3 are: rubber, herbicides, pesticides, plastics, explosives, dyes, resins, cooking utensils, electric appliances, insulators, and nylon. Ammonia synthesis for industrial processes is arguably the least understood, characterized, and analyzed component of the N cycle in California and worldwide. This is problematic as demand for NH_3 for these uses is expected to increase. It is projected that global demand will increase by 21% between 2007 and

2013 alone (IFA cited in ENA 2011). Expansion of the market will partly result from an increased demand for N-containing products and anticipated discoveries of new uses. Consumption decisions of Californians and Californian industries will have some impact on NH_3 demand but the magnitude has yet to be considered. California's influence is likely relatively insignificant due to the comparatively small population in contrast to those throughout the world and in emerging economies.

3.7 Wildfires

Wildfires are an integral part of California's ecology but cause acute loss of N into the environment (Sugihara et al. 2006). During combustion, N contained in the biomass and liter is released to the atmosphere. Airborne N can either be redeposited on the landscape or transported away from the site with air currents, depending on environmental conditions. Incomplete combustion of materials will result in some N remaining in the partially burned biomass. If the fire burns hot enough, N contained in soil organic matter can be volatilized in gaseous N forms as well (Neary et al. 1999). Wildfires change stoichemetric relationships between soil C and N with the lower soil C:N ratios that follow wildfires causing mineral N to release into the soil, predisposing it for loss. It can either be transported off-site as NH_4 by soil erosion or it can leach downward through the soil profile after it is transformed to NO_3.

The degree of N loss is related to a wildfire's intensity. When wildfires burn at high temperatures, e.g., between 400°C to 500°C, 75 to 100% of N is lost; at cooler temperatures, e.g., less than 200°C, only small amounts of N are lost (DeBano et al. 1979, Wohlgemuth et al. 2006). The relationship between temperature and N loss is partially the consequence of more complete and rapid combustion of above ground biomass. The amount of N contained in the biomass (and the latent potential to be released) depends on plant cover. For a mixed-conifer forest, Nakamura (1996) estimates that approximately 10% of the total system N (706 kg per ha) is contained in the biomass. To put this in perspective, complete loss of this N would be more than an order of magnitude greater than soil N emissions from the most intensive cropping systems (assuming 10% gas losses and 600 kg N per ha). Or put another way, the impact on air quality of a single ha burned is greater than 10 ha of the most intensive crop use. Wildfire intensity is also correlated with fuel load and type (e.g., shrubs, litter, or tree canopy). Fuel loads in California have been increasing due to drought, fire suppression, and invasive species. Together, these factors make the probability of ignition more likely and increase the potential intensity of the fire.

Recently the area burned by wildfire in California has increased. Research conducted as part of the 2010 Forest and Range Assessment (FRAP) best characterizes trends and distribution (FRAP 2010). The FRAP indicates that between 1950 and 2008, the area burned by wildfires averaged 128,000 ha per year but ranged between 12,400 and 548,000 ha, a 44-fold difference. Even with high annual variation, recent trends (1990 - 2008) indicate the coverage of wildfires is increasing statewide. Evidence from the Sierra Nevada Mountains and Southern Cascades support this conclusion and show considerable increases in mean area burned since the beginning of the 1980s (Miller et al. 2009). The three years that had the largest area burned all took place in the last decade (2003, 2007, and 2008). However, wildfire has not been equally distributed across ecosystems. Shrubland wildfires have always been the most common, but there has been an exponential increase in burning in conifer forests since the turn of the century (Figure 3.6). The increased extent of wildfires suggests this driver exerts increasing pressure on air and water resources.

[Insert: Fig. 3.6]

3.8 Land use change

Public and private land managers modify land use to maximize societal and personal benefit. Land use change can result from altering the nature of land cover or modifying use. Examples of the former are shifts among natural, agricultural, and urban areas while examples of the latter are shifts in agricultural intensity or urban growth patterns. Land use decisions affect N dynamics in at least two ways. First, they alter the scale and speed of N cycling because N inputs, transformations, and emissions differ considerably among land uses (e.g., intensive versus extensive agriculture; conventional peppers versus certified organic vineyards; see sections 3.2 - 3.9). Second, they modify how the land parcel interacts with the broader N cycle. For instance, agricultural areas tend to be sources of NO_3 to groundwater while urban areas tend to emit N gases into the atmosphere. Nitrogen cycling within various land uses has long been studied. The importance of land use change for N cycling is only recently becoming appreciated and remains poorly characterized. With California's historic land use shifts, it is likely that gross changes to the N cycle have resulted with local, regional, and statewide costs. However, information is too sparse to draw conclusions about the consequences, and knowledge of major land use trends is a first task in understanding the impacts.

3.8.1 Urbanization, urban intensification, and agricultural relocation

Urban growth radically modifies the N cycle. The effects will depend on the type of growth, be it expansive (low-density sprawl called urbanization) or intensive (high-density urban intensification). Each change has corresponding consequences on fuel combustion, fertilizer use and soils, and waste. Urbanization replaces plant cover, often agricultural or natural lands, with a built environment. Natural hydrologic and soil processes are altered or arrested. Fuel combustion per unit area generally increases. Meanwhile, the impacts of urban intensification on some component parts of the N cycle is less clear, can be counterintuitive, and are site specific. For example, one reason for development of high-density communities is to reduce vehicle distance traveled and the associated emissions. Benefits of such designs, however, may be offset due to greater traffic congestion (Melia et al. 2011) and recent evidence from northern California suggests that neighborhood layout is not the determinant of vehicle distance traveled (Handy et al. 2005). Along with both urbanization and urban intensification, comes an attendant increase in N imports (stemming from food, fertilizer, and fuels), impervious surfaces, and engineered drainage, though the magnitude of change differs between the two growth patterns. The structures result in efficient collection and conveyance of N through the landscape. Accumulated N is eventually deposited and stored within the urban areas (e.g., landfill) or exported beyond its boundaries (e.g., sewage disposal into the

Pacific ocean or stormwater disposal into local stream channels). Disposed of N often saturates and overwhelms the environment's buffering capacity and can cause local and regional environmental contamination (Bay et al. 2003, Cadenasso et al. 2008).

Urban areas of California have expanded and intensified increasing their impact on N cycling. Between 1972 and 2000, developed areas increased their land base by 37.5% and are now slightly greater than 4.2% of California's area (Table 3.9). Expansion of urban communities has come at the expense of agricultural and natural areas. According to a recent study that uses historical satellite images to reconstruct land cover and land use change for the eight year period between 1972 and 2000, 697 km^2 ± 306 (sd) of agricultural lands were developed between 1973 and 1980 and 1.4 times this area of grassland was converted to developed areas between 1986 and 1992 (B. Sleeter unpublished data). Concordant with urbanization, population density has risen, but rates are variable depending on the city. The number of people per square kilometer in Fresno and Redding rose by 187 and 382%, respectively between 1970 and 2010 while increases in Sacramento (87%) and South San Francisco (41%) were more muted over the same time period (Census 2010).

Urban growth has indirect consequences on land use in other parts of the state. Urbanization regularly coincides with agricultural relocation; farm operators move to new locations when faced with urban encroachment. Displacement of dairy and citrus producers from the Chino Basin and Los Angeles area to the lower and eastern San Joaquin Valley, respectively are two examples. Agricultural relocation has resulted in only a nominal decline in the agricultural land base despite urban growth. Estimates based on USDA Agricultural Census Data and remote sensing agree, and suggest that there has only been about a 1% reduction in agricultural area statewide since the early1970s (Hart 2003, Sleeter et al. 2010). However, locally conversion of agricultural land may be as high as 10% (FFMP 2010). In the future, urbanization may be more threatening for farmland. Historically much of the growth has taken place in the coastal regions causing farms to relocate to the Central Valley. More recently though, urban encroachment has become an issue of concern in the Central Valley (Schmidt et al. 2010) and rates of urban growth neared 10% for 1992 - 2000 (Sleeter et al. 2010). The Central Valley accounts for 70% of California's cropland and few areas in California with equally suitable conditions remain to absorb further displacement.

[Insert: Table 3.9]

Agricultural expansion to new lands will have unpredictable changes on N cycling. The result will depend heavily on the site characteristics and farm management. Conditions at new sites will require management changes both to maintain profitability and N sustainability. A considerable number of scenarios might be encountered. With expansion, growing conditions may face new combinations of crop, soils, and management that dictate N flows. Thus it is highly speculative to generalize about the impact of agricultural relocation on California's N cycle.

3.8.2 Crop mix

California crop production has become more N intensive. The most obvious consequence of cropland N intensification has translated into an average of 25% higher fertilizer N application rates (section 3.1.1). For the most part, this increased N use has been offset by a simultaneous 37% increase in agronomic efficiency (section 3.1.3). Croplands have become more N intensive in a second, more obscure, way. Plant species require dissimilar amounts of N for growth and reproduction. Differential N recommendations among crops reflect this variation in requirements. Average application rates differ by an order of magnitude between some widely cultivated species. For example, wine grapes receive an average of less than 30 kg N per ha while celery receives closer to 300 kg N per ha. In general, vegetables and nuts are high N users with plant uptake that regularly exceed 100 kg of ha and can be as high as 250 kg per ha. Because of the difference in plant N, changes in crop mix will alter N use and demand. Over the last 35 years, California's crop mix has shifted to include more N-intensive species. Field crops still dominate the agricultural landscape, as of 2008. However, land has been reallocated from field crops to fruits and vegetables (Figure 3.7) and fruits and vegetables were grown on a nearly equivalent amount of land (53% versus 47%). The land area dedicated to field crops declined from 74 to 53% between 1970 and 2007. The shift in crop production towards N intensive crops is at least partially responsible for greater N consumption in the state.

[Insert: Figure 3.7]

3.8.3 Population and intensity of animal production

As discussed, animals require N-rich feed and excrete N-rich manures (section 3.3) and therefore, the size of the animal population influences N cycling by determining the amount of feed needed and waste produced. In California, populations of economically important animal species have grown significantly between 1980 and 2007 (Figure 3.8). The population of dairy cows nearly doubled and the population of broilers tripled. Populations of feedlot steers and other poultry species have varied over this time frame but are were generally equal to or less than levels in 1970. Larger populations require greater resources and create more waste, although the relationships are not necessarily proportionate to the number of animals due to differential efficiencies of animal species and farms. Feed production dictates some cropping patterns in the state (e.g., alfalfa and field corn) and influences those in other regions of the US since a large fraction of the N fed to California animals is grown elsewhere. By changing feed demand, the animal population of California indirectly contributes to concerns of N fertilizer use and soil management in other regions. The amount of waste produced is also a function of the size of the population but it is not directly proportional due to differential efficiency among ranches. Increased creation of manure N becomes a potential pollution concern. This is compounded by the fact that herd/flock sizes have grown at the same time as the total bird population. Without additional land acquisition, ranchers can find themselves in a situation of being animal rich and land poor and thus manure N is concentrated in smaller areas, sometimes without adequate land available for disposal. With proper management, there appears to be sufficient land available to recycle manure in an agronomically responsible way (Pettygrove et al. 2003).

[Insert: Fig. 3.8]

3.9 Conclusion: Universal increases in activity levels

In this chapter, we introduced the eight principal direct drivers that regulate N cycling in California and described historical trends in activity levels. The intensity of each activity has increased universally. The consequence has undoubtedly been more total N released in the environment. The impact changes in an individual action will have on the fate of N is sensitive to variable cause and effect relationships that are significantly influenced by the context of the action. The following chapter (*California's Mass Balance*) provides a detailed accounting of the current state of all the N flows affected by the direct drivers. Thorough consideration of the eight direct drivers described herein is necessary when grappling with ways in which to manage N tradeoffs in California. Technological and policy responses that address critical control points of the direct drivers are discussed in Chapters 8 and 9, respectively.

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Box 3.1. Are University N rate guidelines current?

Since World War II and continuing to the present day, University of California (UC) research has established crop-specific N rate guidelines (Proebsting 1946, 1948, Hartz and Bottoms 2009). An N rate guideline is a range of N application rates expressed as a unit of weight per area (e.g., kg per ha) that are generally able to achieve maximum yield. Most often, N application rate guidelines are printed in University of California Department of Agriculture and Natural Resources (UC DANR) publications and are extended to producers through information channels including: bulletins, production manuals, and field days.

The California Nitrogen Assessment analyzed the current status of N rate guidelines for 33 major commodities grown in California and found publications from UC DANR with N guidelines published within the last 25 years for 28 of the 33 crops. Guidelines for 58, 64, and 86% of the 28 commodities had been published within the last 5, 10, and 15 years, respectively. In most cases, more recent publications were revisions of previous guidelines to incorporate new research, changes in management practices, and crop genetics. Current N guidelines vary widely between their lowest and highest values (Table 3.3). The minimum suggested application rate is often almost 100% less than the maximum rate for any single commodity. When comparing current estimated N application rates with the guidelines, the estimated current rates were above the guidelines for 45% of the crops, and within the guideline for 55% of the crops. For those estimates that were within the guideline, 31% were in the top quartile of the guideline. The results suggest either the guidelines underestimate the N required or producers overapply N.

[Insert: Table 3.3]

Figure 3.1 Synthetic nitrogen fertilizer sales in California, 1946-2009. Since their introduction after World War II, sales (and presumably use) of synthetic N fertilizers has increased an average of 5% per year. Yet they have largely leveled off since the early 1980s. The large rise in fertilizer sales between 2001 and 2002 calls the reliability of these data into question. Source: CDFA (2009).

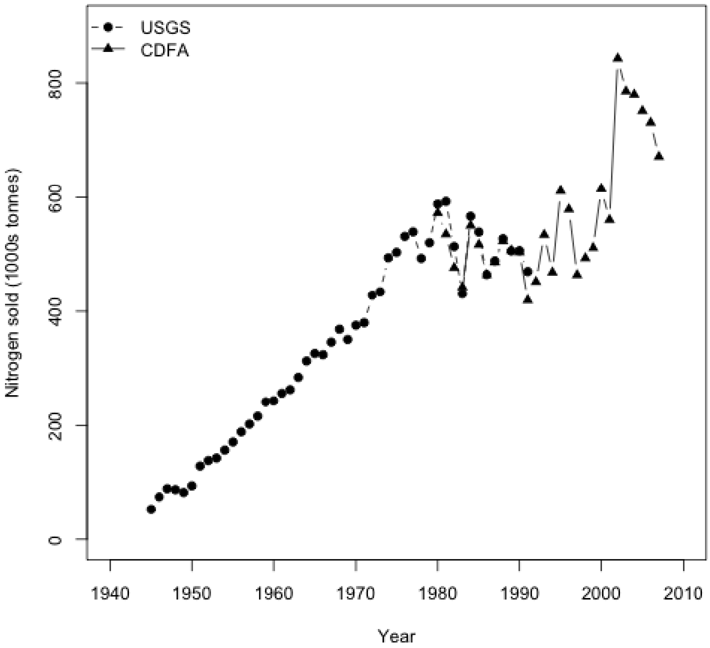
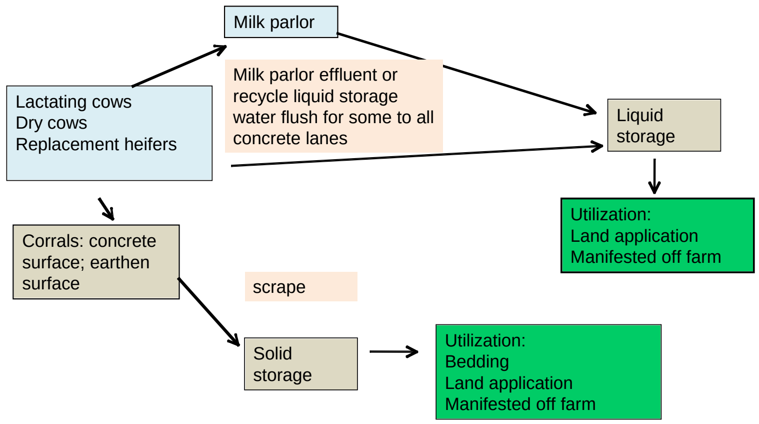


Figure 3.2. Common manure treatment trains on San Joaquin Valley dairies, 2010. (A) Manure flow pathway in freestall systems with or without open corrals. (B) Manure flow pathway in open corral systems. The diagrams shown here demonstrate major processes and the intricacy of manure handling on dairies. Manure management is a complex interdependent system constrained by the facility design. Source: Modified from Meyer et al. in press.

Key: source generation | transfer | storage | utilization

(A) Manure flow pathway in open corral systems.



(B) Manure flow pathway in freestall systems with or without open corrals.

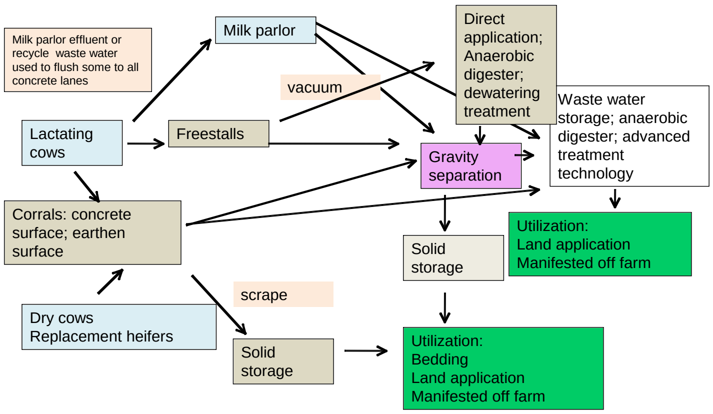


Figure 3.3. Vehicle inventory, total miles driven, and NO_x emissions in California, 1970-2002. Mobile sources are the primary source for NO_x emissions (greater than 86% of the total). Despite large increases in the number of vehicles (population) and the distance traveled (VMT), there has been a significant decrease in emissions. Source: http://www.arb.ca.gov/msei/onroad/images/gallery/catrend.gif.

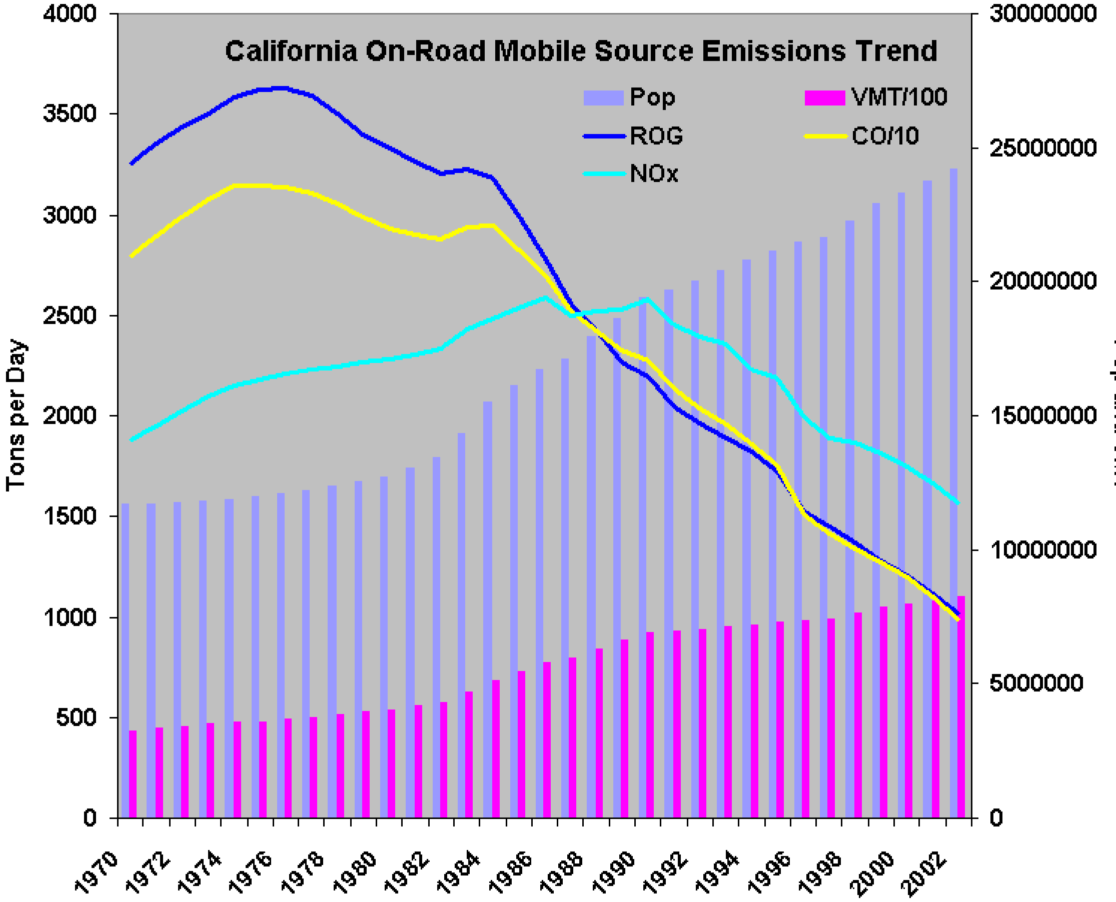


Figure 3.4 Relative contribution of NO_x by major mobile sources in California, 1995 and 2008. The importance of certain sources has recently changed, largely as the consequence of technology forcing policies. Regulations have yet to be implemented to control emissions from diesel engines and port activities but are currently under consideration with CARB. Source: CARB Almanac (1999, 2010).

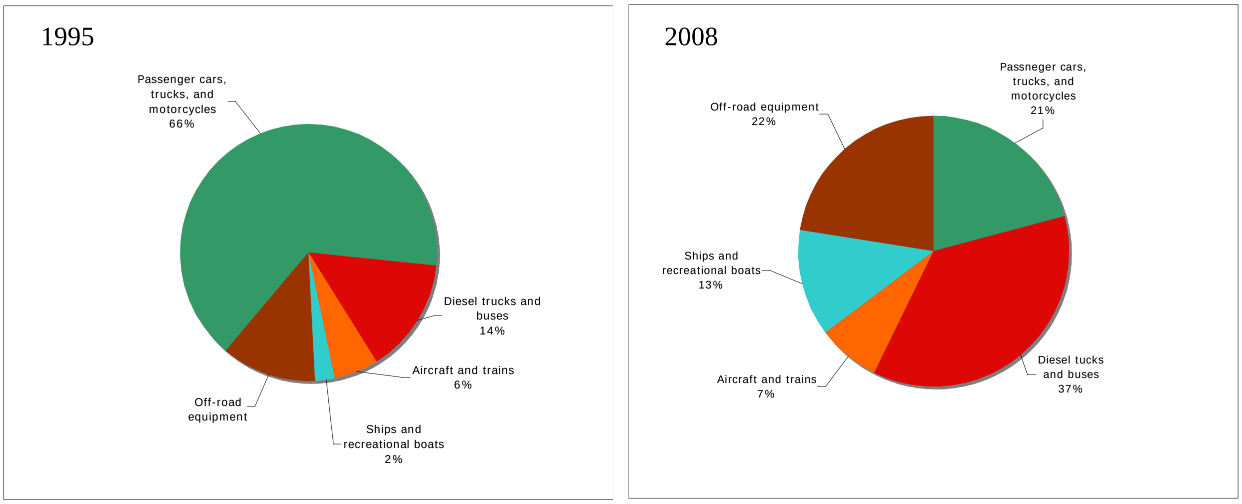


Figure 3.5. Area and productivity of alfalfa in California, 1950-2007. While area has remained relatively the same, productivity has increased markedly. Because biological N fixation is correlated with productivity, these data suggest cropland biological N fixation is an increasingly large source of N into California. Source: USDA (2009).

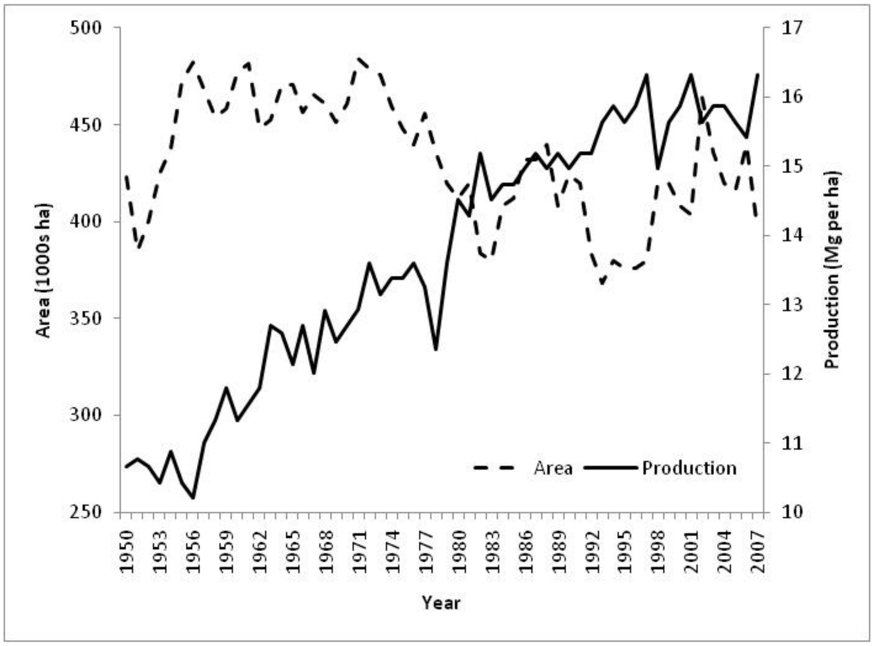


Figure 3.6 Area burned by wildfire in California by decade, 1950 – 2008. Source: FRAP 2010.

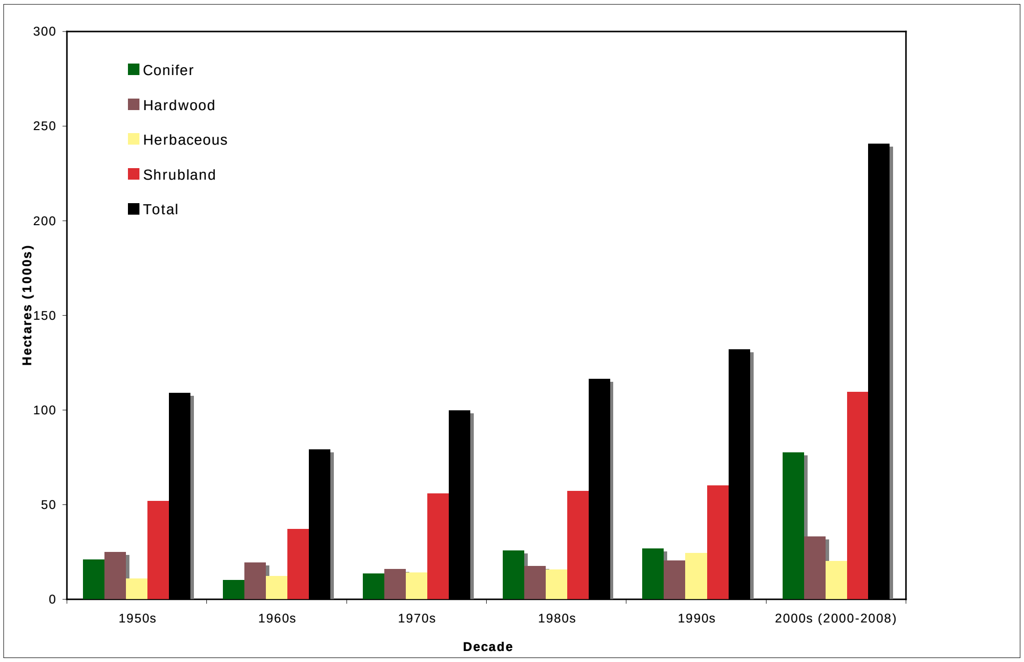


Figure 3.7. Change in cropland area by major crop types in California, 1970-2008. The amount of cropland dedicated to field crops has declined steadily since 1980. Today almost 50% of cropland is used to grow horticultural commodities. Source: USDA NASS (2009).

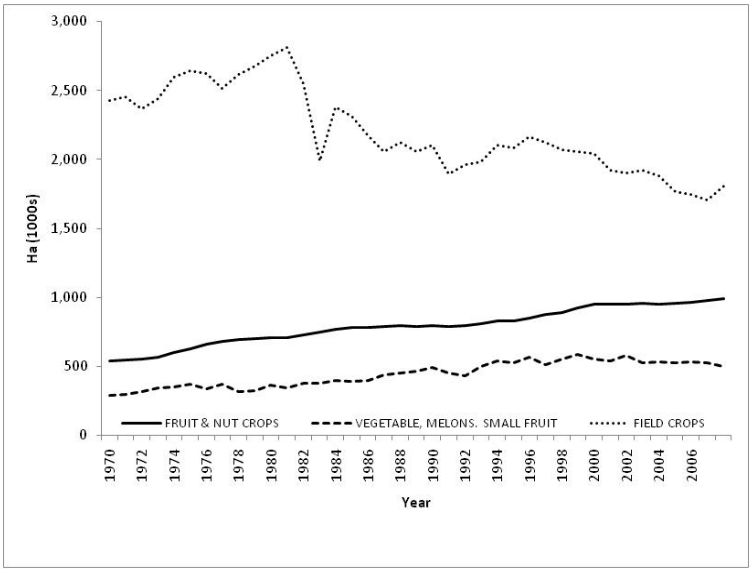


Figure 3.8. Change in California's animal inventory, 1970-2007. The number of milk cows and broilers has more than doubled since 1970 while other animal populations have declined slightly. Source: USDA (2007), USDA (2010).

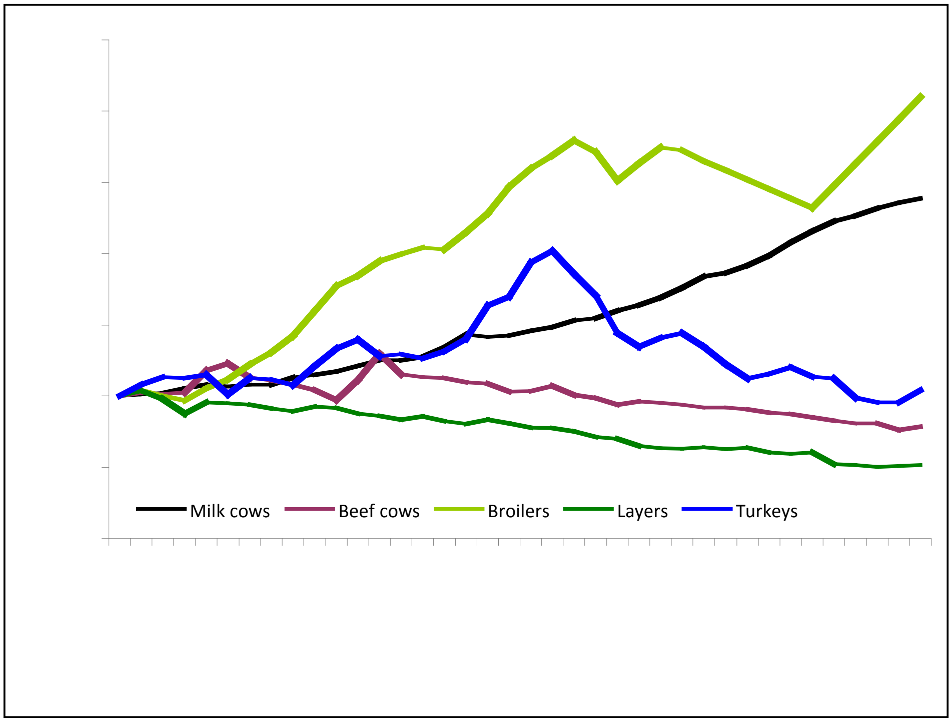


Table 3.1. Fertilizer nitrogen use efficiency (NUE) by ^{15}N, zero-N, and partial nutrient balance (PNB) for select California crops. Compilation of available estimates for fertilizer nitrogen recover for 21 crops. The ^{15}N and zero-N methods are a direct and indirect measure of fertilizer recovery, respectively. PNB is an estimate of total N uptake and does not differentiate fertilizer N from soil N and thus are higher than ^{15}N and zero-N.

	^{15}N[^]		Zero-N[&]		PNB[$]		
Crop	Mean N rate (kg/ha)	Mean RE (%)	N rate (kg/ha): mean [range]	RE_A (%): mean [range]	N rate (kg/ha)	PNB (%)	Source
Almond		17	319 [63, 504]	34 [12, 58]	200	49	Uriu and Micke (1980), Weinbaum et al. (1980), Weinbaum et al. (1984)
Avocado		35			125	19	Rosecrance et al. (unpublished)
Cauliflower	157	44	163 [70, 280]	37 [30, 44]	267	29	Welch et al. (1985)
Celery			327 [168, 504]	61 [26, 41]	290	36	Feigin et al. (1982)
Citrus[%]	128	75			106	36	Feigenbaum et al. (1987), Quinones et al. (2005)
Corn	194	53	210 [90, 360]	50 [28, 66]	239	69	Broadbent and Carlton (1980), Hills et al. (1983), Kong et al. (2009)
Cotton	128	60	135 [56, 224]	24 [2, 52]	195	61	Fritschi et al. (2005)
Grape, raisin-table	50	23	50	65 [54, 70]	49	45	Peacock et al. (1991), Hajrasuliha et al. (1998)
Grape, wine	50	28	67 [56, 112]	9 [1, 23]	30	56	Christensen et al. (1994)
Lettuce	141	26	157 [67, 269]	22 [12, 39]	216	34	Welch et al. (1983), Hartz et al. (2000), Jackson et al. (2000)
Peach/Nectarine			197 [112, 280]	24 [6, 59]	120	28	Johnson et al. (1992), Niederholzer et al. (2001)
Peppers, bell			210 [84, 336]	14 [7, 22]	388	18	Hartz et al. (1993)
Pistachio	418	52			178	56	Weinbaum et al. (1994)
Potato	168	58	168 [68, 270]	54 [19, 93]	278	55	Tyler et al. (1983), Lorenz et al. (2001)
Rice	181	40	125 [101, 188]	50 [11, 73]	146	75	Bird et al. (2001), Eagle et al. (2001), Linquist et al. (2009)
Strawberry			153 [84, 252]	7 [0, 12]	216	34	Bendixon et al. (1996), Welch et al. (1979)
Sugarbeet	155	47	152 [56, 280]	42 [37, 47]			Hills et al. (1983)
Tomato, fresh market			210 [84, 336]	13 [3, 27]	198	61	Hartz et al. (1994)
Tomato, processing	138	33	121 [56, 224]	38 [12, 58]	204	64	Broadbent et al. (1980), Hills et al. (1983), Doane et al. (2009)[*]
Walnut	192	29	212 [90, 359]	1 [0, 11]	155	52	Richardson and Meyer (1990), Weinbaum and van Kessel (1994)
Wheat	194	29	196 [120, 270]	50 [34, 60]	198	56	Wuest and Cassman (1992)

[^] Recovery of ^{15}N measured over one growing season/year except the following (years): almond (2), avocado (0.25), pistachio (2), walnut (6).

[&] Extreme RE_A result from experimental conditions with excessive and deficit N application rates.

[$] Partial nutrient balances calculated in Rosenstock et al. (in review).

[%] Citrus ^{15}N studies conducted in Israel and Spain due to lack of research in California.

[*] Mean ^{15}N RE only includes recovery of isotopically labeled synthetic fertilizer, not treatments with labeled cover crop.

Table 3.2. Trends in California soil management practices. Virtually every practice changes N dynamics in croplands. Few surveys of current management practices are available (i.e., Lopus et al. 2010, Dillon et al. 1999). Information compiled here provides an indication of major changes in management practices.

'+' = increasing, '-' = decreasing, '->' = shift to new practice, '?' = unknown, * = unchanged

Soil management decision	Trend	Description	Source
N application rate	+	The amount of synthetic N fertilizer applied per ha has increased an average of 25% (1973-2005).	Rosenstock et al. (in review)
Source of N	* / ?	Synthetic N fertilizer remains the dominant source of N. Between 1996 and 2007, the distribution of use of synthetic fertilizer products was relatively unchanged, except calcium nitrate increased from 9 to 15% of N sales. The extent of organic N use is unknown. Indirect evidence suggests greater use of organic N.	CDFA (2009), USDA (2010), Klonsky and Richter (2008), Dillon et al. (1999), Expert opinion
Fertilizer placement	?	Perceived shift from broadcast to band placement near plants' roots as solid and extensive distribution of N with irrigation water. Trends are unquantified.	Expert opinion.
Timing of N application	->	Between 1986 and 1996, producers significantly increased the number of N applications per crop. Nitrogen guidelines almost universally suggest split N applications.	Dillon et al. (1999)
Irrigation technology	->	The use of low-volume irrigation technologies has increased by 30% between 1972 and 2001, largely as a result of changes in crop mix.	Orang et al. (2008)
Soil drainage	?	The extent and location of tile drainage is unknown. As much as 1.5 million ha of cropland may be drained throughout the major agricultural valleys.	Pavelin et al.(1987), USDA Agricultural Census (1990)
Tillage	->	The use of reduced tillage and conservation tillage techniques has increased. As much as 17.4% of row crop area may be under conservation tillage in some regions. These numbers may not represent tillage patterns because the intensity of tillage in many crops has been reduced, but the tillage systems may not fit within these categories.	CAWG (2009)
Agro-biodiversity and crop genetic diversity	?	The number of breeds or varieties that dominated California production for top 20 commodities in 1993 ranged between 1-30 with a median of 6.5. Conventional scientific wisdom suggests agrobiodiversity and crop genetic diversity are declining in California but the trend is yet quantified.	Qualset et al. (1995), Expert opinion, Smukler et al. (2010), Brodt et al. (2008).
Field edge / landscape management	?	Installation and management of wetlands, riparian areas, and buffer strips is unknown.	

Table 3.3. Comparison of average 2005 fertilizer nitrogen application rates to University guidelines. The comparison provides a measure to determine if average N application rates are within that suggested by research results. Application rates that exceed the maximum in the guideline suggest that either the guideline does not reflect cropping conditions or growers over-apply N. [1]The percentage of crops with an average N application rate within the UC guideline. [2]The percentage of crops with an average N application rate exceeding the maximum listed in the UC guideline. [3]The amount of N applied above the maximum rate in the guideline.

Crop type	N	Range of guideline (% ± SD)	Within[1] (%)	Over[2] (%)	Mean surplus[3] (lbs. N per acre± SD)
Field crops	4	73 ± 46	100	-	-
Perennials	12	88 ± 54	50	33	14 ± 12
Vegetables and annual fruits	12	101 ± 83	58	42	53 ± 47
All crops	28	90 ± 65	57	36	36 ± 39

Table 3.4. Partial nitrogen utilization efficiencies for select economically important animal species. Partial nitrogen utilization efficiency are calculated as PNUE = (1 - Kg N excreted/Kg N intake)*100). Source: ASAE (2003).

Animal Category	Unit of time	Kg N intake	Kg N excreted	Intake excreted (%)	Partial N utilization efficiency (%)
Layers	20 - 80 weeks	1.04	0.67	65	35
Broiler	48 days	0.13	0.05	40	60
Lactating dairy cow	daily	0.60	0.45	76	24
Feedlot beef cow	153 day on feed	29.38	25.00	85	15
Milk fed calf	daily	0.02	0.01	36	64
Growing finisher pig	120 day grow out	7.12	4.70	66	34

Table 3.5. Manure management practices in California dairy production, 1988, 1997, and 2002. [1]Survey did not include dairies on the North Coast region. [2]Only includes responses from written survey. An additional 45 phone surveys were conducted. [3]Animal housing in SAREP (2004) only reflects the percentage of milking cows under each system. The range for dry cows, bred heifers, calves, open heifers, and other milking livestock are shown in brackets. [4]Flushing in 2002 refers to flushed lanes in scraped drylot and in 1997 refers to "flushing" but does not indicate housing. The management practices used on a dairy will impact N transformations, conservation, and loss, even though managing N was not a primary objective until recently. It is thus important to understand how they have changed over time. Source: Meyer et al. (1997) and SAREP (2004).

	Percentage of respondents		
Practice	1988	1997	2002
Location of dairies	Statewide	Southern SJ Valley	Statewide[1]
Number of dairies		139[2]	428
Housing and manure collection			
Flushed freestall	61.7	77.1	66 [9, 23]
Manure storage ponds	67	95.9	99
Solid separation		54.1	
Settling basins	33	29.7	66
Mechanical separation		9.5	32
Solids processing			
Scraped and piled	60	94.6	
Compost	6	5.4	21
Utilization			
Solid	72	78.4	20
Liquid	91	70.4	48
Both			23
Bedding		27	22
Removed from farm		6.8	3
Sold as liquid		12.2	
Sold as solid	8	58.1	22

Table 3.6. Composition of California's solid waste stream: 1999, 2003, and 2008. Sample #s were 1682, 550, and 751 in the three years respectively. Much of the solid waste disposed of in landfills contains N, raising concerns for N_2O and NO_3^- emissions. Despite becoming a lesser percentage of total waste stream, the absolute amount of organic waste disposed of in landfills has remained relatively constant between 1999 and 2008. Food represents a significant fraction of this waste. Source: CalRecycle (2009).

	1999		2003		2008	
Material	Est. %	Est. Mg	Est. %	Est. Mg	Est. %	Est. Mg
Paper	30.2	9,743,635	21	7,660,512	17.3	6,221,223
Glass	2.8	917,377	2.3	849,792	1.4	513,221
Metal	6.1	1,962,821	7.7	2,825,629	4.6	1,641,383
Electronics			1.2	436,587	0.5	196,181
Plastic	8.9	2,867,672	9.5	3,455,397	9.6	3,453,812
Organic	35.1	11,328,585	30.2	11,034,972	32.4	11,689,451
Construction & demolition	11.6	3,728,247	21.7	7,919,991	29.1	10,501,036
Household hazardous waste	0.3	96,593	0.2	66,754	0.3	109,522
Special waste	3.1	1,007,117	5.1	1,848,857	3.9	1,402,648
Mixed residue	1.8	578,610	1.1	396,765	0.8	300,118

Table 3.7. Regional distribution and use of composting and processing products (Mg) in 2008. Distribution of organic wastes to land represents an important recycling of N into California's N cycle. Based on the recent survey of composting and processors, agriculture and landfills are the primary sinks for recycling organic waste. Source: CalRecycle (2008).

	Region				
Use	Bay Area	Central Coast	Central Valley	Northern	Southern
Agricultural	358,323	413,365	1,534,508	52,048	462,726
Landscape	306,610	97,371	284,656	32,022	461,169
Nursery	60,656	1,768	93,330	1,947	232,499
Caltrans		4,740	16,484		11,385
Alternative Daily Cover	83,694	10,807	140,572	3,131	2,124,637
Biomass Fuel	399,840	56,258	1,014,725	37,858	652,805
Municipal		4,843	4,845	765	10,383
Beneficial Reuse at Landfills	20,425	27,906		174	108,215
Other	112,296	556	88,100		208,297
Total	1,341,844	617,614	3,177,219	127,944	4,272,115

Table 3.8. The level of treatment at California wastewater treatment plants, 1997 and 2008. Considerations: (1) increased treatment decreased N load of wastewater effluent, (2) wastewater is being treated to higher standards, and (3) traditional onsite treatment systems remove only trace amounts of N from wastewater. Source: SWRCB water user charge summaries and SWRCB unpublished data provided.

Treatment level	N removal efficiency (%)	Facility treatment capacity 1996-1997 (%, N = 643)	Facilities treatment capacity 2007-2008 (%, N = 716)[1]	Percent of total CA flow 2007-2008
Primary	3-5	13	12	1.1
Advanced primary	10-50	9	11	19
Secondary	40-60	53	36	30
Advanced secondary		7	15	32
Tertiary	50-90	18	20	18
Onsite systems	3-5			

Table 3.9. Land use change throughout California and in select regions (%), 1973-2000. Statewide, the land dedicated to agriculture has declined only slightly (1%), while developed area has increased 38%. The rate of conversion and specific conversions among land uses is region specific. Source: Sleeter et al. (2010).

Years	Developed	Forest	Grassland / Shrub	Agriculture
California				
1973–1980	9.2	−1.0	−0.5	0.2
1980–1986	6.6	0.3	−0.4	0
1986–1992	11	−1.3	−0.3	−1.9
1992–2000	6.4	−2.1	−1.2	0.7
1973–2000	37.5	−4.1	−2.4	−1.0
Southern California Mountains				
1973–1980	12.7	1	0.2	−1.8
1980–1986	9.7	−2.3	−1.2	0.6
1986–1992	10.7	0	1.5	−3.1
1992–2000	5.7	1.8	−1.6	−0.7
1973–2000	44.8	0.4	−1.1	−4.8
Central Valley				
1973–1980	9.9	−5.7	−8.1	1
1980–1986	5.5	−0.7	−5.6	0.6
1986–1992	8	−0.7	3.8	−1.7
1992–2000	9.8	−2.1	−11.4	1.2
1973–2000	37.7	−8.9	−20.2	1.1

Appendix 3.1. Average N fertilizer application rates by crop, 1973 and 2005. Area is based on a five-year average centered on 1973 and 2005. The average N application rate has only increased 25% over 33 years. However, the magnitude and direction of change is crop specific. Four of the thirty-three commodities comprise more than 50% of total N use accounted for in this analysis: almond, cotton, rice, and wheat. Source: Rosenstock et al. (in review).

	Area (ha)		N rate (kg / ha)			N (% total)	
Crop	1973	2005	1973	2005	N rate (%)	1973	2005
Almond	86462	236800	142	201	41	6	15
Avocado	8144	24728	140	125	-11	1	1
Beans, dry	67760	25600	57	102	79	2	1
Broccoli	17432	47000	204	213	4	2	3
Carrots	12592	28248	134	242	80	1	2
Cauliflower	9264	13624	205	267	30	1	1
Celery	7220	10296	321	290	-10	1	1
Corn, sweet	5680	10224	162	239	47	0	1
Cotton	372840	250400	122	195	60	24	16
Grapes, raisin	96080	96000	64	49	-23	3	2
Grapes, table	26432	33280	64	49	-24	1	1
Grapes, wine	65992	191120	59	30	-49	2	2
Lemons	16608	19360	186	138	-26	2	1
Lettuce	58048	92960	178	216	21	5	6
Melons, cantaloupe	19016	17840	106	182	71	1	1
Melons, watermelon	4480	4768	178	169	-5	0	0
Nectarines	4184	13480	147	116	-21	0	1
Onions	11400	18744	164	237	45	1	1
Oranges	74416	76960	73	106	46	3	3
Peaches, clingstone	20200	11752	149	114	-23	2	0
Peaches, freestone	8440	13360	149	127	-15	1	1
Peppers, bell	3520	8280	181	388	114	0	1
Peppers, chile	1887	2184	181	336	85	0	0
Pistachio		41040	166	178	7		2
Plums, dried	33120	27040	106	146	37	2	1
Plums, fresh	9416	12880	123	116	-6	1	0
Potato	28024	16328	212	278	31	3	1
Rice	165200	214320	96	146	52	8	10
Strawberry	3448	13472	178	216	21	0	1
Tomatoes, fresh market	11272	15520	159	198	24	1	1
Tomatoes, processing	88776	111760	159	204	28	7	7
Walnut	63616	86080	134	154	15	4	4
Wheat	270240	157920	99	198	101	14	10
Average			145	181	25		