Chapter 3: Direct drivers of California's nitrogen cycle

Lead Authors: TS Rosenstock and TP Tomich

Contributing Authors: H Leverenz, D Liptzin, D Meyer, D Munk, J Six, and P.L Phelan

Contents

Boxes:

Figures:

Tables:

What is this chapter about?

The release of nitrogen (N) into the environment is in part a consequence of the inherent properties of the N cycle but is greatly affected by human decisions. This chapter assesses those human and natural processes that directly alter N cycling (hereafter referred to as 'direct drivers'). This chapter considers trends in these on-the-ground actions that influence N use and emissions, following on our examination of N's underlying drivers (chapter 2) and calculations of the relative magnitude of N flows in the state (chapter 4). We document California's relationship with six activities that have and will continue to shape our N cycle : 1) fertilizer use on croplands; 2) feed and manure management; 3) fossil fuel combustion; 4) industrial processes (e.g. chemicals, explosives, and plastics); 5) wastewater management; 6) land use, land cover, and land management.

Stakeholder questions

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

- **What are the current N rate recommendations? Are current nitrogen application guidelines appropriate for present-day cropping conditions?**
- **How is nitrogen use efficiency determined and what are the most efficient and inefficient production systems?**

Main messages

Everyday actions of Californians have radically altered the nitrogen (N) cycle. Basic activities such as eating, driving, and even disposing of waste modify N stocks and flows, transferring N around the state and influencing N dynamics beyond California's border. Six actions fundamentally change N cycling in the state (hereafter referred to as "direct drivers"). Every one of these drivers has become more intensive since 1980 or earlier. Rapid increases in activity levels are, by and large, a function of California's growing population, but some trends can be traced to shifts in contemporary Californian lifestyles and affluence.

Direct drivers catalyze specific N transformations and N transfers between environmental systems. This is significant because it implies there is a close and particular relationship between a direct driver and the N cycle. It also suggests differential relative importance of a direct driver to individual impacts. For example, fertilizer use dominates nitrate (NO_3^-) leaching and nitrous oxide (N_2O) emissions and fuel combustion drives gaseous volatilization of nitrogen oxides (NO_x). By extension, the spatial distribution of activities create distinct regional patterns of consequences (both benefits and costs).

Fertilizer use—inorganic and organic—represents the most significant modification of the N cycle. Sales of chemical N fertilizers (and presumably use) have increased considerably since World War II and risen by at least 40% since 1970, but consumption has leveled off in the past 20 years. Increases in fertilizer use have been exceeded with even greater increases in agricultural productivity. Though the causal impact of N fertilizer is difficult to calculate, N fertilizer has been, and will continue to be, critical for the growth of California's agricultural industry and rural economy. Despite progress, inorganic N fertilizer application rates (kg ha^{-1}) increased an average of 25% between 1973 and 2005. Data show the majority of California crops recover well below half of applied N, with some crops capturing as little as

30%. Similar or even lower N recovery rates are found when organic N sources are used. Differences between the N use efficiency in research trials at plot and field scale and statewide averages suggest there may be substantial potential for improvement in fertilizer N management.

Until recently, manure management decisions were made without much regard to N consequences. The breadth of techniques used, limitations in available information, and large variability among operations, especially for San Joaquin Valley dairies, makes any conclusion about changes in manure management practices tentative. Surveys, however, suggest the recent adoption of manure management techniques help to manage nutrients more effectively. It is important to note that optimal manure N handling is the consequence of many unit processes and thus must be considered as the sum of the entire system; betterment or adoption of individual practices has little impact on the capacity to conserve N in the overall system.

Fuel combustion activities have increased significantly but emissions have declined steadily since 1980. Over the past 30 years, sales of diesel and gasoline fuel, size of the vehicle fleet (both passenger cars and heavy duty trucks), and the number of stationary sources (e.g., energy production and industry) increased measurably, often doubling. Emissions however have been controlled by aggressive technology forcing regulations. This is most evident in the declining importance of the small vehicle fleet for NO_x emissions by comparison to off- and on-road diesel engines.

Ammonia (NH_3) is a common ingredient in a variety of industrial processes - including the production of plastics, nylons, chemical intermediaries, and explosives - however much of its use and impacts are poorly documented. In addition to the direct release of N compounds during production, the longevity of N-derived industrial products (varying from spatulas to counter tops) results in a latent pool of N concentrates in human settlements. Slow degradation of these material means they are a long-term threat to human and environmental health. Assuming reasonable per capita consumption rates for products made with N in developed countries, industrial N use may be responsible for mobilizing an amount of N approximately 55% of that of inorganic N fertilizer use annually.

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 wastewater infiltration systems. Recent 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. Onsite wastewater systems treat the wastewater of more than 3.5 million Californians, with approximately 12,000 new units installed each year. Despite relatively small potential N emissions, improperly sited or functioning onsite systems can cause hotspots of N discharge.

Changes in land cover, land use, and land management fundamentally alter N cycling in ways only recently appreciated. 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, often to lands more marginally suited for these systems. The net effect of urbanization and agricultural relocation/expansion has led to 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. 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 Factors controlling the N cycle

This chapter describes the human actions and natural processes that modify California's nitrogen (N) cycle, referred to hereafter as "direct drivers" (Millenium Ecosystem Assessment 2005). We will first describe relative influence in terms of impact on N stocks and flows[1], and then trace historical trends for each activity. Specific attempts are made to highlight tipping points that have changed the bearing of a direct driver on N cycling in the past and may help to calibrate its future impact on California.

3.1. Relative influence of the direct drivers

Nitrogen is a fundamental component of contemporary society. Its centrality in agriculture, transportation, and industry portends that virtually every human activity, ranging from cooking dinner to waging wars, will affect local and potentially global N cycles, oftentimes in profound, cascading, and multiplicative ways. Population growth, development, and changing affluence have all contributed to a greater quantity of reactive N[2] in the environment today, by an enormous proportion (Davidson et al. 2012). In 1860, humans created approximately 15 Tg of reactive N per year to meet energy and food demand. That amount has now increased by more than an order of magnitude (J. N. Galloway et al. 2009). Few indicators suggest these trends will reverse or even slow significantly in the foreseeable

[1] The chapter does not discuss the underlying economic, cultural, or institutional context shaping human and natural processes or the relative magnitudes of N flows that result from these activities in detail. Those topics are covered in depth in the preceding chapter (*Underlying drivers of California's N cycle)* and the following *(A mass balance for California in 2005), respectively.*

[2] Reactive N refers to all N compounds except inert dinitrogen (N_2).

future. Indeed the opposite, continued rapid growth, seems more likely when one considers forecasts of demand for the two principal factors motivating reactive N creation: food production and energy use.

Analysis summing long-term trends in reactive N creation at small spatial scales (such as California) is unavailable (Box 3.1). However, a recent analysis shows that between 2002 and 2007 reactive N creation in the US increased approximately 4% on balance (Houlton et al. 2012). But reactive N created to enhance food production (cultivation induced N fixation and inorganic fertilizer use) increased ~10% (22.8 to 24.7 Tg) and reactive N from transportation and industry decreased by 19% (5.9 to 4.8 Tg) (Houlton et al. 2012). Differences in the magnitude and trends in N cycling illustrate the significance of developing and deconstructing N budgets by activity to better understand the leverage of individual direct drivers and to target remedial actions (Robertson 1982).

[Box 3.1]

Human activities modify the N cycle through a variety of pathways, each exerting different magnitudes of impact. For example, burning fossil fuels in transportation and industry is the principal source of gaseous reactive N compounds into the atmosphere, the largest fraction of which are nitrogen oxides (NO_x). Ammonia (NH_3) gas is also released but to a much lesser extent. Fossil fuel combustion activities create little threat to groundwater, at least prior to their deposition on downwind landscapes. In contrast, inorganic N fertilizers applied to cropland or urban areas are transported downward through the soil profile (leaching) or laterally on the soil surface (runoff), typically as dissolved nitrate (NO_3^-). The propensity for certain activities to catalyze specific N transformations and transfers between environmental systems implies two significant considerations. One, there is a close and particular relationship between a direct driver and the N cycle. They act to introduce or alter specific N stocks or flows. Wildfires, for instance, liberate organic N contained in soils and biomass (N stocks) and cause acute release of reactive N compounds and dinitrogen gas (N_2) into the atmosphere (N flow). Two, the importance of any direct driver and the likely changes in the N cycle are a function of the extent by which activities take place ("activity level"). The diversity and spatial patterns of human activities in California presuppose that direct drivers will have differential degrees of regional impact. Urban areas of Southern California receive a larger proportion of the reactive N input from fossil fuel combustion or wastewater treatment while fertilizer use determines the introduction and fate of reactive N in the Central Valley.

The California Nitrogen Assessment's mass balance calculations indicate the relative magnitude of N flows in the state (Chapter 4), identifying five direct drivers that control California N cycling: (1) fertilizer use on croplands; (2) feed and manure management; (3) fossil fuel combustion; (4) industrial processes (e.g. chemicals, explosives, and plastics); and (5) wastewater management. Statewide, fertilizer use on croplands and urban areas introduces the largest single source of new N in California, responsible for 32% of new N imports (Figure 3.1). Fossil fuel combustion contributes a significant amount of new reactive N to California each year too (25%), followed by biological nitrogen fixation (21%), and imported feed (12%). Only a few direct drivers regulate the release of reactive N into air and water resources (Figure 3.1). Fossil fuel combustion dominates gaseous emissions (44%). It is worth noting that the vast majority of these emissions are in the form of NO_x which has important consequences for regional air quality. Meanwhile, manure handling is responsible for the majority of the NH_3 emissions, which account for 22% of total atmospheric N release. Croplands are overwhelmingly responsible for N loading into groundwater across the state (88%). Harter et al. (2012) indicate that croplands contributed 96% of the NO_3^- to groundwater in the Salinas Valley and the Tulare Lake Basins in total and 54% and 33% from inorganic fertilizer and manure use, respectively. By comparison to groundwater, multiple sources contribute to surface water N loading including natural lands (40%), fertilizer use (49%), and wastewater (11%). The mass balance is a static model documenting one year's (2005) N flows; it does not capture temporal dynamics. That limitation, and the understanding that individual land uses affect N cycling in vastly different ways, leads us to identify 'land use, land cover, and land management' as a sixth direct driver. The following sections analyze trends in these key activities and provide context for historical changes.

[Figure 3.1]

3.2. Fertilizer use on croplands

Nitrogen availability generally limits plant productivity. Crop producers respond by applying N fertilizer to soils to enhance plant growth and reproduction. Fertilizer N typically stimulates soil N cycling. Not only does the size of soil mineral N pool increase, but soil microbial activity increases, the pace of N transformations and soil N turnover intensify, and the risk of N emissions typically increases. The fundamental nature of the soil N cycle requires producers to apply more N than plant demand to ensure adequate nutrition (Box 3.2; Box 3.3). If managed well, plants capture a sizeable fraction of the fertilizer. However, due to soil N dynamics and practical limitations of production systems, agriculture and lawns are inherently leaky systems and some N inevitably escapes into the environment.

Fertilizer use on croplands introduces the most significant annual amount of new reactive N from a single source into California (Chapter 4). Inorganic N fertilizer use on croplands amounts to 466 Gg N $year^{-1}$. Organic N use introduces nearly an equal amount (459 Gg N $year^{-1}$) through manure application and cultivation induced biological nitrogen fixation (C-BNF; or cropland fixation). Fertilizer use—inorganic and organic combined—thus is responsible for mobilizing slightly less than 1 Tg of reactive N and has a significant leverage on the overall dynamics of N in California.

[Box 3.2]

[Box 3.3]

3.2.1. Inorganic N fertilizer use on farms

Inorganic N fertilizer (synthetic N fertilizer) has played a critical role in increasing agricultural productivity and food security globally. It has been suggested that the scientific discovery (e.g., Haber-Bosch) of creating inorganic N fertilizer has resulted in more than 2 billion people alive today than would be otherwise (Erisman et al. 2008). Smil (2000) suggests inorganic fertilizer N is the basis for more than 50% of food produce. Data from long-term experiments suggest that between 40% and 60% of crop yields in the US and Europe can be attributed to inorganic N fertilizer use, a slightly lesser proportion in tropical environments (Stewart et al. 2005). The fundamental importance of fertilizer N to food security requires that any discussion of past, present, or future inorganic fertilizer use must acknowledge its benefits to society.

3.2.1.1 Trends in inorganic N use and yields

Sales of inorganic N fertilizer have increased 12-fold since materials became widely available after World War II. Prior to this time, inorganic N fertilizers, also known as mineral fertilizers, were derived from Chilean nitrate deposits. However, with the invention of the Haber-Bosch process[3] in 1908, availability of inorganic fertilizer N radically changed (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 inorganic fertilizer ensued. The consequence has been a massive increase in the use of N fertilizer in the developed and parts of the developing world (J. N. Galloway et al. 2009).

In California, inorganic N fertilizer sales (and presumably use) have grown at an average annual rate of 5% between 1946 and 2009 (Figure 3.2). Annual sales grew at their fastest pace prior to 1980. Since that time, sales of N fertilizers have leveled off; recent annual sales of approximately 600,000 Mg of N fertilizers are not distinctly higher than sales in 1980. In the recent past (since 2005), N fertilizer

[3] The Haber-Bosch process uses high temperatures and pressure to break the trivalent bond between N atoms in dinitrogen gas (N_2) from the atmosphere to synthesize ammonia (NH_3).

sales have continuously declined possibly because of increased price and decreased use by the dairy industry. It is worth noting that California no longer fixes NH_3 at industrial scales and all the inorganic N fertilizer sold in California today is imported from beyond the state's borders. California's fertilizer manufacturers refine imported NH_3 into other products, such as ammonium nitrate or specialty fertilizer blends, which are then applied in California's crop fields.

[Figure 3.2]

Statewide sales data present a limited picture of inorganic N fertilizer use. Farm operators make fertilizer decisions at the field-level subject to local constraints. Decisions for an individual parcel of land determine the intensity, effectiveness, and outcomes of N use. Disaggregated knowledge of inorganic N use at this level is thus paramount to understanding the cause and effect of N fertilizer use in California. Unfortunately, finer resolution N use data is practically nonexistent for California (see Data tables).

A first step to identify leverage points, hotspots, and target action is to examine N use by crop. We documented changes in N application rates, yields, and cropped area for 33 important California crops between 1973 and 2005 (Rosenstock et al. 2013). Average N fertilizer application rates (kg ha^{-1}) across the 33 crops surveyed increased 25% over this 33-year period (35% when considered on an area-weighted basis), the magnitude and direction of change being crop specific (Appendix 3.1). 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 significant because the area dedicated to these crops increased simultaneously with higher N application rates (Figure 3.3; Appendix 3.1). Since many high-value vegetable and nut crops saw the greatest increase in fertilizer N use and typically recover a smaller percentage of that N than the field crops they replaced, cropping and N use trends suggest a greater threat for N loading to the environment.

[Figure 3.3]

Whereas our estimates of fertilizer use represent a necessary first approximation of inorganic N fertilizer use in California, they fail to capture the variation in inorganic N applications among fields, farms, and regions. Application rates may easily vary 50% to >100% depending on the soils, irrigation system, weather, and grower preference, even for the same crop due to edaphic characteristics of the production system and climate. A 1973 survey of fertilizer use in California demonstrates the extent of heterogeneity (Rauschkolb and Mikkelsen 1978). Fertilizer N use ranged by an average of 135% between the minimum and maximum reported rates within a region for the 45 commodities surveyed. Among regions, average application rates varied approximately 34%[4]. The ranges reported in the 1973 survey are only illustrative and cannot be assumed to reflect today's cropping conditions. Significantly greater resolution of data is needed to better constrain basic questions on who, where, and how inorganic N fertilizer is used throughout the state and to begin to examine cause and effect relationships (Rosenstock et al. 2013).

Increases in fertilizer N use have supported higher crop yields. Yields of California crops increased dramatically during the period of rapid expansion of use of inorganic N. For example, between 1950 and 2007, yields of almonds, processing tomatoes, and rice increased by 349, 221, and 136 percent, respectively (USDA 2013a). Most cropping systems have seen similar rates of yield increase. The relative contribution of yield increases that can be directly attributed to inorganic fertilizer N has not been systematically analyzed for California agriculture. However, trends of inorganic N fertilizer sales and agricultural productivity in California show increases that parallel global trends suggesting similar benefits to that described previously. Without the availability of inorganic N fertilizer, the substantive growth of California's robust agricultural economy in the last half century would have been improbable, if not impossible.

[4] Calculated as the average of coefficient of variations (standard deviation/mean) among average reported rates for each commodity for all reported commodities.

3.2.1.2 Inorganic nitrogen use on major California crops

Californian farmers grow a remarkable diversity of crops on more than 4 million ha. The total number ranges from approximately 150 to greater than 400 crops, depending on the source of information and year of interest. Despite the variety, much of California croplands are planted with only a handful of species. Fewer than twenty crops are grown on at least 1% of the state's cropland. And alfalfa, almond, grapes, rice, wheat, and corn cover approximately 16, 9, 8, 8, 7, and 6% of the harvested cropland, more than 50% of the total[5, 6].

Crop species require different amounts of N for growth and reproduction. Plant N requirements regularly exceed 100 kg N ha^{-1} and can be more than 250 kg ha^{-1}. Differential N recommendations among crops reflect this variation in demand (Appendix 3.2). Average application rates differ by an order of magnitude among widely cultivated species (Appendix 3.3). For example, wine grapes receive an average of less than 30 kg N ha^{-1} while celery receives closer to 300 kg N ha^{-1}. Total amount of fertilizer N used on a given parcel of land is a function of the cropping pattern. Perennial crops only have one crop per year. Land planted to annuals however is often double or even triple cropped. Rotating annuals on a single piece of land greatly increases the N intensity. Fertilizer N use on a lettuce-broccoli rotation in Salinas may receive between 300 and 550 kg ha^{-1} $year^{-1}$, far greater than either single commodity by itself. Considering the cropping system rather than individual crops is an important distinction for understanding the causes and effects of N use.

Substantial differences in cropped area and fertilizer N application rates suggest certain crops have a larger impact on overall N dynamics than others. Multiplying the area harvested by average fertilizer N application rates for 33 crops in California shows that only four—cotton, almond, rice, and

[5] Based on average reported cropped areas 2003-2007 from California Agricultural Commissioner Reports (USDA 2013b).

[6] These values are based on estimates of inorganic fertilizer use. Manure applications to silage corn and cereal forages are significant and total N applications from manure to these crops may be on par with the crops listed here.

wheat—account for 51% of the total N applied. This supports the notion that a relatively small number of cropping systems have a disproportionate mark on California cropland N use and cycling. Notably, nursery or greenhouse industries were excluded from these calculations because of both data limitations and the fact that ornamental horticulture production systems tend to be among the most intensive N users with use ranging from 100 – 7,000 kg per ha (Evans et al. 2007). Further, fertilizer use is not distributed equally among crops. Of the 345,900 tons of N fertilizer accounted for in the application rates of the 33 commodities, approximately 34% is applied to perennials, 27% to vegetables, and 42% to field crops. Notably, this estimate shows that a relatively few number of crops dominate the total. While conventional wisdom already assumed this to be true, these data provide evidence of the relative magnitude of the difference. For perennials and field crops in this small group, these estimates may be conservative since only bearing and harvested areas, respectively, were used in these calculations. Even with the uncertainty surrounding the precision of our estimates and the relative changes in cropped area that occur year-to-year, it is difficult to imagine a scenario where additional crops could exert as much leverage over total N use in the state in the short term. Understanding N management (and the fate of applied N) in these systems, which include a representative range of crop types that are commonly grown with an array of soil, irrigation, and fertility management practices, then becomes the highest priority.

Differential plant N demand and changes in the extent of planting alter total statewide crop N use. Over the last 35 years, California's crop mix[7] has shifted heavily from field crops that often receive less N fertilizer to more N-intensive species, (e.g., vegetables and nuts). Field crops are still grown on the majority of croplands, as of 2008 (Figure 3.4), but the state's land area dedicated to field crops declined from 74 to 53% between 1970 and 2007. Fruits and vegetables are now grown on a nearly equivalent

[7] Crop mix refers to the composition and relative amount of each species grown.

amount of land (53% versus 47% in 1970). The shift in crop production towards N intensive crops may be partially responsible for increases in N consumption in the state.

[Figure 3.4]

3.2.2. 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 fertilizing materials include manures, composts, waste products, and leguminous cover crops (Mikkelsen and Hartz, n.d.; Hartz and Johnstone 2006; Gaskell et al. 2000).

Organic N use can either represent a transfer of N internally around California or an introduction of new reactive N into the state. Manures and composts are examples of the former. Cows do not create N, it simply passes through them during the conversion of feed into manure. Nitrogen in manure is derived from biological N fixation (e.g., from alfalfa), the Haber-Bosch process (e.g., when fertilizer is applied to feed crops), or from soil reserves. Compost represents another transfer since it is a collection of N from different waste products (e.g., food waste, manure, and urban green waste). Leguminous plants grown for green manures are the exception. They introduce new N into the biosphere by fixing atmospheric N through biological means and incorporating it into biomass and eventually soil.

Organic N use and recycling drives two significant components of California's N dynamics. Manure use redistributes slightly less than half as much N as inorganic N fertilizer applications and alfalfa production (which mostly ends up as animal feed supporting the dairy industry) creates 180 Gg N year^{-1} through cultivation induced biological N fixation (Chapter 4). Thus, organic N drives multiple N transfers internal to California's N system and its fate is important to understanding overall N dynamics.

Some evidence suggests that organic N sources typically improve soil health (Reganold et al. 2001). Additional organic matter applied with organic N is the root cause of many benefits to soil, including improved soil structure, hydraulic conductivity, water holding capacity, biotic activity, and nutrient retention (Laurie E Drinkwater et al. 1995). It has also been suggested that organic systems reduce pollution pressure by stimulating higher rates of denitrification to N_2 (Kramer et al. 2006) and reduce leaching pressure by comparison to inorganic N sources (L. E. Drinkwater, Wagoner, and Sarrantonio 1998). However, organic N cannot be assumed to be less damaging to the environment under all conditions. Research shows that organic materials represent a significant source of reactive N to the environment, both gaseous and solution, because of the difficulty in managing the timing of N release from soil organic matter (Barton and Schipper 2001; Kirchmann and Bergström 2001)).

3.2.2.1 Trends in organic nitrogen use

Unlike inorganic N fertilizing materials, organic N creation and fate is not tracked or the information is not publically available in most cases. What this means in practice is that it is extremely difficult to piece together a coherent account of organic N use in California, today or historically. The consequence is an inability to discern leverage, impact, or evaluate management. A survey conducted by Dillon et al. (1999) suggests that organic N 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 reported growing new crops, organic N use rose to 55% of respondents between 1986 and 1996. Indirect indicators further 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) and the land dedicated to these systems has grown rapidly in recent years, though it still only accounts for a small fraction of actively cultivated cropland in any given year (less than 4%). 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 certified organic in 2008, suggesting nearly a doubling in the 8 years between 2000 and 2008 (Klonsky and Richter 20057; 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). The increased demand for organic N is matched by increased supply. Large increases in animal and human population have resulted in a greater availability of N-rich manures, composts, and urban wastes destined for land application than ever before.

The distribution of organic N may become more transparent in the future. The State Water Quality Control Board (SWRCB) requires documentation of distribution of liquid manure associated with dairy production in the San Joaquin Valley and biosolids for regulatory compliance to minimize water quality concerns. Dairy manure constitutes the largest proportion of organic N utilized in the state and thus more information on characteristics, distribution, and attributes of use would be a major step toward better understanding of this driver. Public availability of these data is questionable, however. And currently these data are not easily evaluated in their 'hard copy' form. Modernizing the reporting system would increase the utility of the data collected and potentially reduce the costs of compliance for producers. Another significant gap in the current reporting system is information on the distribution and application of solid manure. When sold and transported off-farm (often to composters), manure quantities are recorded, but the manure's final location of application is not. With as much as 50% of dairy manure and 100% of poultry and beef feedlot manure exported and applied to land offsite, assessing the significance of the transfer of N from animal systems to croplands is nearly impossible. Additionally, the total size of the solid manure N flow may increase in the future as dairy operators are forced to manifest greater quantities of manure solids offsite to comply with water quality regulations.

3.2.2.2 Manure use on croplands

Approximately 263 Gg of excreted manure N is collected and applied to croplands (Chapter 4). Utilizing manure N as a fertilizer is discussed in other sections, along with organic N sources more generally. This section emphasizes two issues of particular relevance to understanding manure N dynamics: material placement and geography.

Where manure N is applied, either on top of or injected within the soil matrix, preconditions its fate. Manure N applied to the soil surface is more likely to be volatilized. Higher rates of emission from surface applications by comparison to incorporated manure are a function of soils being strong NH_3 sinks and thus injection of liquid manure and incorporation soon after broadcasting solid manure creates a boundary layer between manure N and atmosphere. Placement of manure even 2 cm below the soil surface reduces NH_3 emissions from 25 to 37% (Sommer and Hutchings 2001). Manure incorporation however is not a panacea. It increases soil N concentrations and can lead to higher rates of NO_3^- leaching unless additional abatement steps are taken (Velthof et al. 2009).

Confined dairy systems in the San Joaquin Valley surface apply liquid manure to feed crops close to the production unit. In the most recent manure practice survey, zero respondents reported injecting manure below the soil surface (D. Meyer et al. 2011), which suggests common practice predisposes N to extensive volatilization from fields. Because manure injection requires specialized equipment and resources, switching practices would require transformative infrastructural changes which are likely costly and logistically prohibitive at the current scale and under current economic constraints of California dairying.

A second consideration for land application of manure is the spatial distribution of animals. California animal production has historically been in concentrated areas (e.g., Chino basin and now the Southern San Joaquin Valley) and has become more intensive in the recent past. Intensification has increased herd and flock size, especially per unit area. The result is a concentration of waste and an increased probability of over-application. Operators become N-rich and land/crop poor, putting pressure on ways to dispose of N. However, it is not clear that there is insufficient land associated with animal production units to effectively utilize manure N. Pettygrove et al. (2003) estimate that as much as 200,000 ha of land may be associated with dairies in the San Joaquin Valley and available to receive manure applications. Since manure N application rates are now determined by crop uptake due to the SWRCB General Order for Dairy Waste Discharge, one might expect an increase in the number of operators moving to triple crop practices (3 crops in one year) to increase off-take. Triple crop systems assimilate more than 600 kg ha^{-1}, which permits operators to apply 840 to 990 kg N ha^{-1}, making these the most N intensive cropping systems in the state. By way of contrast, the most N intensive cropping systems (e.g., double cropped cool-season vegetables) typically apply inorganic fertilizer N at approximately 2/3 these rates, ~600 kg N ha^{-1}.

3.2.2.3 Cultivation induced biological N fixation

A specialized and taxonomically diverse group of prokaryotes use the enzyme nitrogenase to convert atmospheric N_2 gas to NH_3. The organisms can be free-living soil and aquatic biota (e.g., *Azobacter* or *cyanobacteria*) or form associative (e.g., *Azolla*) or symbiotic (e.g., *Rhizobium*) relationships with higher plants. Symbionts, *Rhizobium* bacteria, are the most important group of N fixers in agricultural ecosystems. Prior to the invention of the Haber-Bosch process, biological nitrogen fixation (BNF) was the primary way N moved from the atmosphere to the biosphere, and the abundance of N fixers regulated ecosystem productivity. Still, BNF is thought to contribute ~128,000 Gg N yr^{-1} globally (J. Galloway et al. 2004).

Biological nitrogen fixation adds approximately 335 Gg N yr^{-1} to California's terrestrial ecosystems, an amount equal to 65% of that applied as inorganic N fertilizer (Chapter 4). The majority of BNF (58%) is cultivation-induced (C-BNF). That is, production of food and feed drives the planting of crops that utilize BNF to satisfy N requirements. BNF in California takes place in systems planting legumes and rice. While BNF is possible in multiple cropping systems, alfalfa dominates the total C-BNF flux (92% of total; Chapter 4) because of its productivity and areal extent. The relative impact on the overall N cycle in California of other legumes is assumed to be minor because of limited use.

Understanding how alfalfa yields and cropped area have changed provides information on the historical and current importance of C-BNF as a direct driver. Absolute N fixation rates for California alfalfa are difficult to assess because studies have not thoroughly measured above and below ground biomass production across the range of soils and weather conditions. But fixation rates can be inferred from yields. N fixation in alfalfa is proportional to dry matter production (Unkovich, Baldock, and Peoples 2010). Between 1950 and 2007, statewide average alfalfa yields increased 53% from 10.5 Mg per ha to 16.1 Mg per ha. Over the same time period, the area of cropland dedicated to alfalfa remained almost unchanged. It increased 4% from 423,000 to 440,000, but averaged 432,000 ha and ranged between 368,000 to 484,000 across these years (Figure 3.5). Assuming a direct proportionality between N fixation and yield, the yield increase and negligible areal increase suggests alfalfa transfers 44% more N from the air to the land's surface each year on a similar land base (USDA 2013a). Though a significant increase, the rise has been less pronounced than the trends seen for inorganic fertilizers and fuel combustion.

Alfalfa yields are highly regionally dependent. For example, production was more than 50% greater in the San Joaquin Valley than in the Intermountain Region in 2004 and 2005 (Summers and Putnam 2008). Higher yields largely result from a longer growing season that increases the number of cuttings. Latitude is generally a good predictor of yield (and hence fixation). Differential yield suggests that the amount of N fixed and the importance of BNF to N cycling will be unique to each region and hence the total impact will depend on the spatial distribution of crop patterns and the location in the state.

[Figure 3.5]

3.2.3. Agronomic nitrogen use efficiency (NUE)

Higher rates of N fertilizer application are not problematic, if fertilizer N recovery increases in concert or at faster rates. Concerns about field practices arise because growers must apply more N than crops require for growth and reproduction because of inherent inefficiencies of production systems and soil N dynamics (Box 3.2). Hence, the portion of N not taken up by plant roots remains in soil after harvest, is vulnerable to be released as potential harmful reactive N compounds, or is denitrified to inert N_2 gas. The relative proportion attributable to each fate depends heavily on the soil's physical and chemical properties and crop management (Appendix 3.4). Only a small fraction of the N applied beyond plant uptake ('surplus') is used in the subsequent growing seasons, often less than 10% (J. Ladha et al. 2005). Research demonstrates that the surplus N is particularly vulnerable to loss from the soil system as reactive N. Both NO_3^- leaching potential and the rates of gaseous N_2O emissions increase nonlinearly with increasing surplus N (Broadbent and Rauschkolb 1977; van Groenigen et al. 2010). Surplus N emissions may occur either during the season, as in the case with leaching in many irrigated systems, or following harvest when soil N levels are high. Furthermore, surplus N represents an unused resource and expenditure for the producer, an economic loss. Therefore, knowledge of the amount of N fertilizer applied and taken up is critical to understanding the fate of N fertilizer use.

3.2.3.1 NUE when using inorganic fertilizer

Measures of agronomic N use efficiency[8] are ratios of plant N uptake to the amount of N fertilizer applied. NUE is one of the most often cited, and unfortunately, most often misinterpreted indicators of cropland N use. Mistakes arise because there are at least 18 different ways to calculate NUE, each

[8] There is a distinction between agronomic NUE and economic N efficiency. They are not interchangeable terms and care should be taken when discussing NUE to a multidisciplinary audience. Agronomic efficiency measures the ratio of N assimilated to N applied and represents the technical potential of the system. In contrast, economic efficiency measures the rate of economic return for adding an additional unit of N and is subject to market prices of crops and inputs. When the agronomic efficiency of N application rate is at a maximum, economic efficiency is usually not. The remainder of this section refers to agronomic NUE when it discusses 'NUE'.

quantifying slightly different components of the soil-crop system (J. Ladha et al. 2005). Thus, assessment of NUE needs to be executed with caution, explicitly defining the terms and knowing their limitations. Two of the most common methods estimating NUE are the difference method (zero-N) and the isotope dilution method (^{15}N). They are calculated with equations 1 and 2, respectively:

(1) $$NUE_{zero-N} = \frac{U_F - U_0}{N} \, x100$$

where U_F is the amount of N in aboveground biomass measured in a fertilized plot, U_O is the N in aboveground biomass in an unfertilized plot, and *N* is the amount of fertilizer applied. The isotope dilution method applies labeled radioactive N isotopes to determine the amount of plant uptake by the following:

(2) $$NUE_{^{15}N} = \frac{^{15}N_{recovered}}{^{15}N_{applied}} \times 100$$

where the proportion of ^{15}N in the plant (over background levels) is relative to the ^{15}N fertilizer applied. The principal benefit of utilizing these methods is that they differentiate between N sources - fertilizer or soil reserves. The major limitation is their requirement of controlled experimental plots that may not reflect field-scale N dynamics. The representativeness of prior research to current practices is further suspect because much of the work utilizing these methods in California were performed long in the past (1970s), recent work on rice being an exception. Regardless, these methods provide the most accurate characterization available of fertilizer N recovery efficiency in the state's crops (Table 3.1). In general, zero-N methods tend to overestimate the inorganic N fertilizer recovery and the ^{15}N approach underestimates it (Broadbent et al. 1980).

[Table 3.1]

Globally, the efficiency of inorganic fertilizer N applications ranges between 30% to 50% in the first growing season for cereal crops (Tilman et al. 2002) and less than 5% in the second growing season

(Fritschi et al. 2005; Ladha et al. 2005). The NUE of California grain production systems are within this range, or even slightly higher. Recently developed management practices for rice show capacity to increase NUE even further, to >60% (Linquist et al. 2009). By comparison to field crops, fruits and vegetables tend to have lower NUE (Table 3.1). This is important because trends in crop mix show a shift to high-value horticultural commodities that are typically more technically N-inefficient than the crops they replace. Practically every high value horticultural commodity averages zero-N and ^{15}N below 50%, some far below this value. This is significant because recovery of N is significantly lower in farmer fields than the controlled conditions NUE research is typically conducted under. Depending on the crops, low NUE may be attributed to the sensitivity of the crop to N limitation, physiological limitations, scale of production systems, poor knowledge of N demand, or application of N "insurance" against annual fluctuations in crop demand.

Partial nutrient balance (PNB) is one way to measure NUE. A PNB is equal to the amount of nutrient, in this case N, in the material exported off the field divided by the amount of nutrient applied (Dobermann 2007; Snyder and Bruulsema 2007). Given that PNB specifies an input-output ratio, a value near to one denotes a system where applications equal removal, a system in equilibrium. For PNB, greater than one indicates nutrient mining of soil resources; less than one, surplus either builds up in soils or is lost to the environment (benign or otherwise). PNB interpretation relies on the assumption that soil N pool is in a steady-state. That is, the amount of N mineralized from organic matter is equal to the amount immobilized, a zero-sum. Conditional on the field location and management, the assumption of steady-state may be violated, especially when considering short-term dynamics (Lund 1982). In long-term experiments though, the assumption may be more reasonable given that soil N concentrations are not changing rapidly in California's croplands. Resampling previously sampled agricultural soils throughout California 50 to 60 years later indicate an average increase in N of 0.20% (0.09% to 0.29%) or about 0.0036% annum^{-1} (Singer 2001). The advantage PNB presents compared to other measures of NUE is that it can be calculated *post hoc* with data often available. It therefore can provide decision makers a metric to evaluate the performance of fertility programs and a tool to evaluate changes in NUE over time at field-scale, even when NUE was not the original goal of the data collection. We first estimated PNB from data found in fertilizer response trials in California (Table 3.1). Mean PNB rarely neared one, even for tightly controlled experiments, when analyzing N application rates that reflect those used in the field. These findings suggest ample room for improvement. However, it is again true that many of the studies are dated and may not reflect the sophistication of modern production or more importantly the yield levels.

The CNA also calculated PNB for 33 crops based on average yield (USDA 2011a), N application rates (Rauschkolb and Mikkleson 1978; Appendix 3.1), moisture and N content[9] (USDA 2011b) for 1973 and 2005 to examine historical trends in NUE. Results suggest California cropping systems have become more N efficient over the 33-year period, with PNB increasing 37% on average. This was expected, as the rate of average yield increases (>50%) far outpaced that of N application rates (25%) (Figure 3.3). Similar to N application rates, crops differ significantly in the magnitude and direction of their trend. An area-weighted PNB for 2005 suggests that an amount of N equivalent to 54% of statewide sales could be accounted for in crop products and byproducts exported from the field, well below the sustainable threshold of ~1. Assuming the PNB values are representative of California cropland as a whole, this statewide PNB suggests there was a surplus of almost 310,000 Mg of N sold (and presumably applied) in 2005. Though the estimate of surplus is striking, it is worth reiterating here that it is impossible to reduce the amount of surplus N to zero and sustain high yielding agriculture.

[9] Moisture and N content of harvested products can vary and utilizing average values introduces some error into this analysis. Of the two, N content varies to a greater percentage. However, it has a smaller effect on overall PNB because the value is multiplied by the mass of the dry product, which is a small fraction of original yields. Thus, it is our opinion that PNB derived from this analysis are robust to identify trends and are reasonable approximations for absolute values.

3.2.3.2 NUE when using organic fertilizer

Management of organic N is complex by comparison to inorganic sources. Organic N is bound within the soil organic matter and is not immediately plant available. It must first be mineralized into plant available forms, NH_4^+ and NO_3^-. The rate at which mineralization occurs depends on the origin of the material, N concentration, and environmental conditions (e.g., temperature and water), especially with respect to its resistance to microbial breakdown. Variable and uncontrollable rate of N release coupled with the fact that organic N generally must be applied (in the case of manure) or incorporated into the soil (in the case of cover crops) prior to production makes timing soil N supply with plant N demand difficult (Pang, Letey, and Wu 1997). Further, because only a part of the N in organic material mineralizes in a year (e.g., <10% from manures; (Hartz, Mitchell, and Giannini 2000), producers using organic N sources typically apply much more N than would be required using inorganic N fertilizers, at least until new soils carbon (C) and N equilibrium are reached (Pratt and Castellanos 1981).

High application rates and limited ability to control N release suggests that systems utilizing organic N should have a low NUE. Indeed Crews and Peoples (2005) reviewed ^{15}N recovery in legume based rotations and found that between 10% and 30% of N from legumes was harvested in subsequent plant tissue. Low NUE in systems using inorganic N fertilizers, however, may be more concerning. Biologically fixed N appears to be more readily utilized by soil biota and incorporated into organic matter increasing its retention in the rootzone (Crews and Peoples 2005). Results from an unpublished long-term experiment in California show similar low NUE in Mediterranean climates with annual crops. When calculating the difference between N inputs and N in harvested product in an organic corn-tomato rotation, only 27% of the amount of N applied was accounted for (Reed et al. 2006). Fields fertilized with liquid dairy manure have historically had low NUE. Following a series of assumptions about source and sink attribution, Harter et al. (2002) suggest that NUE (as PNB) could be approximately 50-60%, lower than field crops fertilized with inorganic fertilizer (Harter et al. 2002). Overall, the NUE of systems utilizing organic N is poorly documented. This is a function of the inaccuracy in knowing either the amount of material applied (or the N contained within it), the large variation in the rate of release, and few studies tracing radioactive N through organic systems.

3.3. Feed and manure management

Animals require dietary N and amino acids (building blocks of proteins containing N) for maintenance, growth, and production. Meeting the protein demand of California's animal population (cattle, poultry, horses, and pets) requires more than 557 Gg N year^{-1}, 75% of which is fed to dairy cattle (Chapter 4)[10]. The N needed to support California's livestock economy is 15% greater than the inorganic N used to support crop production (557 vs 466 Gg N year^{-1}) and 53% of total cropland N (1038 Gg N year^{-1})[11].

Only a fraction of the N contained in feed is converted into product – milk, meat, or eggs. What N is not converted, passes through and is excreted in manure (Kebreab et al. 2001; Powell et al. 2010). Where manure is deposited and how it is managed determines the fate of embodied N. Managed well, manure N represents a resource for farmers. Managed poorly, manure N is a serious pollution concern. Liptzin et al. (Chapter 4) estimate 416 Gg N yr^{-1} is excreted, 263 Gg N yr^{-1} (63%) of which is recycled to croplands as fertilizer. The balance is released into air or water and stored in soils.

3.3.1 Trends in California livestock production

Since 1980, there has been a considerable increase in the livestock population of California (Figure 3.6). The population of dairy cattle nearly doubled and the population of broilers tripled in only 27 years. There were more than 1.8 million dairy cattle and 266 million broilers in 2007. But not all animal

[10] Actual feed N demand for California livestock and poultry is greater than this amount because this estimate only accounts for the confined animal population. Protein requirements of grazing animals are not included.

[11] Total cropland N is the sum of N applied from inorganic and organic (manure and C-BNF) sources.

populations grew over this time period. Populations of feedlot steers, cattle and calves, and non-broiler poultry species (e.g., layers and turkeys) varied over this time frame. Depending on the species, the populations of these animals in California in 2007 were roughly equal to or slightly less than the population size in 1980.

The increasing size of the animal population has certainly catalyzed more N to be transferred into California's biosphere, though the absolute impact is not known. Additional animals require additional protein. Protein in feed crops originates from the atmosphere and is fixed either via biological (e.g., alfalfa) or industrial means (e.g., inorganic N fertilizer). A fraction originates from California and most of this as alfalfa while the majority of other dietary needs are imported. Increases in feed demand therefore can determine cropping patterns in the state (e.g., alfalfa and silage corn) and influence those in other regions.

Growth in livestock and poultry production has helped fuel California's agricultural economy and US food security. Livestock products were worth 9.8 billion USD in 2010, up 25% from 2009 (CDFA 2012) and contribute nearly 30% of annual agricultural receipts in recent years. Notably, California dairy operators produce 21% of the US milk and cream and egg producers rank 5th among states generating 6% of US total production (CDFA 2012). Receipts from the production of dairy, poultry, and cattle and calves are one reason why California both currently and historically has been the top earning agricultural state (in terms of farm receipts) in the nation for every year since 1948. California livestock hence support the rural economy and fill a vital niche in the US food system.

[Figure 3.6]

3.3.2 Dietary N, N-utilization efficiency, and N excretion

Protein nutrition has a significant impact on productivity, profitability, N utilization efficiency, and sustainability of animal production systems (Figure 3.7). Protein is critical to animal metabolism and animals consuming more protein yield more milk, meat, or eggs (K. H. Nahm 2010; Kebreab et al. 2001). Dairy cows, for example, fed a mixed ration with forages, grains, and protein supplements will generally yield more milk than a cow consuming only forages. Yields of poultry products increase in a similar fashion when fed well-balanced high-protein diets. But like fertilizer applied to the soil, the relative increase in yields declines with increasing protein consumption due to inherent biological limits of the animal. When the physiological threshold of assimilation is reached, excess protein is excreted.

[Figure 3.7]

Animals are often fed more protein than necessary to obtain the greatest possible production. This is, in part, a consequence that the economic and technical efficiencies of feeding N are not equal. That is, the feed N concentration at which the added cost of feeding another unit of protein equals the economic gain in production is usually greater than the feed N concentration where the marginal output begins to decline. For example, NRC (2001) recommends a diet containing 16.5% crude protein (CP) content for lactating dairy cows. However, milk production in some systems can be equivalent when cows are fed as little as 12% CP (Vandehaar and St-Pierre 2006). The actual amount of CP required to meet production goals will depend on genetics and husbandry techniques unique to each environment.

Improvement in analytical techniques and investment in research has allowed formulation of diets to meet animal nutritional needs of crude protein, rumen degradable/non-degradable 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. The general objective in formulating diets is to provide the necessary nutrition for the least cost, so the minimum protein constraint is typically used because 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, cottonseed, or carrot tops, may or may not increase dietary concentrations of proteins or minerals depending on the use of maximum constraints when formulating diets. The widespread feeding of by-products in California highlights another important point in formulating diets. The formulation of diets is constrained by the availability of raw materials, composition, and cost. A balance must be reached between what is scientifically plausible and practically feasible to achieve economic and environmental goals. The major obstacle in achieving a tight coupling between protein supply and animal requirements is cost and the resulting decline in farm profit.

When the protein and amino acid requirements are in balance with the animal's requirement, 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 2005). 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 3.2). 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 (K. Nahm 2002; Hristov et al. 2011).

[Table 3.2]

Diet has a profound impact on N excretion and loss. As discussed, the quantity of protein intake determines the quantity of N excreted but consumption also determines manure characteristics (e.g., form of N and moisture content). Manure composition, in turn, defines 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 2006; Xin et al. 2011). If physical and chemical 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 since organic N must be mineralized first. The greater environmental stability of organic N, by comparison to urea N, increases the feasibility of N collection and conservation, which presents advantages within the animal production facility. However, organic N is of lower utility as a fertilizer than inorganic urea and NH_3 because of the difficulty of predicting and controlling its release (section 3.2.2). A 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.

3.3.3 Manure management

3.3.3.1 Manure management within a confined animal feeding operation

From a rancher's point of view, the goal of manure management is to maintain a clean environment for the animal, reduce nuisance from odors, and improve animal health. From an environmental standpoint, manure management should try to conserve manure N until it can be recycled to cropland. Although manure treatment presents many pathways for N loss, and some emissions are inevitable, the primary loss pathway 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 (Committee of Experts on Dairy Manure Management 2005) and 4 to 70% in poultry houses worldwide (Rotz 2004) is emitted as NH_3. These wide ranges reflect the large impact of management and environmental conditions on emissions. Leaching of NO_3^- to groundwater may also be a concern from concentrated facilities (Cassel et al. 2005).

Significantly elevated soil NO_3^- levels have been found under a dairy corral in Southern California (Chang, Adriano, and Pratt 1973), but the evidence of N accumulation under feedlots and corrals from elsewhere is mixed. Regardless, manure contains 416 Gg N year^{-1}; of which, only 263 are estimated to be applied to cropland (Chapter 4). The remainder (153 Gg N year^{-1}) contributes to air and water pollution, threatens downwind ecosystems, and represents a lost resource.

Because N is lost from multiple components of the manure management train, it 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 conservation throughout the system.

Manure management practices and systems 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 3.8). 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 (Committee of Experts on Dairy Manure Management 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). Modifications of manure management processes can only be made within the context of the facilities structure unless wholesale shifts to new facility designs are adopted. Such transformative changes are typically cost prohibitive within the current dairy economic conditions.

[Figure 3.8]

Until recently, manure management decisions on many California dairies were made independent of N conservation or utilization. Yet, manure handling practices significantly change the form and concentration of N in manure and, therefore, it is imperative to understand unintended consequences of changes in practices. Four surveys documenting California manure management practices have been published, but differences in the geographic extent and questions asked among the surveys make comparisons tenuous (Mellano and Morse Meyer 1996; D. M. Meyer, Garnett, and Guthrie 1997; D. Meyer et al. 2011). Nevertheless, it appears dairy operators are adopting practices that increase ranchers' ability to manage N (Table 3.3). 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 (San Joaquin Valley Dairy Manure Technology Feasibility Assessment Panel 2005). In the most recent survey, more than 95% of respondents now use lagoons to store liquid manure (D. Meyer et al. 2011), helping to provide greater flexibility on when to apply manure. As discussed, changes in only a single component of a complex interdependent system are virtually irrelevant. Many nuances of manure management that potentially alter N dynamics on a dairy facility are not covered in the surveys (e.g., frequency of collection), greatly limiting the ability with much precision to determine how modifications of manure management schemes have affected California N cycling.

[Table 3.3]

Manure management in poultry operations is considered to be more uniform than the dairy industry. In confined poultry production facilities, 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 are habitually in a state of transition as managers seek to make improvements 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. The current regulatory trajectory will likely lead to more of the same and many facilities will be faced with adopting new (often costly) manure management techniques. The question becomes whether the recent perfect storm of events (low milk prices, increasing herd size, and higher costs of compliance) will force producers out of the market.

3.3.3.2 Manure management for grazing animals

Cattle and calves feed on natural lands and irrigated pastures before entering feedlots for fattening and finishing, being shipped out of state, or entering into the dairy supply chain. Grazing lands can be found in almost every part of the state. Depending on season of the year, the foothills of the Sierra Nevada, the Intermountain Region, North Coast and Central Valley are common grazing lands, with animals being transported among them. Historically pastures were fertilized with approximately 88 kg N ha^{-1} (Rauschkolb and Mikkelsen 1978) to increase productivity. Today, fertilization of pasture is rare. A more common practice for improving feed quality and protein content of pastures is to plant leguminous species, specifically clover.

Nitrogen use efficiency of grazing cattle is generally lower than that of confined animals (Powell et al. 2010). Lower NUE results from the inability of operators to assess the CP content of pastures and make adjustments to achieve the dietary balance to increase efficiency. Consequently, an even higher rate of N is excreted in manure per unit of weight gain or product than in confined systems.

Manure excreted on pasture is not collected or stored. The distribution of deposition has a significant influence over the manure N fate. On pasture, urine and feces are deposited in heterogeneous patterns creating small hotspots of N addition. Depending on microbial activity, hoof action, soil type, plant species composition, topography and climate, the N may be incorporated into plant roots, adsorbed to soil particles, lost atmospherically, leached, or runoff (Oenema and Tamminga 2005; Mosier et al. 1998; Liebig et al. 2009). 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 and utilize the deposited manure N. Assuming appropriate stocking densities and pasture management, manure deposited in grazing systems to be relatively N environmentally neutral (Tate et al. 2005).

3.3.4 Whole farm N balances

Livestock production systems are complex operations with multiple co-dependent unit processes taking place simultaneously. Opportunities for N loss during manure processing abound. Because of interactions between treatment processes, N sustainability for livestock production systems is best assessed at the scale of the whole farm, instead of individual system components. N inputs at the farm scale include feed N and sometimes bedding materials contain N (Figure 3.8). N is exported in milk, meat, and eggs and manure (when it is transported off-site). Manure applied to croplands associated with the farm does not factor into the calculations since it is generated and applied on farm. The balance of inputs and outputs then provides a simple but imperfect tool to assess N sustainability of a particular farm.

Reviews of dairy production systems show significant N imbalances at the whole farm scale (Powell et al. 2010; Castillo 2009). European dairy farms yield between 16% and 56% of the N imported in feeds and US dairy farms between 16% and 41%. On 41 dairy farms in the Western US, an average yield of 36% of N was found. N that is not exported in agricultural products (e.g., 64% of imports in Western dairy farms) is volatilized to the atmosphere, leached to groundwater, or stored (temporarily) in soils under cropland and corrals. Whole farm N balances for California livestock production systems are poorly constrained and it is not possible to draw conclusions about their relative environmental performance. However, assuming California systems are within the range of US and European systems, these results suggest significant room for improvement in manure management. Decreasing N imports and increasing N exports would help relieve pressure on the surrounding environment. Strategies that enhance N use such as staged feeding and surge irrigation of manure on croplands are available (Chapter 7). Potential negative consequences of these practices on farm profitability are an obstacle to their adoption in practice.

3.4 Fossil fuel combustion

Fossil fuel combustion during transportation and industrial activities releases reactive N compounds, NO_x and NH_3, into the atmosphere. NO_x is produced in two principal ways[12]. One, "thermal NO_x" is created by the reaction of N and oxygen in air at high temperatures. Relative temperature and the length of time N is at high temperature regulate the rate of NO_x production. Two, "fuel NO_x" results

[12] There is third category of NO_x production called 'prompt NO_x'. It includes all NO_x produced that cannot be explained by either of the other two categories. It generally accounts for insignificant amounts by comparison to the other two mechanisms.

when N contained within fossil fuels, in particular certain oil and coal, is converted to NO_x during combustion. Biogenic processes that occur in soils can also produce NO_x; however, in California, 89% of NO_x, a total input of 359 Gg N $year^{-1}$, result from fuel combustion making it the dominant driver of atmospheric concentration of this gas by far (Chapter 4).

Technologies used to control NO_x emissions sometimes unintentionally cause the release of NH_3. Instead of reducing NO_x to the environmentally benign N_2, catalytic converters can reduce NO_x to NH_3 when the air: fuel ratio is high, a common occurrence during acceleration (Kean et al. 2000; Baum et al. 2001). NH_3 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 NH_3 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.

Once airborne, NO_x and NH_3 travel short and long distances. NO_x can be transported from 0.010 to 1000s of km while NH_3 usually deposits back on land after short distances. One estimate indicates that nearly half of the NO_x and NH_3 produced in Los Angeles lands outside the South Coast Air Basin (Russell et al. 1993). Environmental conditions controlling the atmospheric chemistry and transport of N dictate when and where the N will land. Transport of airborne N compounds away from the source of emissions make combustion derived N an issue of concern beyond the location of initial emission (Ying and Kleeman 2009) and means there is a distinct spatial dimension to atmospheric N pollution (Zhu et al. 2002; Hu et al. 2009; Durant et al. 2010; Karner et al. 2010.

N emissions from fossil fuel combustion are an important source of air pollution and contribute to a multitude of human health concerns (Chapter 5). NO_x reacts with other pollutants in the presence of sunlight to form tropospheric (ground-level) ozone. Atmospheric NH_3 is an ingredient of particulate matter (PM), specifically particles of ammonium nitrate. Creation of N derived PM depends on having sufficient levels of NO_x and NH_3 in the atmosphere, meaning in certain airsheds, PM reactions are NO_x limited (e.g., Southern San Joaquin Valley) and in others NH_3 is limiting (e.g., South Coast).

3.4.1 Transportation

The terms ‘transportation’ and ‘mobile sources’ are not perfectly synonymous. Mobile sources include a wide range of on- and off-road activities of which transportation is a part. Vehicles used in the transportation of humans and goods (passenger cars, light and heavy duty trucks, etc.) dominate the atmospheric NO_x emissions inventory. The cumulative consequence of transportation sources far outweighs the impact of less common mobile sources such as lawnmowers and off-road recreational vehicles, despite higher emissions per quantity of fuel from these other sources. Their significance to the state’s atmospheric N balance justifies focusing the discussion on transportation sources.

3.4.1.1 Temporal and spatial trends

Because N emissions from fuel combustion are somewhat correlated to fuel consumption, fuel sales data provide a starting point to understand emissions from the transportation sector. According to the California Board of Equalization (2009), annual sales of gasoline increased 77% from 8,940 to 15,807 million L and sales of diesel increased by 430% from 0.6 to 3.1 million L between fiscal years ending in 1970 and 2007 (Board of Equalization 2010). The average annual rate of change over the same time period was 2 and 5% per annum for gasoline and diesel, respectively. Sales trends demonstrate that there have been massive historical increases in the consumption of fuel for transportation in recent years, which has undoubtedly heightened the risk for additional atmospheric N loading. The threat is unlikely to be abated anytime soon. A recent projection suggests gasoline consumption in 2030 will be 54% higher than 2008 (Caltrans 2009).

Increased fuel sales have been in part catalyzed by growth of the vehicle fleet and distance traveled per vehicle (Figure 3.9). For example, the vehicle population in California increased 109% from 12.1 to 25.4 million vehicles between 1980 and 2007. The number of light duty trucks on the road increased 212% (from 1.7 to 5.3 million vehicles), medium-heavy duty truck population more than doubled (111%) to 0.24 million, and the number of passenger vehicles increased 68% from 7.6 to 12.8 million vehicles. In addition, vehicles were traveling further distances. In 2007, total vehicle km traveled equaled 1.49 billion km (CARB 2012). That distance was a 129% increase from 0.65 billion km in the 27 years since 1980. The most significant growth in distance traveled was for medium-heavy trucks (109% to 0.014 million) versus 77% for passenger vehicles. This trend contrasts sharply with the comparison between the populations of these two vehicle classes. Of the two, passenger vehicle population increased more rapidly over this time frame. 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; Corbett et al. 1999). Recently, ocean-going vessels have received increased attention because as much as 70% of emissions takes place near ports (Corbett et al. 1999).

[Figure 3.9]

Transportation activities have historically been and continue to be the driving force in combustion derived NO_x emissions (CARB 2010; Cal EPA 2013b). Eighty-six percent of NO_x in 2008 was derived from on and off-road mobile sources statewide (Cal EPA 2013b). However, the relative significance of the various vehicle classes is changing. 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 3.10). Emissions from heavy-duty diesel vehicles are now the largest source of NO_x in the state. Interstate trucks accounts for 14 to 17% of the truck population and 28 to 29% of the distance traveled (Lutsey et al. 2008). This represents a departure from previous trends (Figure 3.10). As little as 16 years ago, NO_x emissions resulted mostly from passenger vehicles. The change in the relative significance of

NO_x sources can be traced to aggressive technology forcing regulations on passenger vehicles and more lax policy for diesel engines (Sawyer et al. 2000). Rules to regulate emissions from the latter sources are currently under various stages of development and implementation with CARB.

[Figure 3.10]

3.4.1.2 Technological change

Source activity[13] alone, however, does not determine N gas production. Emissions are the product of the activity level such as number of cold starts or distance driven and the technology such as catalytic converters or fuel being employed. These factors interact in dynamic ways to create (and control) emissions. Traffic conditions, the age of the vehicle, and gasoline composition significantly affect the context of combustion and hence the total amount and relative proportions of compounds in the emissions profile (Bishop et al. 2010). Technological change is the reason NO_x emissions in California have been declining, despite significant increases in vehicle population and total distance traveled. Changes in engine performance have offset the impact of the transportation sector.

Major components of vehicle design, comprised of vehicle type, engine, and fuel combinations, can be thought of as an integrated system that together affect the risk of emissions and constrain mitigation options. The type of technology in use is largely determined by the fuel and vehicle type. Technological changes for light-duty vehicles that run on gasoline have been the most radical. Utilization of positive crankcase ventilation systems, exhaust gas recirculation systems, and three-way catalytic converters have all helped control NO_x pollution. More recently, computer-controlled fuel injection systems and on-board diagnostic systems provide the engine information that helps it maintain the appropriate stoichiometric air-to-fuel point for the catalysts that convert NO_x to N_2 to function properly. In addition to engine refinements, fuels have been reformulated to enhance engine modifications. Low

[13] Activity refers to human actions that cause emissions such as driving.

sulfur concentrations—which are standard in California now—are a common feature of reformulated gasoline. Use of low sulfur gasoline is significant because sulfur ruins catalysts' effectiveness.

Technological changes for other vehicle/fuel combinations (e.g., medium- and heavy- duty vehicles) have been less extensive. For the most part, however, control technologies are similar for other gasoline-powered vehicles. Some modifications have occurred though. Diesel engines have changed combustion chamber design, operate at lower engine speeds, and use electronic control for improved timing, amongst other improvement to reduce NO_x. Opportunities are now also available to use exhaust gas recirculation systems and particle traps to reduce NO_x and primary PM emissions.

The importance of control technology and technological change to control N emissions from fossil fuel combustion cannot be overstated. Typically only 10% of the fleet is responsible for the majority of emissions, meaning there are a small number of high polluting vehicles on the road. High polluting vehicles are generally, but not always, older. Age of the vehicle is important because it defines the technology in use and often the condition of the technology. Catalysts and other control technologies deteriorate over the lifespan of a vehicle. It is for this reason that fleet turnover and renewal has been critical to past gains and will continue to underscore future N emission reductions from this source.

3.4.2 Energy and industry (stationary sources)

Stationary sources of N emissions include any non-mobile sources. In California, major stationary source categories include: boilers, steam generators and process heaters, utility boilers, gas turbines, internal combustion engines, cement kilns, glass melting furnaces, waste combustion, residential water heaters, and residential space heaters. Stationary sources were only responsible for approximately 11% of NO_x emissions in 2008 in aggregate (CEPAM 2009). Of this, fuel combustion contributes 71%, or roughly 8% of the total NO_x inventory—335 Mg. Fuel combustion by stationary sources is therefore a relatively insignificant driver of N cycling in California today. That was not always the case, however. In 1980, stationary source fuel combustion contributed >21% of the state's NO_x, 954 tonnes. In the 20 years between 1987 and 2007, emissions were cut by nearly two-thirds. Reductions occurred despite the number of stationary sources producing NO_x, increasing from 3,437 to 9,296, a 170% increase, over that timeframe (CARB 2010).

Emissions reductions can be, in part, attributed to the fact that stationary sources are point sources. Though there are a large number of individual NO_x producers (e.g., >9,000 in 2007), the vast majority contribute very small fractions, if any, to the total. Eighty percent of emissions were derived from 152 and 187 facilities in 1987 and 2007, respectively. The skewed distribution towards a relatively few sources improves the ability for targeted response and increases efficiency of source control actions. Compared to agricultural facilities (>80,000 farms), the number of significant stationary sources is virtually zero.

Remedial actions have been further enhanced by development and uptake of control technology. There is some evidence that technological advances that reduce N emission are becoming more prevalent (Kirschstetter et al. 1999; Yeh et al. 2005). Emissions reductions generally are the consequence of either modifying combustion conditions or capturing gases prior to release. Popp (2010) examines 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. Presumably, post combustion technology adoption has been slower due to implementation and operational costs. Power plants in California are not typically coal-fired. However, California energy demands 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, such as SCR, has occurred. More than 60%

of the energy generated with fuel-fired gas turbines in the state apply post-combustion controls (EPA 2004).

The example of California power plants illustrates an important concern; the potential for pollution leakage. Leakage refers to shifting the pollution burden from one entity to another, be it a location or environmental system. In this case, stringent regulatory controls coupled with high market demand have created a system where California needs are accommodated at the detriment of other places. Whereas we illustrate leakage here, the potential for pollution swapping underscores essentially any change in a direct driver.

3.5 Industrial processes

Nitrogen is used for a variety of industrial purposes. Globally, industrial uses account for 18% of synthesized NH_3 (Yara 2009; Yara 2012). In the US, estimates of non-fertilizer use range from 12 to 28% of the total consumption (produced or imported), depending on the year and the data source (Chapter 4, Table 4.18). A recent estimate indicates that non-fertilizer N use accounted for 14% of total US NH_3 consumption in 2010 (USDI 2012).

NH_3 fixed via the Haber-Bosch process is the starting point for N-based chemicals. The resulting NH_3 can be used as a raw material in industrial systems itself or further processed into a series of ingredients—nitric acid, ammonium nitrate, or urea (Appendix 3.5). NH_3 is primarily used in the production of ammonium salts—ammonium phosphates, ammonium nitrate, and ammonium sulfate. Ammonium salts are common fertilizers and by comparison, have relatively few industrial uses. NH_3 does however have a role in the production of certain chemicals, especially melamine and caprolactam, which are important in the production of nylon and plastics. NH_3 can also be used to remove air toxins and reduce pollutant loads of exhaust gases from point sources burning fossil fuels.

Many industrial N uses rely on intermediate N products such as nitric acid and urea. Conversion of NH_3 to nitric acid occurs via the Ostwald process (under high temperature and pressure). Nitric acid is most commonly known for its use in making explosives—e.g., ammonium nitrate and nitroglycerine. California consumed an average of about 35,000 Mg of industrial explosives and blasting agents a year (1994-2009). Nitric acid can also be used in producing primary metals, including as an extracting agent for copper and gold from their ores. Production of nitric acid is now one of the top three most common non-fertilizer uses of N in the US (USDI 2012).

Industrial N use is arguably the most poorly characterized, monitored, and understood parts of the N cycle. This is significant because the demand for industrial NH_3 is projected to increase. Forecasts estimate that global demand will increase by 21% between 2007 and 2013 alone (IFA 2010). Market expansion will partly result from an increased demand for N-containing products and discoveries of new uses.

Just how significant a driver industrial N use is to the overall N cycle of California is difficult to determine. Few statistics are kept at subnational levels. One approach to examine its potential leverage is to estimate the size of the flow based on per capita consumption. Estimates suggest per capita consumption in the US ranges between 2 and 9 kg N capita^{-1} year^{-1}, not including N used for explosives (Domene and Ayres 2001). Though there is a greater than four-fold difference between the minimum and maximum, the higher end of the range may be more probable for today's consumption patterns as similar levels have been found for Western Europe (Stoumann et al. 2011). Assuming 7.5 kg N capita^{-1}, industrial N use would be responsible for transferring approximately 283 Gg N year^{-1} into California. That suggests the industrial N use would be responsible for a transfer of N into California equivalent to more than half that of inorganic N fertilizer applications.

Industrial N use is not environmentally benign, as industrial processes can be a significant source of emissions directly and over the lifespan of the materials created. Emissions from chemical processes may end up in either air or water depending on the product. Nitric acid production released 3% of US N_2O in 1996 (Domene and Ayres 2001). Explosives release most of the embodied N as N_2 but a fraction is NO_x and N_2O. Industrial N end-products also tend to accumulate in high-density settlements, in structures or landfills. Given the longevity of many industrial N products, this pool of reactive N provides a resilient N legacy that releases N slowly into the environment. Where it is concentrated, industrial N may pose considerable long-term environmental and human health concerns.

3.6 Wastewater management

Human consumption concentrates N in settlements and urban areas, much of which is discarded in garbage, refuse, and human excretions creating N enriched wastes. Wastes are then collected, processed, and discarded as part of the municipal solid waste or wastewater stream (Appendix 3.6). Spent water, in particular, contains a substantial latent pool of N due to its constituent mass of feces, urine, industrial waste, and byproducts of food preparation. In California, the size of the excrement-derived wastewater N flow is approximately 174 Gg N yr^{-1} (Chapter 4). Whereas this wastewater represents only a relatively moderate sized flow of N by comparison to others (e.g., fertilizer or fuel combustion), its importance is partially derived from the fact that N removal was not a historical goal of treatment. Consequently, it was discharged directly into receiving ecosystems. Discharge to the ocean is the most common fate of wastewater N in California, with smaller amounts ending up in biosolids, emitted as gases during treatment, applied to soils, or discharged to surface waters (Chapter 4).

Irrespective of the ultimate receptacle receiving the wastewater (freshwater, land, or marine), wastewater N presents environmental concerns. The potential for N to pollute marine systems is well known (e.g., the hypoxic zones in the Gulf of Mexico and Chesapeake Bay), though similar impacts off of California's coast are less pronounced (see Chapter 5). But addition of even small concentrations of N into freshwater systems can often overwhelm them. Background N levels in aquatic systems are typically quite low. Any addition can disrupt the functioning of food webs and ecosystem health Discharges to land are no more environmentally friendly. N not denitrified by natural soil attenuation processes elevates soil N content and increases leaching potential. Lund and colleagues (1976) investigate inorganic N concentrations below sludge ponds and found elevated NO_3^{-}N and NH_4-N levels at multiple depths below the wastes by comparison to control areas indicating downward percolation of N from waste. Therefore, although the results from a study performed nearly 40 years ago may no longer be accurate, this suggests that wastewater discharge can cause acute pollution.

Technologies to remove N from wastewater are available, however. Wastewater treatment takes place in either of two ways. In California, it is typically processed at a centralized, regional wastewater treatment plant (also known as a publicly owned treatment works). Or when sewage systems are not available to collect and convey the material to a centralized location, wastewater can be treated with onsite wastewater treatment systems (sometimes referred to as septic systems). It is necessary to mention that N removal from wastewater is a time, energy, and money intensive process. Discussion of the extent of wastewater treatment must consider the social, economic, and environmental context in concert (Muga and Mihelcic 2008). Chapter 7 of this volume further discusses wastewater treatment options.

3.6.1 Publicly owned treatment works (POTWs)

Centralized treatment plants process about 90% of human wastewater generated in California (Chapter 4). The amount of wastewater treated at each plant is relative to the size of the population it serves, with a typical value around 379 L capita-day^{-1} depending on the degree of water conservation. Wastewater contains about 13.3 g N capita-day^{-1} (Metcalf and Eddy 2003). Based on this estimate and the 2010 population of 37.25 million, POTWs process 180 Gg N $year^{-1}$. This estimate is 10% higher than that found in Chapter 4 in part because of population growth between 2005 and 2010.

3.6.1.1 Wastewater treatment

When considering the effects of wastewater on N cycling, it is useful to start with collection systems. For a majority of the population in California, wastewater and raw sewage are transported through a system of pipes and pumps to a municipal POTW. For a variety of reasons, including cost, most conveyance systems are not maintained adequately. Aging infrastructure, poorly fitted pipes, and seasonally high flow can cause wastewater collection networks to leak through overflow and seepage during transit.

Sewage systems overflows (SSO) and sewage exfiltration (leakage) cause wastewater to escape into the surrounding soil and potentially reach surface waters or leach into groundwater (Wakida and Lerner 2005). Between 1970 and 2011, there were 11,084 SSO incidents reported throughout California (Cal EPA 2013c). Only 10% of the sewage was recovered and 84% or approximately 141 million L reached surface waters (Cal EPA 2013c). Overflows are most significant when the untreated wastewater enters sensitive water systems, which can impact aquatic systems and potable water supply. Common causes of SSO are infiltration and inflow of stormwater, and blockages by grease, debris, or plant material. Sewage exfiltration is more difficult to identify or quantify because it tends to occur below ground. Work suggests that leakage may range anywhere from 1 and 25% of N transport (Viers et al., n.d.).

Once sewage reaches the POTW, it 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 3.4). Broadly, the technologies can be grouped into primary, secondary, and tertiary treatment[14]. During primary treatment, a portion of the floating and settleable solids are removed through screening and/or sedimentation in clarifiers. Secondary treatment converts 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, in accordance with their NPDES permit, many large wastewater treatment plants do not remove nitrogen and instead control the treatment process to prevent nitrification, resulting in high effluent ammonium concentrations. To remove N during secondary treatment, an 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 is referred to as biological nutrient removal (BNR). 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 materials following secondary treatment using filtration. Tertiary treatment is most often performed to meet regulatory requirements for water reuse projects and does not have a significant impact on effluent N content.

[Table 3.4]

It is important to remember that nitrification-denitrification transform a significant portion of wastewater N into N_2 and other nitrogenous gases. N_2 gas is the overwhelming end product of these processes, with more than 90% of the N being volatilized in this form. However, N_2O is produced as a byproduct of incomplete conversion by denitrifying microbes. Consequently, utilizing nitrification/denitrification increases emissions of this climate forcing gas while achieving the goal of reducing the N load in wastewater. A recent study of wastewater treatment in California shows that treatment for N removal increases N_2O production from ~0.5% of the N in influent to as much as 2%.

[14] For a thorough description of wastewater treatment processes and their effect on N removal see Tchobanoglous et al. 2014.

However, the authors also recommend making comparisons to N_2O emissions from high-N wastewater subject to primary and secondary treatment only, which was not available in this study (Townsend-Small et al. 2010).

The amount of N in effluent discharged from POTW depends on the level of treatment, be it primary, secondary, or tertiary and the conditions of biochemical controls. The efficacy of N removal in wastewater treatment processes is related to the availability of carbon, temperature, alkalinity, use of anoxic zones, solid retention time, dissolved oxygen, and hydraulic retention time (US EPA 2008). By using advanced secondary treatment, effluent levels can be well below 10 or even 2 mg/L NO_3-N.

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 5,110 million L of effluent each day are discharged directly into the ocean (Hauser et al. 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” (Asano et al. 2007). It is anticipated that pressure to improve effluent water quality will result in greater implementation of wastewater denitrification systems.

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 use 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 (CASA 2009). These reports also suggest the use of biosolids is changing. In 1988, 60% of biosolids were sent to landfills, 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, and thereby could also reduce the need for synthetic fertilizer, which requires fossil fuels to produce.

3.6.1.2 Trends in wastewater N and treatment

The concentration of N in wastewater is predictable and correlated with the size of the population. The population census can therefore be used as a reasonable proxy for wastewater N, greatly enhancing our knowledge of trends in wastewater N impacts. According to the 2010 Census, California is now home to more than 37 million people. Much of the growth has occurred since the middle of the last century. Ten million people lived in California in 1950, up from less than 2 million in 1900. By 2020, California's population is estimated to reach 42 to 48 million. Assuming direct proportionality and a constant percentage of persons serviced by POTWs, the quantity of wastewater N produced in California has increased more than two-fold over 60 years. Over this period, diets have been changing, which affects N concentrations in wastewater, and population growth has largely resulted in more developed areas, which are usually connected to centralized treatment systems. The increase in population and increased protein consumption suggest that the estimate of a tripling of wastewater N processed by POTWs since 1950 is likely conservative.

Reports suggest 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 3.4). As described in the 2007-2008 report (SWRCB 2008), nearly 80% of processed wastewater receives at least secondary treatment and 50% of the total flow potentially receives advanced secondary and tertiary treatment.

Though the trend seems to indicate enhanced N removal, it is a challenge to estimate the true impact of wastewater management on N at POTWs. Facilities report the levels at which they have the capacity to treat wastewater and the amount of flow they are capable of treating. Neither the proportion of wastewater nor the extent to which it is treated are reported. It can be assumed that N removal will occur to below the minimum necessary to be in compliance with discharge requirements. Furthermore, standard N removal relies on the biological mediated process of nitrification and denitrification, processes very sensitive to environmental conditions—e.g., carbon and oxygen availability and temperature. Because of fluctuating condition through time, wastewater processed with the same unit process at the same facility will have variable effluent N concentrations.

3.6.2 Onsite wastewater treatment systems (OWTS)

Developments in remote areas and some industrial sites cannot be connected economically to sanitary sewer infrastructure. These facilities utilize OWTS, sometimes referred to as septic systems to treat wastewater prior to discharge[15]. Between 1970 and 1990, the percentage of California's population using OWTS declined from 12.2% to 9.8% (USDC 1940-1990) . 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 OWTS to treat wastewater and about 12,000 new OWTS are set-up each year(SWRCB (State Water Resources Control Board) 2015; Leverenz, Tchobanoglous, and Darby 2002).

Historically, a septic tank provided the only treatment prior to land application from OWTS, usually by subsurface infiltration. Because only a small fraction of wastewater N accumulates in the

[15] The term septic system is used because of the widespread use of the septic tank for low-maintenance primary solids removal.

sludge in septic tanks, the effectiveness of the system for the treatment of N is dependent largely on the 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.

At 10% of California's wastewater, OWTS have only limited ability to impact total N cycle in California (e.g., 17.4 Gg N year^{-1}, Chapter 4). In situations where OWTS systems function improperly, sewage discharges N directly into the surrounding environment. OWTS N in these areas may be a threat to local resources (Broehm et al. 2009; Walters et al. 2011).In 2001, a survey of 47 California jurisdictions with 912,949 individual sewage systems issued 4,831 repair permits, a median of 0.5% of the operating systems (CSWRCB and EPA 2003).

Modern onsite systems have been engineered to utilize the same processes used in centralized treatment systems to convert wastewater NH_4 into an inert gas, nitrification and denitrification. A variety of treatment trains for OWTS are available. Nitrification and denitrification can either be performed in conjunction in a single unit or in segregated units. In the single stage process, aerobic and anoxic decomposition take place within the same reactor. Periods of aeration alternate with periods without aeration to accomplish nitrification and denitrification. The availability of carbon (as an electron donor) is the primary limitation of N removal in single stage treatment. The effectiveness of single stage systems range between 40% and 65%, and the efficacy of N removal can reach 75% if effluent is recycled back into the reactor. In the two-stage unit, nitrification occurs in a separate location than denitrification. Moderating pH during the nitrification stage and providing an electron donor in the second stage are concerns with these systems. However, if operated properly, two stage systems achieve high levels of efficiency of N removal (60 – 95%). Theoretically, modern OWTS can achieve high levels of effluent quality, similar to that of centralized facilities, but the vast majority do not. As with POTWs, OWTS must provide the requisite environment to sustain biological treatment mechanisms. Under intermittent management and sewage flow, treatment conditions are typically not optimized. Realized N removal efficiency of advanced, well maintained systems typically are only 40 - 60%, well below efficiency of POTWs that treat for N (Leverenz et al. 2002; US EPA 2008).

Between 70 and 80% of the N in the OWTS influent is derived from human excrement (Lowe 2009). The remainder of the N mass is a function of consumer chemical and product use and food preparation. Isolating waste streams with unique characteristics facilitates tailored management of N properties of each. Source separation of wastewater is an emerging strategy in Europe for nutrient recovery from domestic sewage. However, the cost of retrofitting infrastructure, toilets and domestic pipes is a limiting factor at this time.

Because of lack of control and other challenges associated with incidental N removal in the soil, engineered N removal systems are being required in some areas. The effluent quality requirements for onsite systems are based on site specific considerations, mostly concerned with leaching and accumulation of nitrate in groundwater. 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).

3.7 Land use, land cover, and land management

Public and private entities modify land use, cover, and management practices to maximize societal and personal benefit. Each conversion implies a unique type of change to the physical characteristics of a given land parcel. Land use change refers to a shift between two different classes of use (e.g., among agricultural, natural, or development). Changes in land cover denote transformations of the surficial material (e.g., from forest to grassland). Perhaps the most common changes are those where land use and land cover change simultaneously (e.g., grasslands to agriculture). Land management, a less often discussed third category, does not necessarily change use or cover. It is included here because it typically alters the intensity of N fluxes and flows (e.g., increased N fertilizer use with more intensive agriculture or increased fuel use with exurban development).

Nitrogen cycling and emissions are directly related to land use, cover, and management. Land use, cover, and management decisions affect N dynamics in at least two ways. First, they alter the magnitude and speed of N cycling because the magnitude of N inputs, the potential for specific transformations, and the likelihood of certain N loss pathways differ considerably between the original and the derivative state (Table 3.5). The last effect is particularly significant because it means that landowner choices determine not only N dynamics on the piece of land itself but also how it interacts with the wider N cycle. For example, agricultural areas tend to be sources of NO_3^- to groundwater while urban areas tend to emit N compounds into the atmosphere. Various land uses alter the entry point of reactive N compounds into the environmental systems.

[Table 3.5]

Though N cycling within various land uses, cover, and management has long been studied (see references in sections 3.3 – 3.8), the importance of transitions among land uses, cover, and managements for N cycling and the environment has only recently become appreciated and remains poorly characterized in California. Viers et al. (2012) demonstrate the potential impact. Examining trends in land use area, crop mix, yields, and N fertilization rates since 1945, the authors' analysis indicates that the broad scale conversion of natural areas into intensive agriculture of the Tulare Lake Basin has contributed to higher NO_3^- levels in the aquifer. These estimates are consistent with the hypothesis that land use change in California has the potential to increase non-point source pollution (Charbonneau and Kondolf 1993). However, changes in land use, cover, or management do not necessarily lead to greater N loading to the environment. Between 1971 and 2001, there was a 31%

increase in effluent volume pumped into oceans in Southern California as a result of development, yet mass emissions of NH_4 decreased 18% (Lyon and Stein 2002). Improved management at large POTWs mitigated development's impact. Historic land use, cover, and management shifts have caused massive changes to N input, exports, and storage in California's landscape. The net effect is a function of factors that often interact in ways difficult to predict. Incompletely documented transitions mean most conclusions are only speculative at this time. Quantification of major land use, cover, and management activity trends is a first task in understanding the potential consequences of California's landscape in transition.

3.7.1 Developed areas

The size and density of developed areas of California have been expanding over the past forty years. Between 1973 and 2000, developed areas increased their land base by 37.5% and now account for 4.2% of California's total area (Table 3.6). Over the same time period, regions experienced a variety of development patterns. Development declined by 5.0% in the East Cascades and Foothills. In the Southern California Mountains, it increased 44.8%, near an order of magnitude difference. Population density has risen concordant with the expansion of developed areas but growth rates are variable depending on the city. The number of people per km^2 in Fresno and Redding rose by 187 and 382%, respectively between 1970 and 2010. Larger cities grew less rapidly. The Sacramento population rose 87% and South San Francisco grew 41% over the same time period (USDC 2013). Not surprisingly, data clearly show that California has become more urban and populous in the last 40 years. A question relevant for this discussion becomes, what was lost during this evolution?

Expansion of developed areas has come at the expense of agricultural and natural areas. Reconstructions from historical satellite imagery between 1973 and 2000 show that 3,884 km^2 of agricultural land, grasslands and shrubland have been developed (Sleeter et al. 2010). The relative proportion of the converted land has shifted over time. Development was largely built on top of agricultural land between 1973-1980 and 1992-2000, with 697 km^2 and 470 km^2 converted, respectively (Sleeter et al. 2010). Conversion of agricultural land to development has reached double-digit growth rates in some regions since the early 1980s (CDC 2010). During the two intervening periods, development occurred more on grasslands and shrublands than agricultural lands with 448 (1980-1986) and 1,037 (1986 – 1992) km^2 converted.

[Table 3.6]

Growth of developed areas radically modifies the N cycle. Development increases N imports from food, fertilizer, and fuels creating N hotspots. Urban expansion replaces plant cover, often agricultural or natural lands, with a built environment. Natural hydrologic and soil processes are altered or arrested. The extent of impervious surfaces and drainage increases, though the magnitude depends on the type of development—high-density, suburban, or exurban. Expansion of engineered structures results in efficient collection and conveyance of N around the landscape. N accumulated on pavement moves in stormwater runoff, trimmed grass becomes green waste, and waste discarded by human becomes sewage or trash. All eventually is deposited and stored within the urban areas (e.g., landfill) or exported beyond its boundaries (e.g., into the Pacific Ocean or local streams in California). The high concentration of N in wastes has the tendency to saturate and overwhelm the receiving environment's buffering capacity and can cause local and regional environmental contamination (Groffman et al. 2004).

[Box 3.4]

3.7.2 Agriculture

Relocation and intensification are two dominant processes shaping California agriculture in recent history. Agricultural relocation is a significant phenomenon for N cycling. It completely reengineers the

N cycles in the new location since N flows and turnover in agricultural systems are generally much larger than that in natural areas.

When faced with urban encroachment, farm operators have historically transferred their operations to new locations in new regions. 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 from the 1970s. More recently, high value horticultural crops—e.g., vineyards—have been spreading into the foothills of the Sierra Nevada or the North Coast, replacing grasslands and oak woodlands. Merelender et al. (2000) show that more than 4500 ha of grapes were planted in seven years (1990-1997) in Sonoma County alone (almost 25% of the total). Farmers often take the opportunity to change management practices by updating technology or shift to new commodities when they move (Hart 2001).

Relocation has allowed there to be only a nominal decline in the agricultural land base despite urban encroachment. 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 early 1970s (Hart 2003; Sleeter et al. 2010). The statewide balance may be deceiving, however. Some regions have lost most of their agricultural heritage to development. Others, such as the Imperial Valley, have seen considerable growth in agricultural area. Agriculture has generally moved from prime locations with high quality agricultural soils and water access to more marginal lands. According to the FMMP (2010), average annual rates of decline of “prime farmland” and “farmland of statewide importance” in their surveyed regions were 21 and 9%, respectively between 1984 and 2006. Shift in production to farmland of lesser quality may have negative but also counterintuitive effects on N cycling processes. Marginal lands typically are steeper and have thin, erodible soils, and may require more N fertilizer. The combination of these factors would likely increase the potential for N loading to the surrounding environment. Since at least 1993, this indirect consequence of agricultural relocation has been recognized in California (Charbonneau and Kondolf 1993), but this hypothesis is difficult to test. Interactions between the environment condition and the chance of compensation by management practices complicate generalization about the consequence of relocation on N, though N loss is probable without significant adjustments in management.

Conversely, agricultural intensification (a change in land management and sometimes cover) presents one of the clearest effects on N cycling. The most obvious result of agricultural intensification is increased N fertilizer use. We estimate California croplands are becoming more N intensive; an average of 25% more N fertilizer was applied per crop per acre in 2005 versus 1973 (Section 3.2.1). For the most part, this increased N use has been offset by simultaneous increases in yield (Section 3.2.3). 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 among widely cultivated species. For example, wine grapes receive an average of less than 30 kg N ha^{-1} while celery receives closer to 300 kg N ha^{-1}. Plant N uptake regularly exceeds 100 kg N ha^{-1} and can be as high as 250 kg ha^{-1}. Because of the difference in plant N demand, changes in crop mix will alter total statewide crop N use. Over the last 35 years, California's crop mix has shifted heavily from field crops that often receive less N fertilizer to more N-intensive species, e.g., vegetables and nuts. As of 2008, field crops are still grown on the majority of croplands (Figure 3.4), but the land area dedicated to field crops declined from 74 to 53% between 1970 and 2007. Fruits and vegetables are now grown on a nearly equivalent amount of land (53% versus 47%). The shift in crop production towards N intensive crops is at least partially responsible for greater N consumption in the state.

Animal production has become more intensive too, with significant implications for the N cycle. As discussed, animals require N-rich feed and excrete N-rich manures (Section 3.3). 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 3.6). 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 generally equal to or slightly less than levels in 1970. Larger populations require greater resources. Demand for animal feeds is responsible for a greater amount of N entering California's terrestrial biosphere than fertilizer used on crops and lawns, when summing N fixed by biological and synthetic means. Not only does feed production dictate N dynamics in the state (e.g., alfalfa and field corn), it influences N cycling in other regions of the US. Approximately one-third of the N fed to California animals is grown elsewhere. By changing feed demand (and increasing dependence on off-farm feeds), animal production in California indirectly contributes N fertilizer use concerns in other regions including the Mississippi River Basin.

Larger animal populations create more N rich waste, although the relationships are not proportionate to the number of animals due to changes in N utilization efficiencies over time. The pollution concerns that increased manure creates is compounded by the fact that herd/flock sizes have grown at the same time as the total population. More intensive production concentrates manure N in a smaller area, sometimes without adequate land available for disposal. Without additional land acquisition, ranchers can find themselves in a situation of being manure N rich and land poor. Because there is uncertainty about how much land area is associated with confined animal facilities and how manure is spread, it is difficult to ascertain whether there is sufficient land available to receive the manure. Preliminary calculations suggest there is more than enough N demand in California crop production to absorb manure N (Chapter 7). This appears to be true for the entire state but in particular for Central Valley Dairy Production, a system of high concern (Pettygrove et al. 2003). Concerns about the economics of manure distribution (e.g., geography of supply and demand do not overlap) and agronomics of manure N use (e.g., temporal and N content variability) impede its use, but it is not necessarily a land availability question alone.

3.7.3 Other land uses: Forestry, wetlands, and grasslands and shrublands

Land use beside agriculture and urban areas influence the statewide N cycle. Typically, N cycling in natural lands is at a much lower magnitude than that of intensive agricultural production. However, because of the extent by which they occur, impacts aggregate to a considerable cumulative fraction of the whole N budget.

Forests, grasslands, and shrublands accumulate and emit N compounds. Many naturally occurring and exotic plants species in these areas of California have the capacity to form symbiotic relationships and biologically fix nitrogen, in much the same way as in croplands, however not all BNF results from symbiotic relationships. Free-living N fixers are also common and it is estimated that approximately 10% of statewide BNF in natural lands may result via this mechanism (Chapter 4). The actual amount of fixation, symbiotic or free-living, is sensitive to soil N availability. Hence, with increasing rates of atmospheric N deposition, N fixation in many areas may be being suppressed, lowering the total influence of this mechanism.

Simultaneous to N being added to the system, N is lost through gaseous and solution emissions. Land cover change processes in natural lands can have acute impact on N cycling. Wildfires are an important example of this in California. During combustion, N contained in the biomass and liter is released to the atmosphere (Sugihara et al. 2006). 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 stoichiometric relationships between soil C and N. Lower soil

C:N ratios that follow wildfires stimulate N mineralization causing N to be converted from organic to inorganic forms and released into the soil where it is predisposed for loss). 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 species and density. 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 severely burned is greater than 10 ha of the most intensive crop use. Wildfire intensity is also correlated with fuel load, fuel type (e.g., shrubs, litter, trees, logging slash, fallen woody material), and the vertical and horizontal continuity of fuels. Fuel loads in California have been increasing due to periodic droughts, fire suppression, and, in some cases, invasive species. Increasing annual precipitation in some areas of the central and northern California mountains may also be leading to more fine fuels growth. Together, these factors make the probability of ignition and fire spread 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, Cascades, and Klamath Mountains supports this conclusion and shows considerable increases in mean area burned since the beginning of the 1980s (Miller et al. 2009, 2011). The three years that had the largest area burned all took place in the last decade (2003, 2007, and 2008). And the trend will likely not abate. Modeling efforts agree that fire activity and intensity are likely to increase over the next 50 to 100 years (Leinihan et al. 2003, 2008; Hayhoe et al. 2004; Miller and Urban 1999). Past wildfire, however, 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 and future projections of wildfires suggests this driver has and will continue to exert pressure on air and water resources.

3.8 Universal historical increases but future uncertainty

In this chapter, we introduced the six activities and processes that drive N cycling processes in California and traced historical trends in activity levels. Data clearly show that the intensity of the activities regulating N cycling in California have increased. The consequence of universal intensification has undoubtedly been a greater perturbation of California's N cycle and more total N released in the environment, on balance. But the impacts are uneven. Certain N emissions have been tempered dramatically, despite increased use (e.g., NO_x emissions from fuel combustion). Others such a NO_3^- losses from croplands have seen contrasting trends. Despite the likelihood of continued increases in activity levels well into the future, impacts are highly uncertain. Currently on-going technological and policy discussion will undoubtedly change the trajectory of their future impact. Technological and policy responses that address critical control points of these direct drivers are discussed in Chapters 7 and 8, respectively.

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*Asterisk indicates consistently updated database.

Box 3.1. From microns to miles: The significance of 'scale' to N cycling [Navigate back to text]

'Scale' is a critical framing concept when thinking about N cycling. Depending on the context of its use, scale can refer to two ideas.

One, scale can be used as a synonym for spatial extent. This is significant for N cycling because each process that affects the turnover, transformation, and transmission of N compounds and the consequential impacts have characteristic spatial extents for which they occur, from local to global. For example, denitrification takes place at a very small, local spatial scale within the soil complex, that of microns, but a product of denitrification and a principal concern, N_2O, has global effects. Leaching is a function of local soil texture and moisture conditions. Regardless, if the rest of the field is dry, a depression or local soil fissure may be a hotspot of leaching activity. The local scale nature of N cycling processes contrasts with the more regional and global nature of N cycling concerns. The principle N issues happen at large spatial scale – kilometers - based on the aggregate of local dynamics.

Two, scale can also be thought of as a synonym for magnitude. Here it is important as a consideration for source activities and impact. As shown in the mass balance calculations (Chapter 4) and re-reported throughout this chapter, the rates at which some of the source activities take place differ considerably. When considering the inherent extent and magnitude properties of various N sources activities and impacts requires keen attention to scale issues.

Box 3.2. Brief description of N cycling in soils [Navigate back to text]

N occupies various pools in the soil, including inorganic N, microbial N, and organic N, the latter of which comprises a broad range of carbon compounds with varying susceptibility to microbial mineralization. The vast majority of N in soils, especially in natural ecosystems, is bound within soil organic matter or stored in microbial biomass, from which it is slowly released as plant available N over time and hence does not pose an immediate threat to the environment or humans. Each year, a fraction of this organic N reservoir is mineralized to NH_4^+. Mineralization serves an essential function for plants, and in agricultural systems supplies as much as 50% or more of the N assimilated by crops. Mineralized N is highly mobile and is readily transformed by soil microbes among different N species: organic N, ammonium (NH_4^+), ammonia (NH_3), nitrite (NO_2^-), and nitrate (NO_3^-). In the reverse process, immobilization, inorganic N is integrated into the living biomass of plants and microbes. The amount of organic matter returned to the soil, soil moisture, and management practices like tillage combine to affect soil microbial populations and activity and the rate of N storage or release. Adding inorganic N fertilizers can increase the total amount of N cycling through soils which can promote long-term fertilityHigh inorganic N availability may promote high plant productivity, but can also be associated with large surpluses of N. This excess N can lead to environmental degradation either by percolation through the rootzone (leaching) or through volatile emissions of N gases into the atmosphere (e.g., NH_3, nitrogen oxides (NO_x), or nitrous oxide (N_2O)). Inert dinitrogen (N_2) is the gaseous emission released in the highest quantities. Though it is difficult to measure because of the relative concentrations in ambient air, N_2 to N_2O ratios in agricultural systems are an average of 1.8:1 (Schlesinger 2008) but can be higher than 75:1.

The most important aspects that distinguish nutrient cycling in conventional agricultural soils relative to those of natural terrestrial systems are: 1) conditions of nutrient saturation 2) the decoupling of N, P, and C cycles, and 3) an inadequate synchrony and synlocation of nutrient sources and sinks (Drinkwater and Snapp 2007). The inputs of inorganic N fertilizer generally exceed the demands of plants and the soil community, a situation made worse by the decoupling of nutrient cycles, which disrupts the primary mechanisms for inorganic N immobilization and storage. When sufficient C is present, microbes are the major channel for immobilization of inorganic N in the soil. In most natural systems, microbial N far exceeds inorganic N, whereas the reverse is true in agricultural soils with low C inputs. Under steady-state conditions, the balance in rates of mineralization and immobilization, combined with the rapid turn-over of the microbial community leads to a low-level, but stable supply of N availability to plants.

Need for better understanding of natural processes

While N cycling in natural systems is understood on a gross level, more needs to be known about the fine-scale mechanisms and processes and about the relative roles of various organisms to regulate, store, and provide feedback for nutrient retention. Comparative studies of N pools and flux between them in agricultural soils versus unmanaged native grasslands and forests are instructive in how the natural processes have been altered by different soil management schemes. In addition, as a result of the doubling of reactive N globally by human activity (Vitousek et al. 1997), natural terrestrial ecosystems have experienced chronically high levels of N deposition. Studies indicate variation among ecosystems in how quickly they reach nutrient saturation, indicating differences in their capacity for N retention. In a spruce forest subject to decades of high N deposition, Kreutzer et al. (2009)

describe a dynamic system of N cycling, characterized by high rates of microbial mineralization and immobilization of N, accompanied by rapid turnover of the microbial community. This high internal flux between N pools mediated by the microbial community produces relatively high N retention while maintaining plant-available N levels sufficient to cover the entire budget of all of the trees. The potential coexistence of both high rates of ammonification and nitrification with low accumulation of ammonium and nitrate at any point in time, as demonstrated in this and other studies provides encouragement for ecomimetic agricultural fertility management.

It is also worth noting that a multitude of abiotic and biotic factors (e.g., pH, temperature, organic carbon, microbial activity, soil texture, etc.) affect the N cycle in soils over a wide range of temporal scales, from as short as minutes (e.g., gaseous NH_3 volatilization) to decades (e.g., movement of NO_3 through the vadose zone to the aquifer). Spatiotemporal heterogeneity in soil N cycling is one factor contributing to the diversity of N fertilizer use and pollution potential among fields and farms.

Box 3.3. Links between the N and hydrologic cycles

Water regulates the nitrogen cycle. For example, nitrate in soils will not move toward plant roots without water (mass flow), the extent of soil moisture alters microbial activity, N transformations, and the form of gaseous emissions (nitrification and denitrification), dissolved N in solution is transported in streams and waterways (runoff) and airborne N falls is transported to the ground with precipitation (deposition).

Given the presence or absence of water governs N dynamics through physical and biological processes, changes in the natural hydrologic cycle or management of water resources by humans, climate change, or both will have cascading effects throughout the N cycle at plot and larger spatial scales. For example, on-farm it may catalyze a shift to low-volume irrigation with the potential to reduce solution N losses at the threat of greater gaseous emission. But equally plausible is a reduction in agricultural area reducing total inputs. At watershed-levels, altering the timing or amount of precipitation may cause erratic pulse of nutrients. It is not possible to forecast the net impacts the changes in the hydrologic cycles are yet to exert on N cycling in the state at this time because of the multitude of drivers and potential responses. However, it is important to consider the significant linkages between the two global cycles when reflecting on potential future N trajectories.

Box 3.4. Cities: The definitive driver of California's nitrogen cycle in the 21st century? [Navigate back to text]

Cities and their populations significantly influence N cycling (Grimm et al. 2000; Pickett et al. 2008). Transferring food, fiber, fuel, and industrial materials from the surrounding landscape into more densely inhabited settlements causes a large influx of N to concentrate in cities. Once imported, fundamental components of the built environment – roads, buildings, waste handling facilities, and engineered drainage – have a profound impact on how N is used, processed and transported (Kaye et al. 2006; Kennedy et al. 2007). Infrastructure advertently and inadvertently changes N dynamics in cities, transforming it (e.g., wastewater treatment plants), storing it (e.g., landfills), and shifting its location (e.g., impervious surface). Which environmental system ultimately receives the previously imported urban-N and the N composition depends on policies, processing, and disposal activities (Bernhardt et al. 2008). For example, approximately two thirds of N in California wastewater is dumped into the Pacific Ocean from coastal cities while inland urban areas generally treat wastewater N due to regulations limiting land and freshwater N disposal (Section 3.6).

Understanding the impact of cities on N cycling in California is desperately needed. Consequences of N use range from freshwater pollution in drainages from lawn fertilizer to species endangerment due to wastewater discharge. A systematic examination of city-N cycling for a diverse range of cities is clearly warranted to create ideas on how to mitigate N transfers and pollution because the impacts of cities on N dynamics can be counterintuitive. For example, one might imagine that high density growth would decrease vehicle miles traveled and reduce NO_x as a result. However, the opposite seems to be true. Evidence from two California cities shows there is no relationship between urban planning and vehicle miles traveled, demonstrating a paradox (Melia et al. 2012).

Though not formally codified, the current and historical importance of California cities to the state's N cycle is apparent. Today, the vast majority of Californians live in urban areas. According to the 2010 US Census, 36.4 million people lived in urban areas in the state, more than 97% of the total population and approximately double the urban population in 1970 (USDC 2013). Thus, it stands that changes in the N cycle resulting from activities used to support the livelihoods of most Californians can be attributed directly (for example with fossil fuel emissions from the small vehicle fleet), or indirectly (as with food production), to cities. Food production, in particular, demonstrates the power of cities to affect N cycling in distant regions. A large fraction of food consumed in the state is imported from beyond the state's borders, despite the net food balance being relatively small and positive. Assuming population geography and N dynamics continues along the same trajectory as in the past 10 years (i.e., business as usual), the impact of urban areas will continue to grow. Urban population grew 10% between 2000 and 2010 (from 33.1 to 36.4 million people) while the rural population increased 6% (from 796,198 to 845,229) (USDC 2013). With an estimated population of 50 million people living in California in 2050 and almost 49 million of them living in urban areas, demand to support their everyday activities and reduce the harm of the N influx will be enormous. Indeed it may well be nearly 50% greater than apparent today.

Figure 3.1. Relative importance of the direct drivers on California's nitrogen cycle, 2005. [Navigate back to text]

Values are percentages of the total and may not add to 100% because only drivers contributing at least 5% to the total are included in charts and/or due to rounding. Colors display source-sink relationships: green-biological/agricultural and brown-industrial. N-BNF and C-BNF refer to natural lands and cultivation-induced biological nitrogen fixation, respectively. It is important to note that 'fertilizer use and soil management' for groundwater and surface water includes both inorganic and organic N sources (e.g., chemical fertilizers, C-BNF, and manures) used on croplands. Source: Chapter 4.

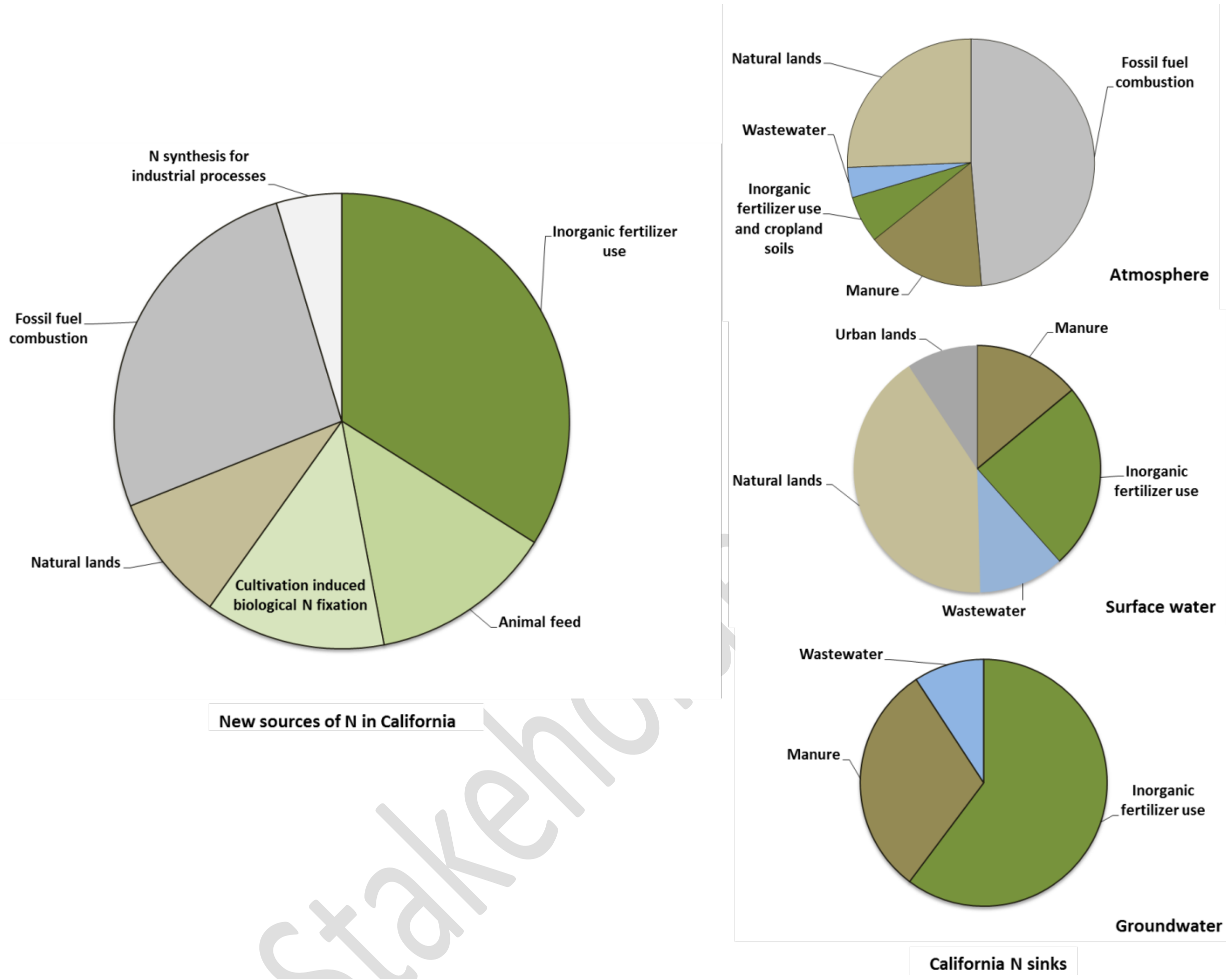


Figure 3.2. Synthetic nitrogen fertilizer sales in California, 1946-2009. Since their introduction after World War II, sales (and presumably use) of synthetic N fertilizers has increased an average of 5% per year. Yet they have largely leveled off since the early 1980s. The large rise in fertilizer sales between 2001 and 2002 calls the reliability of these data into question. Source: CDFA (2009). [Navigate back to text]

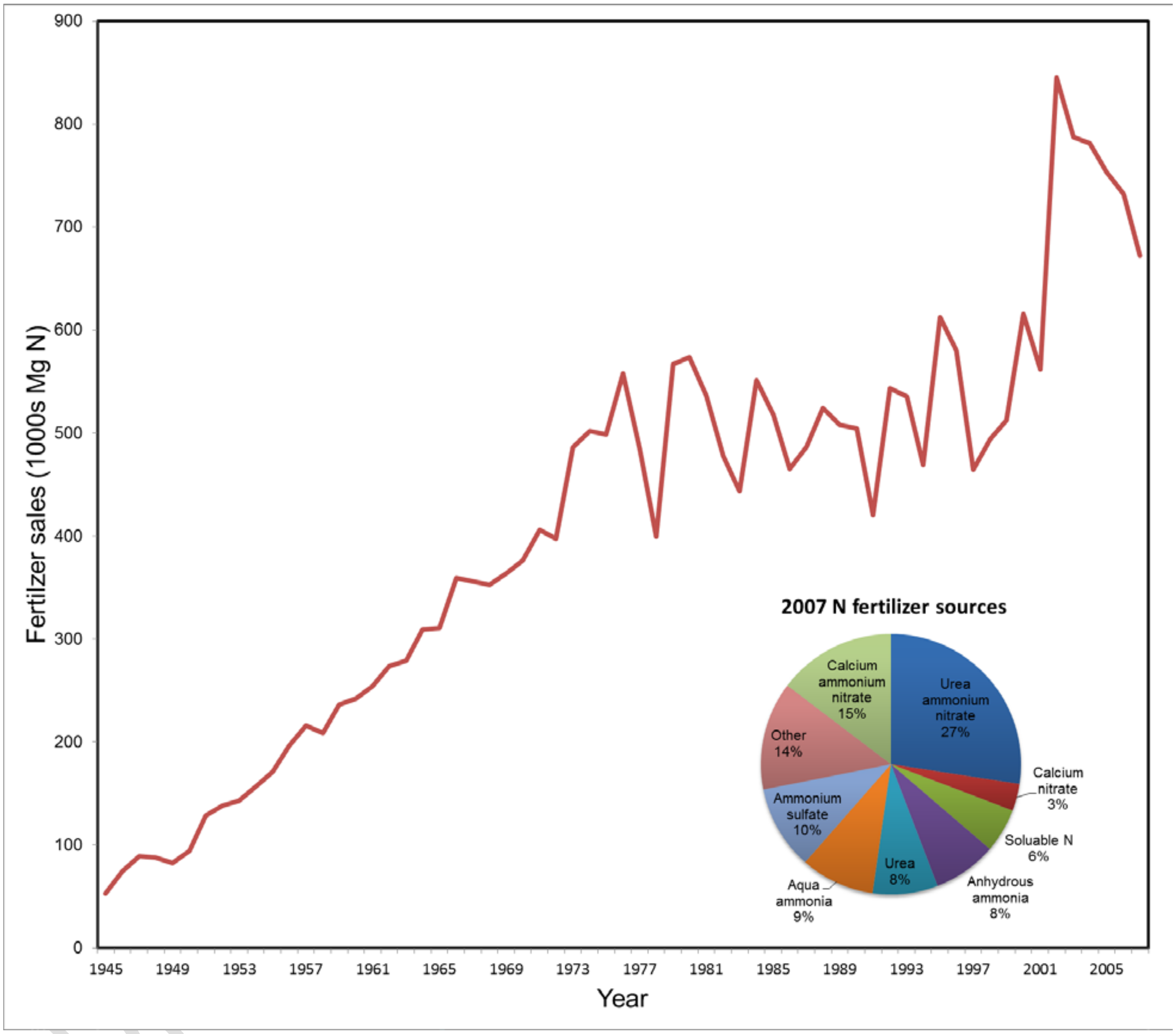


Figure 3.3. Changes in N application rates, yields, and cropped area for 33 crops, 1973 to 2005. The size of circle represents the percentage change in the area cultivated for that particular crop between 1973 and 2005. Closed circles represent increases in cropped area and open circles are declines in area between 1973 and 2005. Source: Rosenstock et al. 2013. [Navigate back to text]

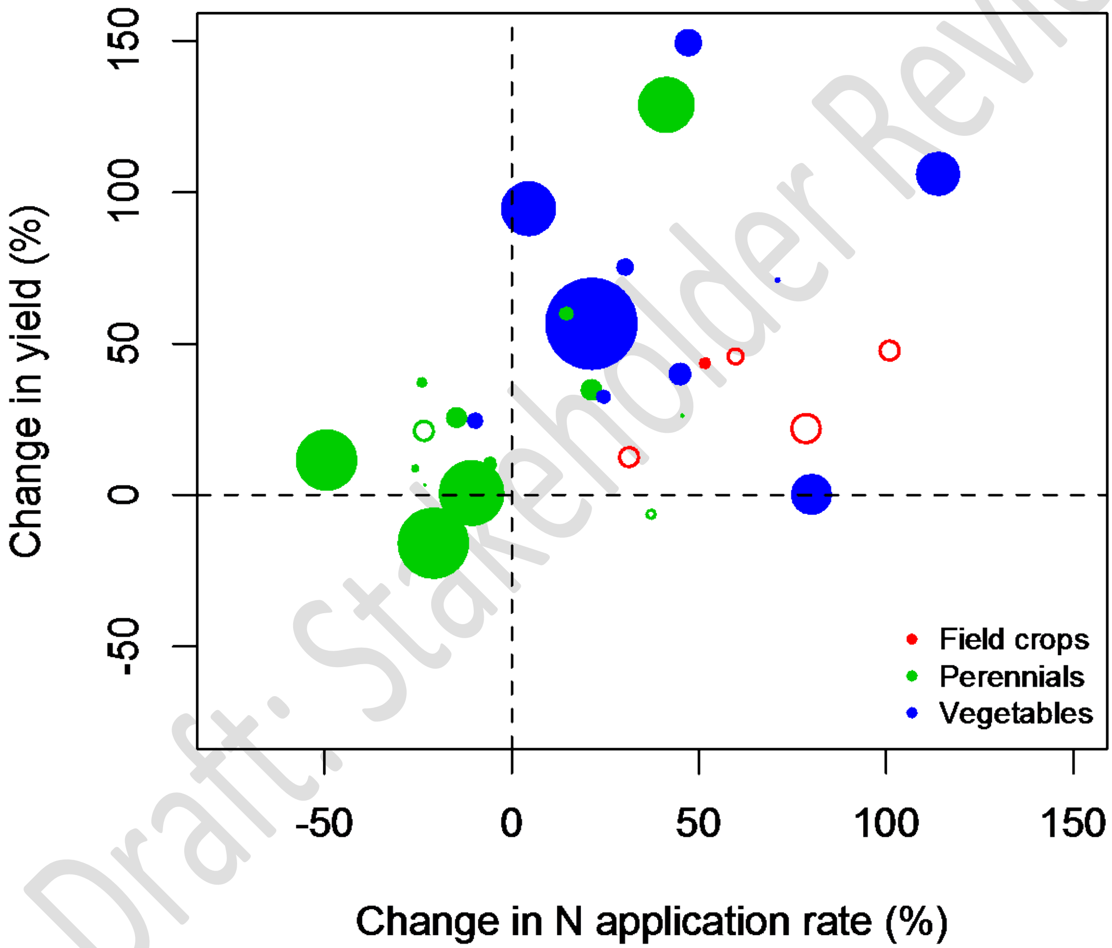


Figure 3.4. Change in cropland area by major crop types in California, 1970-2008. The amount of cropland dedicated to field crops has declined steadily since 1980. Today, almost 50% of cropland is used to grow horticultural commodities. Source: USDA (2009). **[Navigate back to text]**

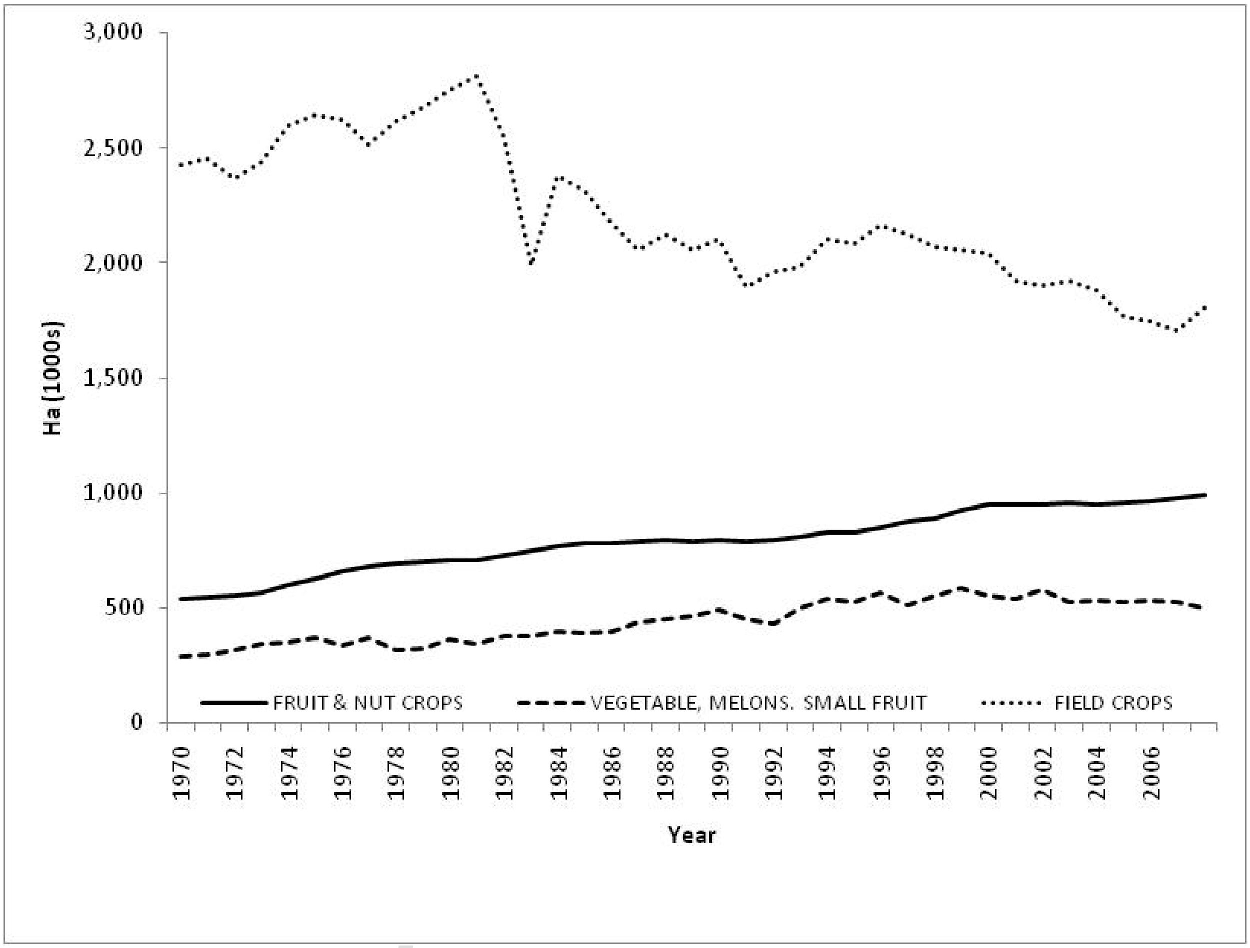


Figure 3.5. Cropped area and yield of alfalfa in California, 1950-2007. Data show that area has remained relatively the same but productivity has increased markedly. Because biological N fixation is correlated with dry matter production, data suggest C-BNF introduces considerably more N into California's biosphere than a half century ago. Source: USDA (2009). [Navigate back to text]

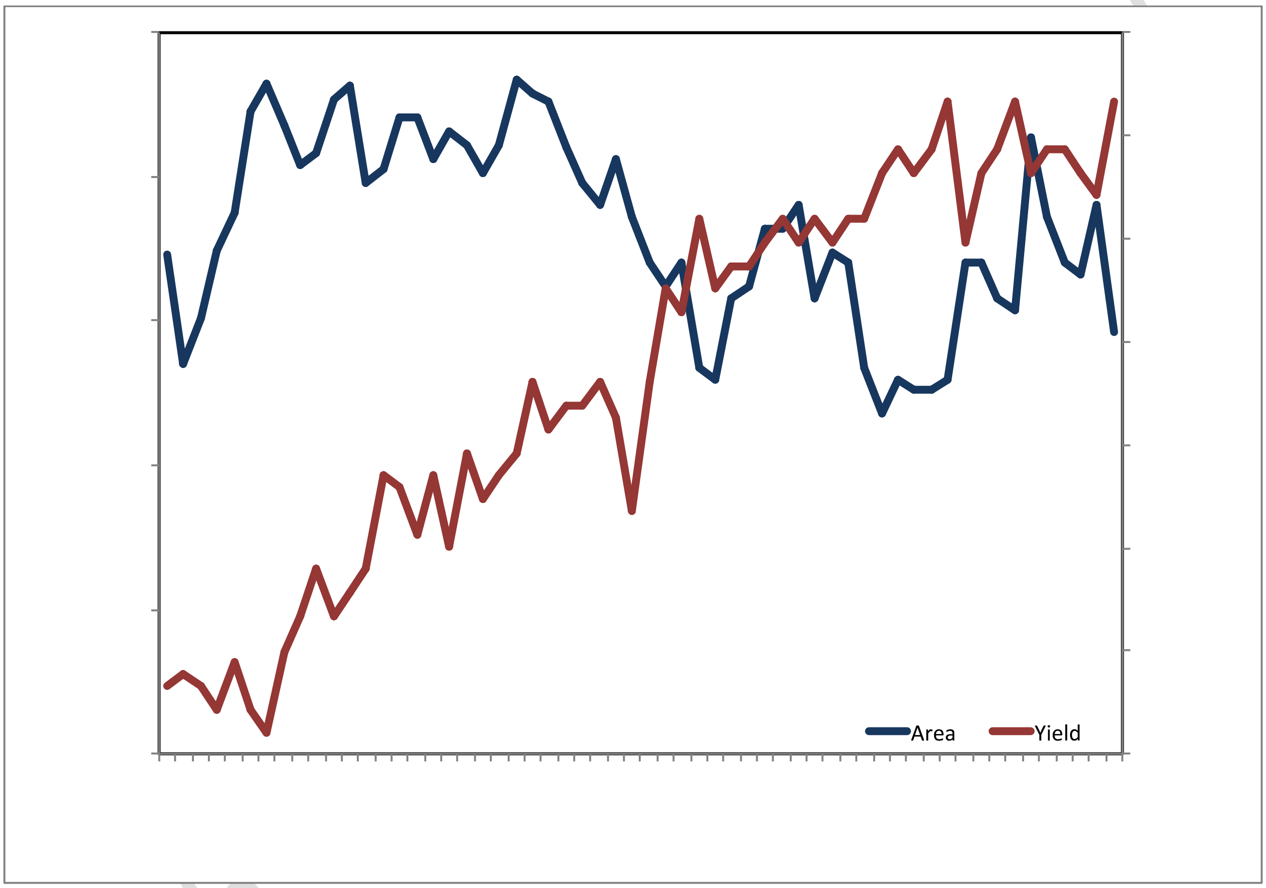


Figure 3.6. Change in California's animal inventory, 1970-2007. The number of milk cows and broilers has more than doubled since 1970 while other animal populations have declined slightly. Source: USDA (2007); USDA (2010). **[Navigate back to text]**

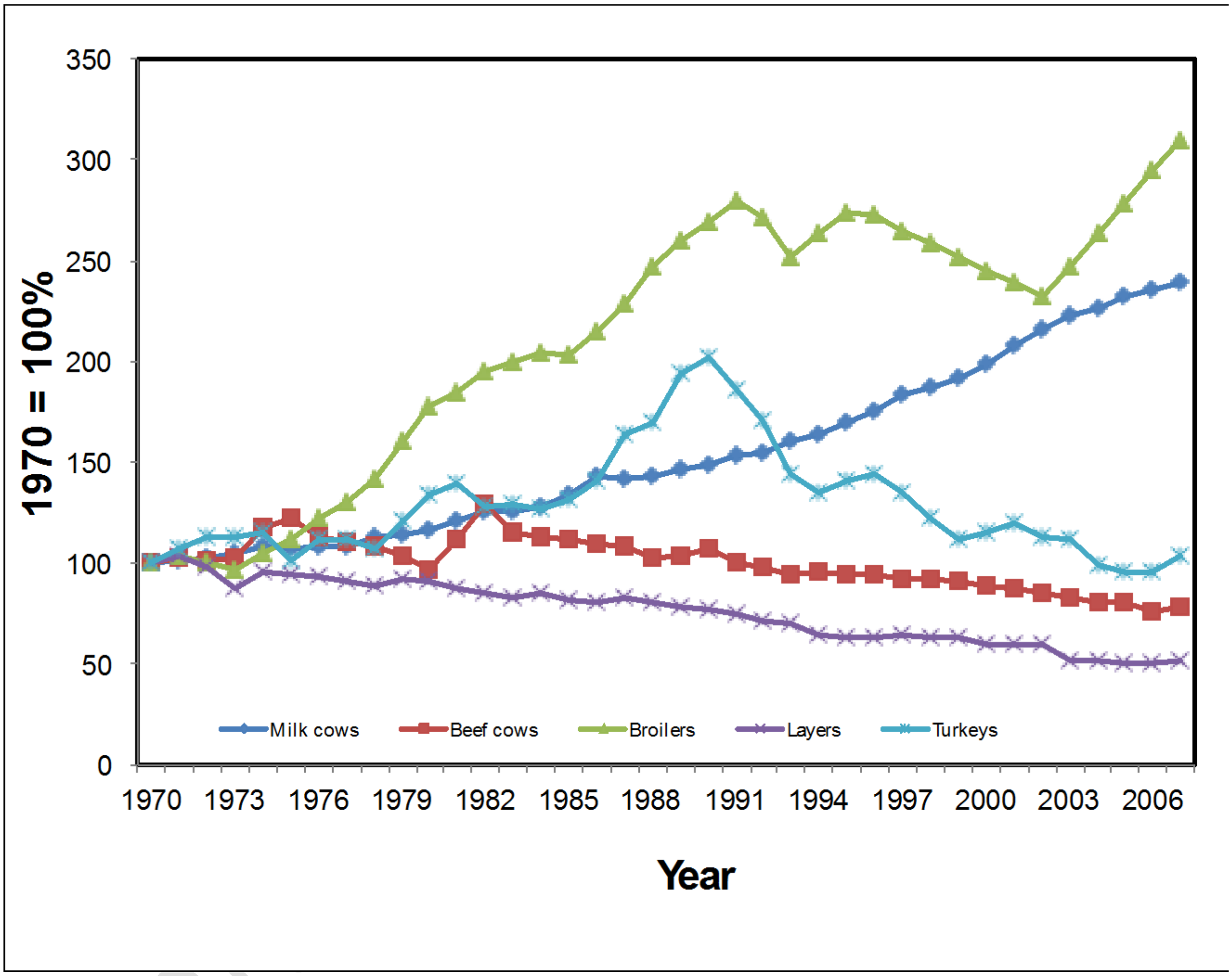


Figure 3.7. Relationship between feed nitrogen intake and (a) faecal nitrogen, (b) urine nitrogen, (c) milk nitrogen, and (d) milk nitrogen efficiency. As N intake increases, part of the additional N may increase milk N but the majority is excreted as highly volatile urea in urine. Source: Dijkstra et al. (2011). [Navigate back to text]

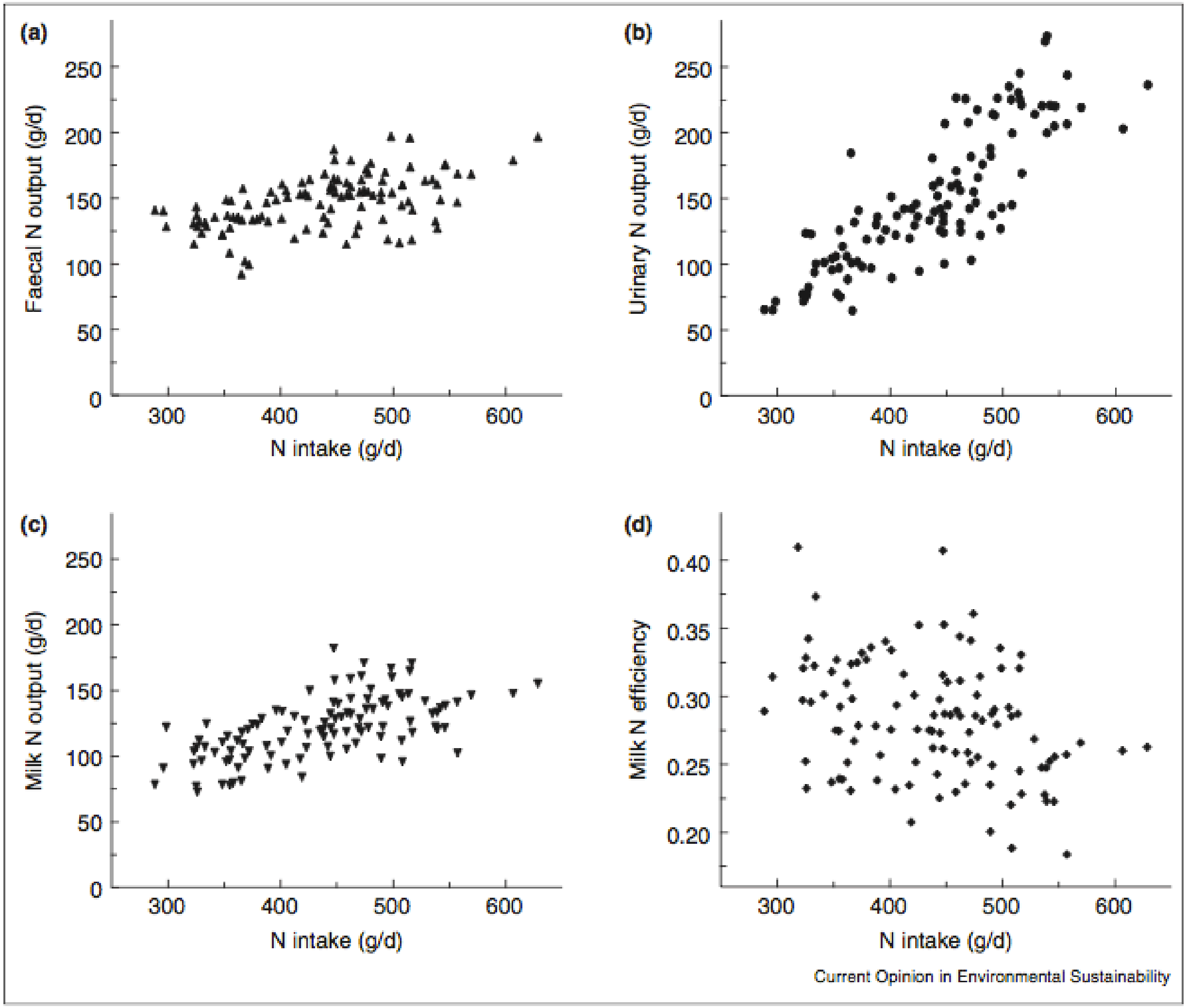
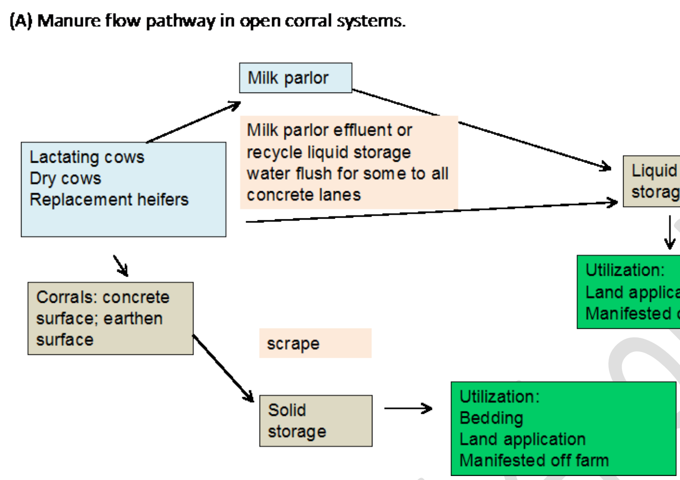


Figure 3.8. Common manure treatment trains on San Joaquin Valley dairies, 2010. (A) Manure flow pathway in freestall systems with or without open corrals. (B) Manure flow pathway in open corral systems. The diagrams shown here demonstrate major processes and the intricacy of manure handling on dairies. Manure management is a complex interdependent system constrained by the facility design. Source: Modified from Meyer et al. 2011.

[Navigate back to text]

Key: source generation | transfer | storage | utilization



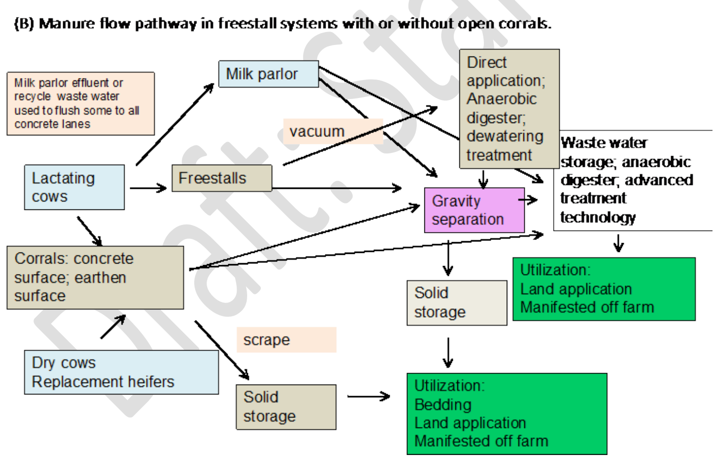


Figure 3.9. Vehicle inventory and total distance driven in California, 1980-2007. Mobile sources including on- and off-road activities are the primary source for NO_x emissions (greater than 86% of the total). Despite large increases in the number of vehicles (population) and the distance traveled (VMT), there has been a significant decrease in emissions (CARB 2012). [Navigate back to text]

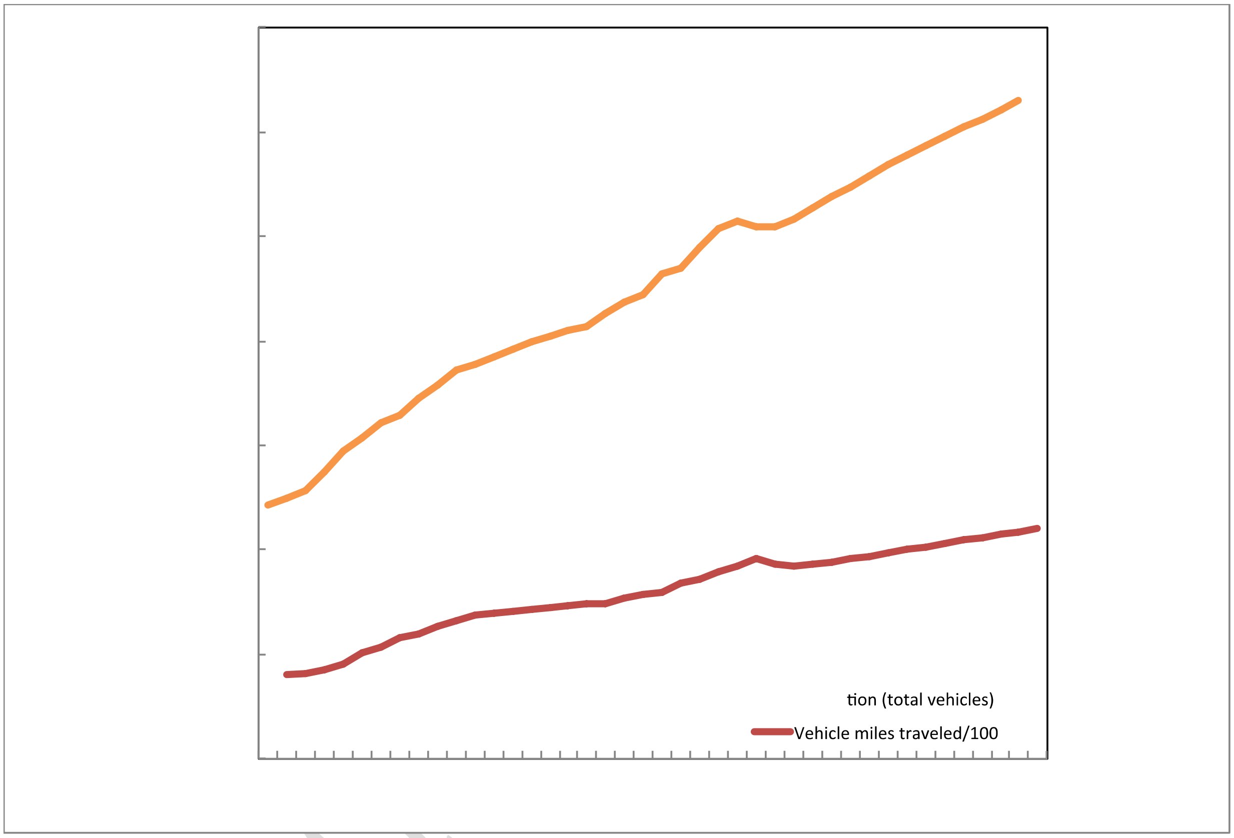


Figure 3.10. Relative contribution of NO_x by major mobile sources in California, 1995 and 2008. The importance of certain sources has changed recently largely as the consequence of technology forcing policies. Regulations have yet to be implemented to control emissions from diesel engines and port activities but are currently under consideration with CARB. Source: Cal EPA (1999, 2009). [Navigate back to text]

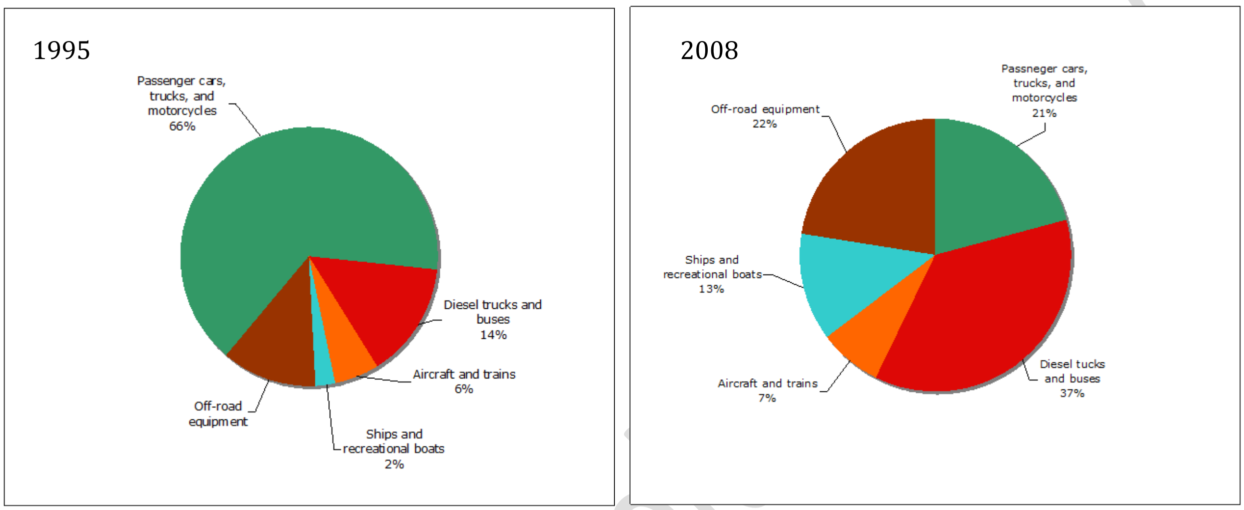


Table 3.1. Fertilizer nitrogen use efficiency (NUE) by ^{15}N, zero-N, and partial nutrient balance (PNB) for select California crops. This table compiles the available estimates for fertilizer nitrogen recover for 21 crops. The three measures differ in their methodology (see Data tables). The ^{15}N 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.

[Navigate back to text]

	^{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	Micke (1996); 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	Weinbaum 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)
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. (1998); 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.

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

[$]Partial nutrient balances calculated as part of this assessment.

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

Table 3.2. 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). **[Navigate back to text]**

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 3.3. Manure management practices in California dairy production, 1988, 1994, 2002, and 2007. [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 1994 refers to "flushing" but does not indicate housing. Even though managing N was not a primary objective until recently, manure management practices used on a dairy will affect N transformation, conservation, and loss. It is thus important to understand how they have changed over time. Source: Meyer et al. (1997), SAREP (2004), and Meyer et al. (2011). **[Navigate back to text]**

	Percentage of respondents				
Practice	1988	1994	2002[3]	2007	2007
Location of dairies	Statewide	Southern SJ Valley	Statewide[1]	Glenn County	Tulare County
Number of dairies		139[2]	428	19	88
Housing and manure collection					
Flushed freestall[4]	61.7	77	66 [9, 23]	63	39
Manure storage ponds	67	96	99		
Solid separation		54		63	71
Settling basins	33	30	66	42	32
Mechanical separation		10	32	5	11
Gravity & mechanical combo.		15		16	27
Solids processing					
Scraped and piled	60	95		80	93
Compost	6	5	21	26	11
Utilization					
Solid	72	78.4	20	89.5	62.5
Liquid	91	70.4	48	100	100
Both			23		
Bedding		27	22	81.8	79.4

Removed from farm		6.8	3		
Sold as liquid		12.2			
Sold as solid	8	58.1	22	26.3	69.3

Table 3.4. The level of treatment at California wastewater treatment plants, 1997 and 2008. Three pieces of information are important to understand: (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 survey reports (1997, 2008). [Navigate back to text]

Treatment level	N removal efficiency (%)	Facility treatment capacity 1996-1997 (%, N = 643)	Facilities treatment capacity 2007-2008 (%, N = 716)	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 3.5. Relative size of N flows on different land uses. Symbols are relative both within row and within column (on a per unit area basis). Source: Expert opinion. [Navigate back to text]

	Land use class				
	Urban core	Suburban-exurban	Crop agriculture	Animal agriculture	Natural areas
Nitrogen sources					
Deposition	+++	++	++		++
Fertilizer (all sources)	++	++	+++	+	
Food	+++	++			
Feed	+	+		+++	
Nitrogen exports					
Food			+++	+++	
Wastewater	+++	++			
Manure				+++	
NH_3	+	+	+	+++	
NO_x	+	+	++	++	
N_2O	+	+	+++	++	
NO_3	+	++	+++	++	
Flow control process					
Biological activity	+	++	+++	+++	+++
Human engineering	+++	++	+	++	
Soil infiltration	+	++	+++	+++	

Table 3.6. Land use change throughout California (%), 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. In short, develop refers to land covered with built structures and impervious surfaces; forests have greater than 10% tree cover; grassland/shrubs have at least 10% of grasses, forbs, or shrubs; agriculture includes croplands and confined livestock areas; mechanically disturbed are transiton areas such as clear cuts or human-induced changes; non-mechanical disturbed are transition areas caused by natural phenomenon such as fire, wind, or flood. See original source for descriptions of each land cover class. Source: Sleeter et al. (2010). [Navigate back to text]

Year	Developed	Forest	Grassland/Shrub	Agriculture	Mechanically disturbed	Non-mechanically disturbed	Year	Developed	Forest	Grassland/Shrub	Agriculture	Mechanically disturbed	Non-mechanically disturbed
California							*Southern California Mountains*						
1973–1980	9.2	−1.0	−0.5	0.2	−29.4	151.9	1973–1980	12.7	1	0.2	−1.8	−88.3	−71.2
1980–1986	6.6	0.3	−0.4	0	57.8	−51.0	1980–1986	9.7	−2.3	−1.2	0.6	117.5	453.6
1986–1992	11	−1.3	−0.3	−1.9	98.9	3.5	1986–1992	10.7	0	1.5	−3.1	116.4	−75.4
1992–2000	6.4	−2.1	−1.2	0.7	−25.8	457.4	1992–2000	5.7	1.8	−1.6	−0.7	11.6	110.2
1973–2000	37.5	−4.1	−2.4	−1.0	64.3	611.7	1973–2000	44.8	0.4	−1.1	−4.8	−38.8	−17.8
Coast Range							*East Cascades and Foothills*						
1973–1980	2.6	−0.6	−0.8	−0.2	24.9	0	1973–1980	−22.6	−0.9	1.8	0.7	46.7	−100.0
1980–1986	2.3	76	6.5	4.4	2.2	0	1980–1986	3.6	0.2	−0.2	−0.2	5.6	0
1986–1992	4.1	74.5	6.8	4.3	3.5	0	1986–1992	8	−1.3	−0.5	1.1	93.9	*
1992–2000	3.5	2.7	2.9	−1.6	−29.5	−100.0	1992–2000	9.7	−1.5	1.5	1	−3.1	−100.0
1973–2000	13.1	−7.0	31.6	−5.3	170	0	1973–2000	−5.0	−3.5	2.7	2.5	191.2	−100.0
Sierra Nevada							*Mojave Basin and Range*						
1973–1980	0	−0.4	3	0	−72.6	34.9	1973–1980	12.5	0	−0.2	−8.9	−9.0	0
1980–1986	0	−0.2	0.7	0	191.5	−99.6	1980–1986	12.2	−3.5	−0.5	6.1	59.7	0
1986–1992	2.5	−0.8	−0.7	0	169.1	25922.2	1986–1992	36.8	0	−1.9	−2.2	194.2	*
1992–2000	3.6	−3.5	3.5	0	−47.8	949.4	1992–2000	3.6	0	0	−6.4	−6.1	−100.0
1973–2000	6.1	−4.9	6.7	0	12.2	1449.6	1973–2000	79	−3.4	−2.6	11.5	301.9	*
Chaparral and Oak Woodlands							*Klamath Mountains*						
1973–1980	10	−3.8	−1.7	−2.1	−56.1	519.7	1973–1980	3.5	−0.3	7.4	0	−64.4	−38.8
1980–1986	5.9	2.9	0.3	−2.1	107.7	−62.3	1980–1986	2.4	0.5	−4.0	0.2	61.4	−96.9
1986–1992	7.7	−0.1	0.1	−3.3	68.4	−47.6	1986–1992	2.3	−0.8	−1.9	0.4	41.6	32654.5
1992–2000	6.2	−5.8	−3.3	−1.1	−46.4	898.4	1992–2000	3.5	−0.1	4.2	0.6	−43.4	−20.6
1973–2000	33.1	−6.9	−4.6	−8.3	−17.6	1122.2	1973–2000	12.2	−0.7	5.4	1.2	−53.9	386.6
Central Valley							*Sonoran Basin and Range*						
1973–1980	9.9	−5.7	−8.1	1	75.8	0	1973–1980	4.1	0	−0.1	*	650	0
1980–1986	5.5	−0.7	−5.6	0.6	−17.5	0	1980–1986	35.3	−9.3	0	0	−70.0	0
1986–1992	8	−0.7	3.8	−1.7	48	0	1986–1992	0.5	10.2	0	0	23.8	0
1992–2000	9.8	−2.1	−11.4	1.2	106.3	0	1992–2000	1.9	0	0	0	25.6	0
1973–2000	37.7	−8.9	−20.2	1.1	342.7	0	1973–2000	44.3	0	−0.2	*	250	*