Chapter 8: Responses: Policies and institutions

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8.1 Experience with nitrogen policy instruments in practice

What is this chapter about?

Because nitrogen emissions from agricultural sources are geographically dispersed, cannot be easily observed, and are difficult to precisely control, this problem presents unique challenges for effective policy design. A suite of integrated practice and policy solutions may be needed to achieve both adequate source control and mitigation of the existing N stock within reasonable timeframes. This chapter provides an overview of available policy instruments for nonpoint source pollution control and examines specific outcomes when these mechanisms have been implemented to control nitrogen pollution in practice. Policy characteristics are then organized into a coherent methodology for assessing candidate policies for controlling nitrogen emissions from agricultural sources in California.

Stakeholder questions

The California Nitrogen Assessment engaged with industry groups, policy makers,, farmers, farm advisors, scientists, government agencies, and non-profit organizations (including environmental, community, and agricultural organizations, and commodity groups). 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:

- How might policy be used more effectively to both monitor and address non-point source agricultural pollution?
- What are the hurdles to having a coordinated and cohesive nitrogen policy across regulatory jurisdictions?

What would the impacts of policy regulating nitrogen use be on farm profits, food prices, and rural economic activity?

Main messages

California's long-term success in achieving environmental goals through regulation of the main sources of nitrogen pollution from combustion (tailpipes and smokestacks) is largely irrelevant to challenges of addressing numerous, spatially-dispersed, highly-variable, context-specific ("non-point") sources of nitrogen pollution typical of agriculture.

Any successful strategy to reduce nitrogen emissions from agriculture must take a comprehensive approach to the most important forms of nitrogen leakage into the environment, particularly ammonia and nitrate, but also including nitrous oxide. Effort to control any one alone, while neglecting the others, is very likely to be counterproductive -- "solving" one problem can worsen others.

There have been apparent improvements in the ability of producers to implement the 4Rs of nutrient stewardship in crop production: right amount, right time, right place, and right form.   Overall, however, although technologies and practices that can reduce nitrogen pollution from agriculture certainly do exist, they typically are costly (in money and management) for farmers and ranchers, thus voluntary adoption tends to be low.

It is well established that voluntary participation in best management practice (BMP) programs typically cannot achieve significant reductions in nitrogen pollution from agriculture.

Dairy waste is a significant source of nitrogen pollution in California, both to water and to air. It is critical to develop and implement cost-effective polices to effectively reduce nitrogen pollution from dairy operations. The California Dairy Quality Assurance Program plays an important role in helping dairies comply with existing regulations. While not a panacea by any means, this is an example of how a voluntary, largely information-based educational program can play a supporting role to other environmental regulations.

Even if policies somehow could perfectly control nitrate leakages from farms and dairies starting immediately, California will be living with the consequences of past nitrate leakages to groundwater for decades to come. Thus, for communities where drinking water supplies are unsafe because of high nitrate concentrations, point-of-use treatment or some other approach will be needed in the short run in order to assure safe drinking water for all California communities.

There is very limited information on the magnitudes of economic benefits that would be achieved through reductions in nitrogen emissions. For this reason it is currently not possible to estimate the economically efficient level of nitrogen emissions—the level that balances marginal benefits and costs—nor the relative efficiency of policy instruments. However it is possible to compare policy instruments in terms of cost to achieve desired emission levels.

Over the longer term, five types of policy instruments appear to be most promising: emission standards, emission charges, tradable emission permits, abatement subsidies, and auction-based abatement contracts. However, theory provides little guidance on which of these instruments would be most effective under specific circumstances. The general lack of evidence, rigorous experimentation, comparative study, or integrated assessment of the impact of alternative policy instruments for controlling nitrogen pollution from agriculture is a major barrier to development of sound policy.

Given the monitoring challenges presented by nonpoint source nitrogen pollution, and the importance of having adequate data to enforce pollution control policies, efforts should be made to develop the technologies and tools needed to acquire the necessary data and to appropriately model the movement of nitrogen in the environment. Doing so facilitates the transition of nitrogen from a nonpoint source problem to a more manageable point source problem.

This assessment concludes that integrated policy solutions are needed to take advantage of existing technology and to develop new technologies and practices necessary to transition California to a sustainable nitrogen future. While a necessary step, design and implementation of an integrative strategy for nitrogen policy holds many challenges, including the need to fill key information gaps, address existing administrative rigidities, and identify conflicting policies.

8.1 Framing of the California nitrogen policy problem

This chapter addresses a number of practical issues in framing policy problems related to nitrogen in California, focusing on policy options for the two most prominent nitrogen problems in the agricultural sector: groundwater nitrate and atmospheric ammonia emissions. First, and foremost, in order to identify workable options that will have the intended effects, it is necessary to be clear about specific economic, social, and environmental objectives and possible relationships between them. By design, the previous chapters in this assessment address these questions regarding overall objectives.

8.1.1. Overview of nitrogen issues

The California Nitrogen Assessment began with a basic question: Is there a nitrogen problem in California? If so, is the problem about agricultural profitability or production costs, climate change forcing, air pollution, surface water pollution, groundwater pollution, public health threats, all of these, some of these, or none of these? In the course of the consultations described in Chapter 1, California stakeholders raised additional, more specific policy-relevant questions, paraphrased as follows:

- What are the main sources of nitrogen pollution and the resulting impacts on the environment and human health?
- How might technology and policy be used more effectively to both monitor and address nitrogen pollution, particularly non-point source pollution from agricultural activities?
- What would the impacts of policy options be on farm profits, food prices, and economic activity?

8.1.1.1. Nitrogen drivers, flows, and impacts

As shown in Chapters 2-4, two broad categories of human activity dominate California's nitrogen cycle: fossil fuel combustion and agricultural production. Despite increases in fuel combustion in California since 1980, emissions have declined steadily. Over the last 50 years, global demand for food has been a fundamental driver of expansion of agricultural production in California. These effects of demand drivers on California agriculture are likely to continue. Over the same period, long-term reduction of both transport costs and international trade barriers increased access to international markets. Long-term decline in synthetic nitrogen fertilizer prices resulted in a large increase in nitrogen use from the 1950s through the 1970s. Thereafter, nitrogen prices were relatively stable relative to the prices of crops until 2000. The future course of these drivers of production and marketing costs are uncertain, particularly regarding energy prices and trade policy. Synthetic nitrogen fertilizer sales in California have risen dramatically since World War II and increased by at least 40% since 1970. Although consumption of synthetic nitrogen fertilizer has leveled off in the past 20 years, the mass balance calculations in Chapter 4 indicate that synthetic fertilizer was still the largest inflow of nitrogen statewide in the last decade, with manure production in second place – a finding corroborated by Harter and Lund (2012). California's livestock herd has continued to grow, despite some slow down during the recent recession. Manure monitoring efforts are underway, but the fate of manure is largely unknown at policy-relevant scales. Nevertheless, the assessments of drivers in Chapters 2 and 3 and the mass balance calculations in Chapter 4 suggest that nitrogen leakages from agricultural activities – both cropping and livestock – are unlikely to decrease and in fact it is likely that they will continue to grow absent major technological and/or policy changes.

So while nitrogen is indispensable in California agriculture, much of the nitrogen applied as fertilizer or manure is lost to the environment, resulting in a variety of negative effects on ecosystems and potential risks to human health. Groundwater contamination is the greatest environmental concern directly related to agriculture. Fossil fuel combustion, fertilizer use, and livestock all contribute to air quality concerns. Chapter 5 reviews the evidence linking groundwater contamination and air pollution resulting from nitrogen leakages to human health risks, which are believed to be significant (albeit neither well-established nor generally accepted in all cases). Costs of treating nitrate contaminated drinking water pose significant financial burdens, especially for low-income households and the water systems that serve disadvantaged communities (Chapter 5).

8.1.1.2. Technological options

Countless technologies and practices are available today that could improve nitrogen use efficiency and reduce nitrogen leakages in agriculture, industry, transportation, water treatment, and waste management. And, as documented in Chapter 7, priorities for mitigating nitrogen emissions include: manure management to reduce ammonia emissions to the atmosphere, soil nutrient management to reduce nitrate contamination of groundwater, fertilizer management and fuel combustion efficiency to reduce nitrous oxide leakage to the atmosphere, and enhanced wastewater treatment to reduce ammonium emissions to surface water.[1] In agriculture, there has not been widespread adoption of available technological options because these measures involve increased costs, greater management effort or effort to access new information, or some combination of one or more of these, while the incentives for adoption have been relatively weak.

8.1.1.3. Rationale for focusing on agriculture

While the evidence in Chapters 2-5 indicates existing policy interventions have been increasingly effective in reducing air pollution, especially from fossil fuel combustion, the same cannot be said about reversing groundwater contamination from agriculture. Indeed, because of the long lag times between initial leakage and eventual groundwater contamination, none of the scenarios presented in Chapter 6 – not even “the end of agriculture” -- reverses the deterioration of groundwater quality in the immediate future. On the other hand, over the longer term, some combination of technological and policy change in agriculture will be necessary to achieve a better overall balance of production, environment, and human health objectives. Thus, while point-of-use treatment will be needed urgently in some communities to address immediate problems of drinking water contamination, this assessment focuses on agricultural source control policy because of the broader benefits to be gained from it (albeit over a much longer time frame).

8.1.2. Policy-relevant characteristics of agricultural nitrogen emissions

Nitrogen emissions from agricultural sources present unique challenges for effective policy design. Foremost, agricultural nitrogen emissions are “nonpoint” emissions: they enter the environment from sources that are geographically dispersed, diffuse, and seemingly random – in other words, emissions

[1] It is well-established that surface water in California is relatively low in nitrate, with the majority of surface water bodies containing average concentrations well below the EPA drinking water MCL. Thus in this chapter we focus on priorities for mitigating nitrogen emissions to groundwater. See Chapters 4 and 5 for more on surface water.

emanate from many locations, cannot be easily observed, and are difficult to precisely control. The broad geographical distribution of sources means that the relationships between production, emissions, and abatement are likely to exhibit significant heterogeneity due to differences in factors such as climate, soil type, hydrology, farming practices, local policies and economic conditions, and other site-specific characteristics. Uniform state-wide policies thus will operate across many local jurisdictions and will have the potential to concentrate abatement effort (and associated costs) due to spatial heterogeneity. The difficulties associated with observability imply that policies requiring knowledge of source emission levels must be based on estimates rather than direct measurement, and also implies that even agricultural producers may not have good information about the magnitudes of their own nitrogen emissions.

In addition, the emitted compounds themselves present specific challenges for policy design. Because the main nitrogen species of concern, nitrate and ammonia, are not conservative pollutants (i.e. they tend to react in the environment to form other compounds), policies should account for potential transformations of nitrogen emissions and associated shifting of pollution damages. Such transformations may be undertaken deliberately by sources in response to policies or may occur naturally in the environment. Furthermore because it is the stock of nitrogen in the environment (e.g., nitrate concentrations in groundwater) rather than the flow of nitrogen into the environment (e.g., nitrate leaching rates from agricultural fields) that is the proximate cause of pollution problems, policies should account for the uncertainties and time lags that are inherent in the relevant environmental fate and transport mechanisms. Importantly this means that relationships between nitrogen sources and their specific impacts may be both spatially and temporally distributed and not well understood. It also suggests that a suite of policies may be needed to achieve both adequate source control and mitigation of the existing stock within reasonable timeframes (See Chapter 5 on spatial and temporal trends and impacts).

Finally the spatial distribution and incidence of damages caused by the stock of nitrogen pollution have implications for policy design. Nitrate and ammonia are not global pollutants – rather the appropriate spatial scale is local to regional. Nitrates tend to contaminate aquifers and ammonia tends to exacerbate air quality problems relatively close to their sources. Policies therefore should target local conditions rather than aggregate state-wide measures of emissions or impacts. This approach not only promotes economic efficiency (Baumol and Oates 1998) but also helps ensure that environmental improvements are attained where they are most needed.

8.2 Overview of available policy instruments for nonpoint source pollution control

Pollution control may be achieved through a variety of policy mechanisms. Each mechanism has its own advantages and disadvantages, some of which may depend greatly on the specific context in which a policy is implemented. However some generalization is possible and even beneficial for understanding the basic properties of broad categories of mechanisms. A taxonomy often used for this purpose includes three such policy categories: (i) education, (ii) standards, and (iii) economic incentives. In this section we describe each of these classes with attention to both textbook treatments of their properties as well as published research and assessments. In the following section we examine specific outcomes when various policy mechanisms have been implemented to control nitrogen pollution in practice. Readers who are familiar with textbook presentations of pollution control policy instruments and their economic properties, especially in the context of nonpoint source pollution control, may find it beneficial to spend relatively less time on this section and relatively more on the subsequent section.

8.2.1 Education

Policies that fall into the Education category are information-based, and are predicated on the idea that when stakeholders have better information about a pollution problem, they will be inclined to modify their behavior in ways that help to mitigate the problem. The type of information provided can vary, and may include labeling or certification of products, training and outreach for producers, reporting inventories or registries, and awards or recognition. In the training and outreach category we include both centralized education programs, typically delivered by third parties such as cooperative extension, agricultural commissioners, farm bureaus, or private consultants, as well as community-based learning efforts in which producers share information with each other about promising production practices (possibly with facilitation by third parties).

Education offers some advantages for addressing non-point source agricultural pollution problems. Information-based policies are flexible: they leave decisions in the hands of stakeholders who tend to have the best information about their own decision environments. Information-based policies tend to be relatively easy to implement and receive relatively little opposition from stakeholders compared to other types of policies. Importantly, education tends to be relatively low-cost for both regulators and producers (Ribaudo et al. 1999). Certain types of labeling programs potentially can incur significant costs due to the need to monitor and certify products. However if this activity can be sourced to an independent third party that specializes in such activities, such as USDA Accredited Certifying Agents for organic standards, then costs to the regulator can be substantially reduced. Training and outreach programs benefit from the pre-existing infrastructure of county extension services, Natural Resource Conservation Service field offices, and land grant universities (Daberkow et al. 2008). These institutions can deliver new content to stakeholders without incurring the potentially large fixed costs that characterize the establishment of such infrastructure. They can also facilitate community-based learning efforts. All types of education-based policies also benefit from the falling costs of generating, storing, processing, and disseminating information (Tietenberg 1998).However such policies must nonetheless compete with a host of other diverse information flows for the attention of the stakeholders they target, some of whom may hold attitudes or ideologies that are unreceptive to new information or may be averse to the risk and uncertainty that tend to be associated with new information. Awards or other types of recognition can be very inexpensive policies. But if the main benefit of the award accrues to only a small number of recipients out of a large number of stakeholders (e.g. a financial award to the farm with the most sustainable production practices), then its overall effect on behavior can be mitigated by the low probability than any single stakeholder wins the award and the associated low expected benefit of pursuing the award. However awards like this also can produce ancillary benefits through information dissemination (e.g. an evaluation of the sustainable production practices utilized by the award winner) that may lead to behavioral changes by stakeholders who are not necessarily motivated by the chance to receive recognition.

The potential impact of education-based policies depends heavily on the specifics of each situation. Perhaps the main challenge of using education-based policies to bring about large changes in pollution loading is that such policies tend to be purely voluntary. Even in cases where training is mandatory, stakeholders who are targeted by the education efforts must subsequently choose to act on the new information. This choice can be costly, and stakeholders typically have little, if any, incentive to act; an education-based policy will not alone influence a change in behavior if there is an economic cost to the stakeholder.

Education-based policies tend to be seen as potentially useful only in “win-win” situations—that is, when pollution can be reduced and stakeholder welfare increased—when such outcomes have not already been achieved. One example is certification of a product for which (1) consumers exhibit significant willingness to pay for more sustainable production practices, (2) suppliers can produce sustainably without significant extra cost, and (3) the certification process is cost-effective and reliable. Under such conditions, which may not occur frequently in practice,, providing the previously missing information to consumers will likely make consumers and producers better off while also reducing negative environmental impacts. Another example is a production practice that both reduces pollution and increases profits, perhaps by more efficiently utilizing inputs. But because producers already face private incentives—a "green" reputation and increased profit—to use such practices, the role of education-based policies in promoting adoption of established "win-win" technologies tends to be limited (Daberkow et al. 2008). However an important exception is when new technologies become available. Enhanced education and outreach efforts can hasten the rate of adoption of such technologies, provided they are not perceived by producers to be significantly risk-increasing and they provide similar benefits in practice as they do under the more carefully controlled settings that tend to characterize research and development efforts.

Although there are many examples of environmental labeling and certification programs (e.g. Energy Star, Green Seal, USDA Organic, Dolphin Safe, Forest Stewardship Council, ISO 14000) as well as inventories and registries (e.g. US Toxics Release Inventory and similar registries in Australia, Canada, Mexico and the EU), training and outreach to producers has had a major role in efforts to control agricultural nonpoint source pollution (Daberkow et al. 2008), even though outcomes have fallen short. Training and outreach efforts include activities like demonstration projects, direct technical assistance, newsletters and seminars, and community-based learning. Successful examples of education-based policies include those promoting the adoption of conservation tillage (Gould et al. 1989), soil and tissue nitrogen testing (Wu and Babcock 1998; Ribaudo et al. 2011), and farm-level information systems (Knox et al. 1995). However there are also examples of education failing to produce significant differences between treatment and control groups for water quality protection practices in Wisconsin (Nowak et al. 1997) and nitrate reduction strategies in California (Franco et al. 1994). Ribaudo et al. (2011) also find evidence that soil and tissue nitrogen testing have much less influence on nitrogen application rates when both commercial fertilizer and manure are applied to crops. Daberkow et al. (2008) conclude that "education by itself cannot be considered a strong tool for water quality protection." (p.904). Daberkow et al. (2008) cite three conditions needed for effective education efforts: (1) a "win-win" scenario, (2) producers with strong altruistic/stewardship motives, and (3) high private costs of water quality degradation (Ribaudo et al. 1999). Unfortunately the convergence of these factors is not common in practice. Daberkow et al. (2008) recommend that education is probably best used as a component of other pollution control policies, such as a mechanism to help producers meet a pollution standard cost-effectively or to effectively utilize new technologies. For example, Bosch et al. (1995) find evidence that additional education makes producers more likely to utilize soil nitrogen testing information. The IPCC (2007, ch.13) reaches a similar conclusion in the context of climate change, stating that "there is only limited evidence that the provision of information can achieve emissions reductions, but it can improve the effectiveness of other policies (*high agreement, medium evidence*)."

8.2.2 Standards

Standard setting, often referred to as "command-and-control" regulation, is the traditional and still most commonly used form of environmental regulation. Policies that set standards obligate producers to meet certain requirements or face consequences for failing to do so. A standard may specify how certain inputs, technologies or management practices may be used in a production process (input standard); or that emissions do not exceed a specified limit (emission standard); or that the ambient concentration of a pollutant in an environmental medium does not exceed a specified limit (ambient standard). Input standards are also referred to as "design" standards while the other types of standards are often referred to as "performance" standards. Consequences for failing to meet a standard can include fines, legal liability for pollution damages and remediation costs, forfeiture of performance bonds, loss of licenses or other privileges (i.e. cross-compliance provisions), and even criminal conviction and imprisonment (Sterner 2003, p.115).

The fact that standards remain the most common type of environmental regulation suggests that they do offer some advantages. The main advantage may be their simplicity and intuitive appeal: when faced with too much (little) of a bad (good) thing, we are inclined to set a limit and enforce it with a penalty. This is a familiar concept that is easily understood both by those who generate and those who are affected by pollution. It also provides for a seemingly decisive response to an issue that elicits strong public concern, which appeals to policy makers who operate on relatively short political timeframes. Another advantage of design standards specifically is that they are relatively inexpensive to monitor (USEPA 2011, p.59). This can be an important consideration when it is difficult or costly to accurately monitor emissions or ambient concentrations and/or predict the environmental effects of varying emission levels. Such was the case in the United States when the Clean Water Act and Clean Air Act were passed, which helps to explain the reliance of those acts on design standards (USEPA 2011, p.59).

The main drawback of design standards is lack of flexibility in both the short and long run. In the short run, when producers are required to utilize or avoid certain production practices in order to achieve a desired environmental outcome, they are rendered unable to fully utilize the private information they have about their specific operations—information that could lead to a different set of production choices that would achieve the same environmental outcome at lower cost. In other words, design standards tend to be cost-ineffective for an individual producer (Sterner 2003, p.76). Furthermore, design standards typically are allocatively inefficient across a group of producers when there is substantial heterogeneity within the group and yet design standards are applied uniformly, as is often done to keep administrative costs low (Sterner 2003, p.77). In the long run, the rigid nature of design standards may undermine incentives for technological innovation because potential innovators must consider the possibility that their costly research and development efforts will produce technologies that are not incorporated into revised design standards and thus cannot be utilized.

Performance standards provide greater flexibility than design standards because they leave production choices in the hands of producers. Thus producers are able to take full advantage of their private information and select the most cost-effective set of production practices that meets the performance standard. However emissions monitoring has presented a considerable challenge for using performance based standards to mitigate agricultural nonpoint source pollution (Ribaudo et al. 1999; Shortle et al. 2012). The diffuse nature of nonpoint source pollution means that monitoring effort cannot be focused on a limited number of pollutant loading points to implement an emission standard. Accurate monitoring of nonpoint source emissions thus tends to be prohibitively expensive, but advances in environmental monitoring technologies can help reduce this cost. Examples of such technologies include the use of Landsat satellite data to estimate field-level evapotranspiration (Idaho DWR 2013) and the use of remotely sensed normalized difference vegetation index (NDVI) to estimate field-level nitrogen application rates (Shaver et al. 2011). Such technologies have not yet been applied to agricultural nonpoint source pollution control and so, in practice, the traditional monitoring problem remains. However we discuss the potential for and implications of improved monitoring in section 8.5.

An alternative to an emissions-based performance standard is to implement an ambient-based performance standard by monitoring pollutant concentrations at a limited number of points in the environment. For example, instead of attempting to monitor runoff and leaching throughout a watershed, surface and groundwater quality would be monitored only at the base of the watershed. While this approach can reduce monitoring costs, other types of administrative costs can be significant because knowledge of the pollutant transport mechanism would be needed to assess penalties on individual producers if/when the ambient standard is violated. Pollution transport mechanisms for agriculture tend to be characterized by spatial heterogeneity, incomplete observability, time lags, and a great deal of uncertainty, implying that they are difficult to specify accurately. However progress continues to be made in this area. For example, Srivastava et al. (2001) integrate a pollutant loading model with a geographic information system (GIS) to assess nonpoint source pollution potential at the watershed level. Alternatively, Millock et al. (2002) give producers the choice of incurring the cost to monitor and verify their own emissions in exchange for a preferable set of regulations that are not ambient-based. Thus each producer can reveal at least some information to the regulator and effectively convert a nonpoint source problem into a point source problem. Kurkalova et al. (2004) find evidence that costly source monitoring can produce significant benefits for the case of carbon sequestration by agricultural producers in Iowa.

A potential compromise between design standards (lower information cost for the regulator, higher implementation cost for producers) and performance standards (higher information cost for the regulator, lower implementation cost for producers) is a standard based on estimated emissions (Ribaudo et al. 1999). In this case, rather than attempting to monitor actual pollution emissions, the regulator utilizes a simulation model that approximates the true pollution generation process to an acceptable degree. The model incorporates input and technology choices by producers and generates an estimated level of emissions to which a standard is applied. A producer then uses the model to determine the most cost-effective set of production practices that meet the standard, and the regulator monitors the producer's choices. Penalties can be assessed in two ways. First, the design-based approach is to assess a penalty if the producer does not implement the agreed-upon production practices. Second, the performance-based approach is to assess a penalty if the estimated emission or ambient standard is violated regardless of the producer's choices. Standards based on estimated emissions thus have potentially low information costs because they rely on an approximation of the true pollution mechanism, as well as potentially low implementation costs (and greater incentives for technological innovation) because they allow producers to make production choices based on their private information. However, as Ribaudo et al. (1999) emphasize, the latter is true only to the extent that the pollution model is able to incorporate private information. Thus the main trade-off associated with standards based on estimated emissions is between incurring the cost of improving the model versus incurring the cost of implementing a suboptimal set of production practices. Monitoring costs tend to be similar to those for design standards, though perhaps somewhat higher due to the associated site-specific modeling efforts that are needed to implement the policy. Daberkow et al. (2008) also note that there may be legal problems with using estimated rather than measured emissions if an acceptable level of model accuracy cannot be achieved.

There are many examples of standards-based environmental policies. Examples include the National Ambient Air Quality Standards, motor vehicle emission standards, technology-based emission standards associated with NPDES permitting and state air quality implementation plans, maximum contaminant levels for drinking water, bans on pesticides and other toxic substances, workplace safety standards, and hazardous waste treatment, storage and disposal standards. However, until recently there have been relatively few examples of standards being used to control agricultural nonpoint source pollution (Daberkow et al. 2008). An exception is a 1986 Nebraska law requiring Best Management Practices (BMPs) that are tied to local groundwater nitrate concentrations.[2] Bishop (1994) found that areas with the highest nitrate concentrations experienced moderate improvements in groundwater quality after this policy was implemented. Another example is concentrated animal feeding operations (CAFOs) for which many states, including California, have adopted design standards for animal waste storage lagoons. Furthermore CAFOs, although nonpoint in nature, are regulated as point sources and thus must adhere to NPDES permitting requirements including a nutrient management plan (NMP) that mandates certain production practices (i.e. a technology-based emission standard).[3] In theory, NMPs

[2] BMPs are technologies and procedures that are determined by regulators to provide an appropriate balance between profitability and pollution mitigation. Our use of the term is distinct from any policy that might require or incentive the use of such practices (i.e. design-based standards or incentives).

[3] Point and nonpoint sources can be distinguished either in legal terms (i.e. applicable regulations) or in physical and economic terms (i.e. whether they emit diffuse pollution that is costly to monitor). Large CAFOs are legally designated under the Clean Water Act as point sources. This has proved effective for reducing emissions to

should be effective in reducing nutrient pollution because they directly limit land application of nutrients in accordance with crop nutrient uptake rates. However studies have estimated that strict NMP compliance by representative CAFOs would produce potentially significant losses in net revenue, depending on farm location and characteristics (Ribaudo et al. 2003; Aillery et al. 2005; Huang et al. 2005; Ribaudo and Agapoff 2005; Baerenklau et al. 2008; Wang and Baerenklau 2014). Importantly, the relatively larger losses—upwards of 27% of net revenues—estimated by Baerenklau et al. (2008) and Wang and Baerenklau (2013) apply to large California dairies that must limit nitrogen application rates to 1.4 times the agronomic rate of crop nitrogen removal (CRWQCB 2007). Thus without adequate enforcement, NMP compliance could be low due to these anticipated loses. Furthermore anecdotal evidence in California suggests that enforcement may be lacking due to both the complexity of each NMP and limited enforcement resources. Therefore NMPs, as currently designed and implemented, may fall short of their pollution reduction potential in practice. Several studies have found that using site-specific information to target nutrient restrictions can reduce economic losses; but when information costs are considered, such targeting may not be worthwhile (Moxey and White 1994; Carpentier et al. 1998).

Another important example of standards applied to nonpoint source pollution are known as "critical N-loads" in the European Union and Total Maximum Daily Loads (TMDLs) in the United States. A TMDL itself serves as an ambient standard: it limits the daily estimated pollutant load to a receiving water body. Furthermore, implementation of a TMDL may involve specific design or performance standards being placed on individual pollution sources. Although the Clean Water Act of 1972 created the legal framework for TMDLs, states have only more recently begun to implement and enforce TMDLs

surface waters in part due to the relative ease of observing discharges from manure handling and storage facilities (Doug Patteson, SWRCB Region 5; personal communication, March 12, 2015). However nitrogen emissions to groundwater and the atmosphere are more difficult to monitor and remain persistent problems.

and therefore more time is needed to monitor and assess their environmental and economic effectiveness.

Ribaudo et al. (1999, p.60) conclude that "performance standards based on runoff or ambient quality are not feasible policies for controlling nonpoint source pollution, given current monitoring technologies." However this does not apply to model-based performance standards that are linked to estimated emissions or ambient quality. Such performance standards as well as design standards are both feasible regulatory instruments for agricultural nonpoint source pollution control, although each has seen limited use in this context. Sterner (2003, p.77) presents some general conditions under which a design standard may be a preferred policy option:

1. Technical and environmental information is complex.
2. Producers do not hold crucial private information.
3. Producer choices are price inelastic and investments are costly to reverse.
4. Standardizing a technology produces significant ancillary benefits.
5. There exists a clearly superior technology.
6. Monitoring technologies are relatively inexpensive.

The IPCC (2007, ch.13) echoes this reasoning, concluding that standards "generally provide some certainty of emissions levels, but their environmental effectiveness depends on their stringency. They may be preferable when information or other barriers prevent firms and consumers from responding to price signals (*high agreement, much evidence*)." In addition, design standards tend to be more preferable when there is limited scope for input substitution. In such cases, standards applied to a few key inputs and practices (thus requiring less monitoring) can achieve close to the same outcome as a complete set of standards applied to all practices that would produce the theoretically optimal input set.

8.2.3 Economic incentives

There is a large body of literature on the use of economic incentives for pollution control. Fundamentally, an economic incentive creates scarcity where none previously existed. This typically involves setting a new price or establishing a new market with a limited supply. Most generally, economic incentives include charges, subsidies, or tradable permits for inputs/technologies/practices, emissions, or ambient quality. As with standards, a similar distinction is made between design incentives that focus on inputs/technologies/practices, and performance incentives that focus on emissions or ambient quality. Charge and subsidy instruments set prices directly whereas tradable permits fix the supply of a desirable good and allow the price to set itself through market transactions for that good. In each case, the price sends a signal to producers and consumers that certain resources are scarcer than previously thought and provides a financial incentive for using less (or providing more) of the resource. A specific example is an emission charge: pricing emissions signals to the producer that the waste disposal services of the environment are not unlimited. Use of these scarce services incurs an opportunity cost that is represented by the price. Now that the service is priced, the producer will tend to use less of it than if it were free. A similar result is achieved when a cap is placed on total emissions and producers must compete for the right to emit. Competition in the market for emissions rights produces a price for emissions that sends the same signal as a charge.

Economic incentives offer several advantages. One of the main advantages of charges or tradable permits for emissions or ambient quality is allocative efficiency. When diverse producers face the same price for emitting a unit of pollution, each will adjust its production such that the marginal cost of abating pollution is equal to the price of emitting. This means that there will be relatively less high-cost abatement and relatively more low-cost abatement which facilitates cost-effective attainment of the aggregate pollution level. Another important benefit of economic incentives is that they provide incentives for innovation in the long-run. Because producers naturally try to minimize production costs, pricing environmental services tends to encourage them to seek out cheaper ways to effectively abate pollution. Some economic incentives (e.g. charges and auctioned tradable permits) also provide a revenue stream that can be used to administer environmental policies, improve environmental quality, and potentially to reduce the distortionary effects of other revenue generating mechanisms such as income charges. Subsidies can be appealing when producers have property rights to the polluted medium and/or the polluting industry produces other uncompensated external benefits.

Economic incentives also exhibit certain drawbacks. The fact that diverse producers will select different abatement levels is advantageous from an allocative efficiency standpoint but it also potentially produces local or regional differences in ambient pollution levels that may be undesirable. Some economic incentives (e.g. charges and auctioned tradable permits) also entail higher costs for producers who must pay not only for abatement but also for emissions. Although this may not create problems related to economic efficiency, it can be a political issue that affects the policy making process. On the other hand, a subsidy tends to be more politically feasible because it ostensibly spreads the cost out over many people (the public) rather than a small group of producers (Sterner 2003, p.104); however the ultimate allocation of costs depends on the specific characteristics of the regulation and how changes in production costs are passed on to consumers. Regardless, either type of price-based instrument may fail to achieve desired levels of environmental quality when—as is common in practice—producers have private information about abatement costs and the regulator has substantial uncertainty about those costs. In such cases, the regulator lacks the information needed to establish an appropriate price level: if costs are higher (lower) than expected, then a given charge/subsidy level will produce too little (much) environmental quality. Furthermore a subsidy creates an additional incentive for entry into the industry which can offset pollution reductions at the individual source level. As is the case for standards, monitoring and enforcement also is an issue for emission based economic incentives since the regulator must verify emission levels in order to collect payments. Ambient quality based economic incentives claim to circumvent this problem but exhibit other attributes that make them problematic in practice for agricultural nonpoint source problems (e.g. the need for large side payments for ambient charge schemes; potentially thin markets for ambient permit schemes when spatial heterogeneity matters). As with standards, a model-based approach to developing incentives linked to estimated emissions or ambient quality may be a good alternative provided an acceptable model of the pollution process is available.

There are many examples of incentive-based mechanisms that either have been applied to nonpoint source agricultural pollution problems in practice or have been studied in the empirical literature. Examples include charges on irrigation water, fertilizer, or runoff; charges on estimated runoff, leaching or groundwater discharge; point-nonpoint trading schemes; and subsidies for agricultural BMPs. There is relatively little empirical information on performance incentives due to the nature of nonpoint source pollution and the inherent emissions monitoring problem. However exceptions include a cropland charge that is tied to phosphorus runoff levels and that has been successfully used to help mitigate phosphorus loading in the Florida Everglades (Ribaudo et al. 1999; Light 2010); and a point-nonpoint phosphorus trading program in eastern Ontario in which all of the regulated point sources have chosen to purchase nonpoint offsets rather than pay for costly upgrades (Wainger and Shortle 2013). There is substantially more information about BMP subsidies (historically the most common policy approach), model-based incentives, and water quality trading.

Examples of BMP subsidy programs include the USDA Environmental Quality Incentives Program (EQIP) and the USDA Conservation Reserve Program (CRP) at the federal level, as well as many state nonpoint source pollution control programs. Subsidies typically take the form of direct payments, cost-sharing agreements, or cross-compliance provisions that require BMP use as an eligibility condition for other agricultural subsidies (OECD 2007). Cross-compliance provisions are seen by some as a promising approach to nutrient management in an era of reduced public funding (Claassen 2004). However, and despite their widespread use, BMP subsidy programs have not achieved significant reductions in agricultural nonpoint source pollution. Shortle et al. (2012, p.1316) conclude that "a 'pay-the-polluter' approach to getting farmers to adopt best management practices has not succeeded in improving water quality in many impaired watersheds." A 2009 report by the State-EPA Nutrient Innovations Task Group echoes this sentiment, finding that "current efforts to control nutrients have been hard-fought but collectively inadequate at both a statewide and national scale" (USEPA 2009, p.1). Part of the problem has been that producers are keenly aware of the long run economic consequences of changing production practices when subsidy programs may be short-lived, combined with the fact that "win-win" scenarios—production changes that reduce pollution without reducing profitability—are uncommon (Daberkow et al. 2008). Shortle et al. (2012) recommend moving away from such voluntary subsidy programs and towards the "polluter-pays-principle" with an emphasis on performance outcomes rather than BMPs.

Daberkow et al. (2008) survey a decade of work (1991-2001) on model-based economic incentives for nitrogen pollution control and conclude that both input and emission charges are problematic due to the costs imposed on producers. Those studies suggest that such instruments may be able to bring about moderate reductions in nitrogen pollution, but more significant reductions (e.g. to drinking water standards) likely will entail substantial economic losses. However more recent work that accounts for the spatial variability of irrigation systems (Knapp and Schwabe 2008; Baerenklau et al. 2008) and also crop choice and the potential to recycle shallow groundwater for irrigation (Wang and Baerenklau 2014) suggests that the economic losses associated with charges on estimated emissions may be substantially less than previously thought. With regard to input and output charges, Sterner (2003, p.100) concludes that "if the abatement possibilities are severely limited and monitoring is difficult, then [charges] on inputs or outputs may be good second-best instruments."

Water quality trading is a relatively new policy innovation that has received significant interest from stakeholders and has experienced relatively fast growth in practice. Water quality trading essentially allows sources with high abatement costs to compensate sources with low abatement costs for reducing pollution on their behalf, thus promoting cost-effective pollution abatement. A 2004 survey by Breetz et al. found that 40 trading initiatives had been created in the U.S. since 1990. Most of these include provisions for trading with agricultural nonpoint sources, however the authors find relatively low participation rates by such sources. Several challenges to establishing effective water quality trading programs have been documented, including: thin markets; difficulty measuring reductions in nonpoint source emissions; insufficient compliance by nonpoint sources with the terms of trade when trading is driven by point source regulations; additional risk borne by point sources when they pay nonpoint sources to abate but cannot also transfer the legal obligation for abatement; aversion by nonpoint sources to being labeled as a polluter for participating in such programs, and the requirement that nonpoint sources satisfy costly baseline conditions (i.e. implementation of prerequisite BMPs) before they can enter the market (Ribaudo et al. 1999; Ribaudo et al. 2011). One area where nutrient trading could overcome some of these barriers is the Chesapeake Bay. This is a large watershed with a regulatory structure in place (TMDL) and the potential for both in-state and out-of-state trading that could overcome the thin markets barrier. An analysis of nutrient trading in the Chesapeake Bay found significant cost savings potential (Van Houtven et al. 2012). An analysis by the USEPA Science Advisory Board (2011) suggests that auction-based contracting, in which BMP subsidies are effectively targeted to desired areas, may be a more appropriate market mechanism than tradable permits for controlling agricultural nonpoint source pollution. Rabotyagov et al. (2012) demonstrate a model-based approach to such contracting for nutrient management. And Selman et al. (2008) estimate that a BMP auction for phosphorus reduction on Pennsylvania's Conestoga River that did not attempt to target contracts was seven times more cost-effective than the standard BMP subsidy approach utilized by EQIP.

8.2.4 Summary

Performance based standards and incentives generally are considered to be infeasible for nonpoint source pollution control problems due to the inherent nature of these problems and the associated monitoring and information costs (Ribaudo et al. 1999). Design based and model based policy instruments are feasible options, with education preferably used in a supporting role. A central tradeoff to consider is the cost of acquiring information that could be used to incorporate site-specific heterogeneity into the policy design, versus the higher abatement costs typically associated with simpler policy instruments that are applied more uniformly across all sources. Both incentives and performance standards create greater flexibility for producers than do design standards and tend to facilitate lower abatement costs and more abatement innovation provided they are costly and enforced. However the greater flexibility provided by price-based incentives also is associated with greateruncertainty over environmental outcomes. Based on experience to date, it is generally accepted that heavy reliance on voluntary participation in BMP subsidy programs cannot achieve significant reductions in agricultural nonpoint source pollution. Rather a new approach that relies on a combination of more effective policy instruments appears needed.

8.3 Framing of Assessment Criteria for Policy Instruments

Section 8.2 provides a general overview of available policy instruments for nonpoint source pollution control, and in the process discusses multiple characteristics to consider when evaluating policies. These include cost-effectiveness, allocative efficiency, administrative cost, complexity, acceptability, environmental effectiveness, feasibility, and uncertainty, among others. The present section considers policy characteristics deliberatively, organizing them into a coherent methodology for organizing evidence about (section 8.4) and assessing candidate policies for (section 8.5) controlling nitrogen emissions from agricultural sources in California.

8.3.1 Policy assessment criteria

The widely used text by Hanley et al. (1997, p.91) lists four main evaluative criteria: effectiveness, efficiency, equity, and flexibility. The authors also provide (p.95) a list of practical considerations that includes information availability, administrative capability, institutional structure, and political feasibility. The IPCC (2007, ch.13.2.2.2) also utilizes a relatively short list of evaluative criteria for climate change policies that is similar to those presented by Hanley et al.: environmental effectiveness, cost-effectiveness, distributional considerations, institutional feasibility. Sterner's (2003, p.214) list of general policy criteria includes some of these common themes (i.e., distributional considerations, political feasibility, economic efficiency) but also includes some additional considerations: general equilibrium effects, number of dischargers, overall complexity, information asymmetry (i.e. monitoring issues), and risk/uncertainty. The European Nitrogen Assessment (Oenema et al. 2011, pp.75-77) adds two more criteria to this list: technological feasibility and enforcement/compliance. Ribaudo et al. (2011, p.47) include geographic coverage for nonpoint source policies while Canada et al. (2012, p.19) also consider revenue generation.

We distilled these suggested considerations into a set of six key criteria for evaluating policies for controlling nitrogen emissions from agricultural sources in California. The criteria are:

1. Environmental effectiveness
2. Technological feasibility
3. Cost effectiveness
4. Distributional effects
5. Institutional compatibility
6. Adaptability

A policy is "environmentally effective" if it has an acceptable likelihood of achieving the desired environmental goal in practice. Here we impose the additional requirement that the policy is not expected to substantially exacerbate the other nitrogen issues identified in this assessment. Therefore a policy that achieves desired nitrate reductions largely by increasing emissions of ammonia is not environmentally effective; one that does so by increasing emissions of inert nitrogen gas is environmentally effective. It is worth emphasizing that if the policy goal is unrealistic or somehow incompatible with the environmental system, then no candidate policies may be environmentally effective. For nonpoint source pollution, such a goal might be an ambient standard that neglects the uncertainty or time lags inherent in the pollution transport mechanism.

A policy is “technologically feasible” if it relies on currently available technologies and practices that are suitable for use by the regulated industry and by the regulator. This includes both abatement and monitoring technologies. Clearly there is a need for judgment to define what is “feasible” or “available” or “suitable” which can be related to the willingness to incur costs for environmental improvements. Here we use a relatively generous definition of technological feasibility and note this may increase our assessments of policy-related costs. This definition also is consistent with a primary emphasis on environmental effectiveness when evaluating policies. If currently available technologies are unproven and thus inherently risky, then a policy is still technologically feasible but the associated risks and learning costs should be considered as additional costs. If currently available technologies can only achieve the policy goal when production is significantly reduced, then this also should be considered as an additional cost. The extent to which a policy drives (and accommodates) technological innovation—a potentially important mechanism for improving cost-effectiveness in the long-run—is more appropriately considered as an element of adaptability.

A policy is “cost effective” if the total economic cost of implementation is expected to be less than that for other policy alternatives. Policy implementation can have ripple effects throughout an economy and thus a full accounting of the associated costs can be elusive. In practice judgments must be made about the scope of an economic policy analysis. This includes defining both the set of relevant stakeholders (those to whom costs accrue) as well as the set of costs that will be evaluated. For our purposes, the set of stakeholders is defined as California's agricultural producers and regulatory agencies. Clearly other residents and businesses in California may incur policy-induced costs, but these will be secondary effects due to responses by producers (e.g., changes in prices for agricultural inputs and outputs). Evaluating welfare effects in secondary markets can be challenging. When those markets operate efficiently, secondary effects often can be ignored (Boardman et al. 1996, p.87); but otherwise secondary effects may be significant. Here we do not attempt to evaluate such effects due to a general lack of good information to do so. Instead, we focus on costs associated with both market (budgetary costs) and non-market goods and that are thought to have potentially significant effects on producers and regulators. Thus this criterion considers: the time cost of producer education and information acquisition; expenditures on abatement technologies; production losses; risk premiums; and education, administration, monitoring and enforcement costs incurred by regulators. It also implicitly considers the allocation of abatement effort across producers and the extent to which this allocation is sensitive to differences in abatement costs because this can have important implications for cost-effectiveness.

Monitoring and enforcement of policy compliance is complicated by political and institutional considerations. Institutional issues are considered later in this chapter. One key purpose of this assessment is to elevate political debate on solutions. To avoid removing promising options from consideration prematurely (before debate can occur) and because there is little basis for assessing *ex ante* the politics of specific options, this analysis deliberately abstracts from the use of political capital to shape policy choice, instrument design, and implementation processes. While it is well recognized that this political power to influence policy is concentrated within specific interest groups and that it is an important aspect of the policy process, assessment of these political realities is highly context specific and there is little relevant scientific literature to be assessed for the specific case of nitrogen in California agriculture.

Characterizing the "distributional effects" of a policy typically involves assessing how the significant policy impacts—both costs and benefits—are allocated across specific groups of stakeholders. We also incorporate an evaluation of that allocation in terms of prevailing notions of equity and social justice—in other words, the extent to which we think a policy will be perceived by stakeholders as "fair," particularly in terms of its impacts on disadvantaged groups. While distributional effects and their perceptions also can have a substantial impact on the political viability of a policy, such political calculus is outside the scope of this criterion.

A policy is "institutionally compatible" if its implementation does not conflict with the larger institutional framework in which it exists. This criterion thus considers legislative authority, legal and jurisdictional constraints, and existing policies and administrative structures, as well as historical/cultural expectations. Compatibility should be assessed at federal, state and local levels. A policy can be institutionally compatible if it creates, for example, the necessary administrative structure when none currently exists; however this likely entails increased administrative cost that would have implications for cost effectiveness. A policy that is at odds with cultural expectations might be made institutionally compatible through education and outreach efforts, but again with similar cost implications.

Finally, a policy is "adaptable" if it is both flexible enough to accommodate changing economic, political and environmental conditions across both space and time, while also being resilient enough to maintain its essential characteristics (i.e. environmental effectiveness,, cost effectiveness, etc.). For example, an emission charge that is tied to an inflation index is more adaptable than one that is not; a technology standard that includes provisions for site-specific characteristics is more adaptable than one that does not. Adaptable policies also accommodate, and preferably incentivize, technological innovation. To the extent possible, policy-makers should consider where significant, "game-changing" innovations may occur, and design policies that will both promote and adapt to these changes if/when they occur.

Economic efficiency is notably absent from this list. This is not because it is unimportant: a thorough accounting of both costs and benefits should inform any policy debate. For many pollutants, including nitrogen, benefits assessments can be challenging because they involve valuing changes in environmental quality and the myriad ways those changes may affect individuals, both now and in the future. Often these benefits are realized outside of the market economy, meaning there are no immediately available price signals to use for developing demand functions and assessing welfare effects. Examples of such nonmarket benefits include recreational use of natural resources, the existence value of biotic species, health outcomes, and the desire to leave a cleaner environment for future generations. Furthermore these benefits often are realized in the future, meaning their actual magnitudes are uncertain and may even accrue to individuals who are not yet alive to express their values. For all of these reasons, formal benefits assessments can be onerous, particularly for the case of a non-conservative pollutant with cross-media pollution potential such as nitrogen.

Despite these challenges, benefits are considered in various ways during the environmental goal setting process, since it is these goals that largely determine the benefits of a policy. Outcomes from this process can include things like maximum contaminant levels for drinking water and ambient air quality standards, both of which apply to compounds of nitrogen. Due to the difficulties associated with undertaking a formal evaluation of benefits, the uncertainties associated with what we currently know about the benefits of alternative levels of nitrogen pollution, and consistent with textbook evaluations of alternative policy instruments, we take the outcomes of the goal setting process as given and focus on evaluating policy approaches for achieving these predetermined goals. Thus “environmental effectiveness” appears as a criterion, effectively serving as a surrogate for benefits. “Distributional effects” also is a criterion, in which the relative allocation of benefits across groups is considered. However the aggregate level of benefits, and by extension economic efficiency, is not considered because inadequacy in current data and methods make it impossible to include in this assessment. .

8.3.2 Policy assessment matrix

Table 8.1 shows the evaluation criteria in table format along with the policy instruments we have selected for evaluation. Labeling and certification programs apply to products and/or producers, thus the information associated with such programs is targeted at consumers. This contrasts with training and outreach programs, and some reporting inventories or registers, for which information is targeted at producers. Examples of the latter include soil testing results, N application rates and methods, or a nitrogen emission inventory. Participation in education programs can be voluntary or mandatory.

[Table 8.1]

We consider standards applied to inputs, emissions, or ambient quality. Input standards implies a broad definition of "inputs" that includes land, production technologies and practices, variable inputs, and abatement technologies and practices. Thus zoning and local ordinances would be input standards. We do not consider mandatory training programs to be input standards unless the policy also mandates use of certain production practices; otherwise use of those practices is unenforceable and thus the program is purely educational. For the case of nonpoint source pollution, emission standards would involve estimated quantities whereas ambient standards can be based on actual observations. We do not consider liability rules as a separate policy instrument because we view it instead as a mechanism to enforce ambient standards. Such a mechanism has institutional precedent (i.e., Cleanup and Abatement Orders in Porter-Cologne) but would present significant technological challenges (i.e. adequate environmental modeling).

We consider seven different categories of economic incentives that cover price (charge) and quantity instruments applied to inputs, emissions and ambient quality; and one auction-based mechanism. As with standards, a broad definition of "inputs" is used here. We consider both positive charges (taxes) and negative charges (subsidies). Thus input charges also include subsidies for pollution reducing inputs; emission charges also include abatement subsidies; and ambient charges also include subsidies when ambient quality exceeds a specified level. Each of the quantity instruments places an aggregate limit on an activity in the pollution process (i.e., purchase of polluting inputs, generation of emissions, or delivery of pollution to receptor points) and allows dischargers to buy and sell rights to these activities in permit markets. As with standards, emission charges and tradable permits and the auction-based mechanism would involve estimated quantities whereas ambient charges and tradable permits can be based on actual observations.

We do not consider any hybrid instruments (e.g. an ambient standard that is implemented through enforceable input standards) or combinations of instruments (e.g., input standards with training and outreach) in detail. However we revisit these possibilities in a subsequent section.

8.4 Experience with nitrogen policy instruments in practice

The overview provided in section 8.2 briefly mentions some specific examples of policies that have been used to control nitrogen pollution in practice. The current section revisits some of these examples along with additional case studies that can inform the policy discussion. The presentation is deliberately brief, but the reader can find supporting details in an online appendix.

8.4.1 Case Study Overview

We consider a total of twelve case studies: five California programs, five nutrient-impaired waterbodies in other states, an overview of European nitrogen policies, and a previously published review of state-level nutrient programs (USEPA 2009). The last of these is qualitatively different from the others and includes both program assessments as well as recommendations for the future. Table 8.2 lists the case studies and shows the topical coverage in terms of policy instruments considered in this assessment.

The figure clearly shows that there is a heavy emphasis on three policy instruments: training and outreach, input standards, and ambient standards. A fourth instrument, input charges, also appears fairly often. Overall this is consistent with the standard approach to regulating nonpoint source agricultural pollution. Section 8.2 argues that this approach has not achieved desired improvements in agricultural nitrogen pollution. The case studies offer some additional insights into and lessons learned from this approach, as well as some information about other less commonly used policy instruments.

[Table 8.2]

8.4.2 Education

Most of the case study evidence for education-based policies applies to training and outreach programs, with each case study including such a component. For two of them that have demonstrated significant nutrient reductions—the Neuse River Basin and the Florida Everglades—education is believed to have played an important role in achieving the program results. More generally, the common theme across case studies is that training and outreach are potentially valuable components of a broader regulatory strategy. Reporting inventories are utilized in three case studies (the State EPA review notwithstanding), but two of these instances apply to greenhouse gas emissions (e.g. N_2O) rather than the pollutants of concern in this assessment. The third instance (the Agricultural Waiver Program) addresses nitrates but was adopted as policy very recently, in 2012. Therefore the effectiveness of such inventories as a practical regulatory policy cannot be inferred from these case studies. However the State-EPA Nutrient Innovations Task Group identifies a "nutrient releases inventory" as a potentially useful approach, as well as "green labeling" for proper nutrient management.

8.4.3 Standards

The ubiquitous use of standards in practice is readily apparent in Table 8.2. Consistent with the diffuse nature of nonpoint source pollution, input and ambient standards are universal in this set of case studies while emission standards are much less common. Nitrogen-related emission standards are limited to CAFOs that are classified as point sources under the Clean Water Act, and combustion sources in Europe and the U.S. Standards for nutrient control typically are implemented through a familiar BMP framework: in areas where ambient standards are not met, education and incentives (usually input subsidies) have been offered to producers to encourage the adoption of approved best management practices (input standards). Some of the more salient lessons that can be drawn from this approach are listed below.

- *Technologies that reduce nitrogen pollution exist, but they are costly to implement and produce relatively small private benefits, thus voluntary adoption tends to be low.* Multiple state-level nonpoint source programs—including California's—have demonstrated BMP effectiveness in reducing nonpoint source pollution in specific cases; but BMP implementation and thus pollution reduction have not been widespread. Experience suggests that producers often perceive the economic costs to outweigh the benefits of participating in such programs.
- *Voluntary programs have not been successful enough, but they have helped to inform questions about BMP effectiveness under different conditions.* One of the benefits of subsidized BMP installations in California and elsewhere is a better understanding of how BMPs perform under real operating conditions. This is potentially useful for future policies that may rely on changes in management practices to achieve pollution reductions.
- *A compulsory yet flexible BMP program with ongoing monitoring, research, and education components has proved to be highly environmentally effective in Florida.* Two relatively unique features of Florida's Everglades Regulatory Program are that participation is compulsory rather than voluntary; and that monitoring was relatively good due to the existence of a network of

drainage canals. And although participation is compulsory it is also flexible: producers must select and implement a minimally sufficient combination of BMPs from an approved menu.

- *Strong collaboration and communication across all parties helps foster success.* North Carolina's Neuse River basin is another BMP success story with similarities to Florida's Everglades. Here again, participation (by both point and nonpoint sources) was mandatory yet flexible. In this case, flexibility was achieved by affording producers the option of working collectively to achieve an aggregate nitrogen reduction target (similar to a tradable permit instrument).
- *A coordinated mix comprised largely of mandatory standards has produced measurable improvements in Europe.* Nitrogen management in Europe is largely governed by multiple EU policy "directives" aimed at reducing nitrogen emissions to water and air. These directives tend to rely heavily on mandatory standards as well as cross-compliance provisions. Implementation is the responsibility of the Member States. Monitoring data shows significant reductions in multiple nitrogen loads from 1990-2010, but also variability across regions.

Regulations have not substantially improved nitrogen related air pollution from California's agricultural sources. Policies regulating nitrogen air emissions in California include an agricultural burning policy, NOx emission limits, and regulations on the disposal of animal carcasses. These policies have had no detectable effect on the number of exceedances of the NO2 standard in agricultural regions.

8.4.4 Economic Incentives

The set of case studies includes a relatively small number of instances where innovative economic incentives have been used to achieve policy goals. However three important approaches are represented: emission charges, tradable permits, and auction-based contracts. Emission charges were implemented in the EU by the Netherlands through the Mineral Accounting System (MINAS). MINAS levied a tax on estimated excess nitrogen and phosphorus flows through agricultural systems. According to Mayzelle and Harter (2011), this approach was popular for its simplicity and had strong support from the Dutch government. Furthermore Westhoek et al. (2004) estimates that it reduced the nitrogen surplus on Dutch dairy farms by approximately 50 kg/ha with very low cost to the affected farms. However the EU determined that the approach did not satisfy the EU Nitrate Directive requirements, so it was ultimately replaced with nutrient application (input) standards. This appears to be a case in which conflicts between federal and state policies undermined an otherwise successful local policy.

Tradable permits have been implemented in the Netherlands (prior to MINAS) and currently in the Chesapeake Bay. Although the Dutch trading system achieved measurable load reductions, implementation was burdened by the anticipated change to the MINAS emission tax and associated uncertainty (Wossink 2003). Trading in the Chesapeake Bay has been very limited, to date. Auction-based contracting was utilized successfully in Pennsylvania's Conestoga River Watershed, which was impacted by excessive phosphorus loads largely from agricultural producers. Two auctions, conducted in 2006, allowed producers to submit bids for installing and maintaining one or more BMPs on their properties. Bidders worked with Lancaster County Conservation District technicians to estimate with computer models their expected phosphorus reductions based on site-specific characteristics. These estimated reductions were used with the bid prices to determine a cost per pound of phosphorus abatement for each bid. Bids were then ranked by cost effectiveness from lowest to highest cost per pound, and contracts were awarded in order of cost-effectiveness until the auction budget was exhausted. The auctions mitigated an estimated 92,000 pounds of phosphorus. This load reduction would have cost more than seven times as much to achieve using standard EQIP subsidies (Selman et al. 2008). Overall the auctions were a success, despite the novelty of and high degree of unfamiliarity with, using of this policy instrument for achieving environmental goals. Additional use of this approach would benefit from a robust outreach, education, and technical assistance component.

8.4.5 Broader Lessons

Several case studies provide broader lessons that are applicable across multiple classes of policy instrument. We list some of the more salient topics here and direct the reader to the online appendix for additional details.

- *Granting the authority to regulate a pollutant does not mean that the pollutant will be regulated: authority is necessary but not sufficient.* California had the authority to regulate nonpoint sources of nitrogen pollution for decades, but allocated relatively little attention to the problem until 2004. Similarly, state implementation of federal TMDL legislation has been slow. Because regulatory resources are limited, specific prioritization of issues is needed to achieve progress.
- *Cross-jurisdictional conflicts can seriously impact program effectiveness.* California's Agricultural Water Quality Grants Program requires disclosure of BMP locations and monitoring points, but this conflicts with privacy provisions of the farm bill and has limited participation. Also California's General Obligation Bond Law requires projects be capital improvements with a useful life of at least 15 years, however most BMPs have a much shorter useful life and thus do not qualify for such funding. Furthermore there are no requirements that matching funds obtained from the federal government through EQIP be used to install desired BMPs.
- *Grant programs are dependent on state financial situations.* The California "bond freeze" of 2008 impaired the ability of grantees and subcontractors in the Agricultural Water Quality Grants Program to complete the work or receive payment for work completed, resulting in a number of stopped or delayed projects.
- *Most programs lack adequate data collection, reporting and evaluation components—particularly of environmental outcomes—but persistent, wide-spread nutrient problems demonstrate that past programs generally have not achieved the desired results.* The State-EPA

Task Group identified this unfortunate information gap in their survey of several programs nation-wide.

- *It has been difficult to document environmental progress, particularly over short time horizons, due to time lags and uncertainties in the pollution delivery mechanism.* This is particularly true for larger watersheds with long distances between sources and receptors, and for groundwater.
- *A one-size-fits-all approach at the federal level can undermine otherwise successful local approaches.* This was the case in the Netherlands when the successful MINAS program was deemed insufficient under the EU Nitrate Directive.
- *Flexibility is crucial for cost-effectiveness.* Programs that account for local conditions, that allow producers to make more of their own choices, and/or that allow for coordination and cooperation among sources (e.g., the Conestoga Watershed and the Neuse River Basin) tend to be more cost-effective.
- *Policies should be designed with the complexity of the larger socio-economic-environmental system in mind.* A narrow focus on nitrogen emissions, or a particular type of nitrogen emissions, can create additional problems, environmental and otherwise. Coordination of nitrogen policies with other environmental, social and economic policies is preferable.

8.5 Assessment of policies for nitrogen regulation in California

8.5.1 Policy assessment rationale

This assessment has identified numerous nitrogen related pollution problems in California. Here we consider potential policy approaches for mitigating agricultural contributions to two high priority nitrogen issues: nitrate emissions to groundwater, and ammonia emissions to the atmosphere. For each issue, we rate different policies in each of the six evaluation criteria using a three point scale: “good” (or “small” for distributional effects) implies an advantageous or beneficial attribute of a policy, “moderate” implies a generally neutral attribute, and “poor” implies a disadvantage or drawback. We then elaborate on these ratings in the main text. The ratings represent qualitative judgments informed by the available evidence presented in this assessment; therefore they are best interpreted in relative terms, by comparing policies against each other. Where ratings depend on specific policy attributes (e.g., voluntary or mandatory, uniform or non-uniform) that imply tradeoffs across criteria (e.g., environmental effectiveness vs. cost effectiveness), we select attributes that emphasize environmental effectiveness provided they remain reasonably technologically feasible and institutionally compatible. Elsewhere, where such tradeoffs do not exist, we assume policies would be well-designed and forego evaluating inferior policies.

As mentioned previously, policy assessment requires first establishing a specific policy goal before proceeding to apply the evaluative criteria. Such goals currently are not available for the problems we consider here. While it may seem appropriate to set goals of achieving the relevant drinking water and ambient air quality standards, the timeframe for doing so and the anticipated contributions from non-agricultural sources remain ambiguous. Therefore we assess candidate policies that have the potential to successfully reduce emissions from agricultural nitrogen sources to levels compatible with long-run attainment of current environmental standards. We do not define the “long-run” specifically but we do note that it is shorter for air quality and longer for groundwater quality—the latter may be on the order of many decades. We thus separate the long-run problem of effectively stewarding resources from the short-run problem of remediation.

Two additional comments on the use of these criteria are worth emphasizing. First, the criteria are most useful when applied to a specific policy mechanism under a specific set of conditions. For the case at hand, there is both uncertainty about the details of any future nitrogen policies that might be adopted and heterogeneity in the conditions under which those policies would be applied. Therefore we necessarily must make some simplifying assumptions and generalizations when evaluating candidate policies. We highlight these where appropriate. Second, the criteria can be used both to evaluate policies and also to assess uncertainty about policy characteristics. As in preceding chapters and sections, we do both of these here by providing specific assessments of central tendencies within each criterion and later making more general observations about the level of evidence and consensus.

8.5.2 Groundwater nitrate policy assessment

[Table 8.3]

8.5.2.1 Education

There is general agreement that education alone is insufficient for mitigating nitrogen pollution problems, including groundwater nitrate. We score each of these instruments relatively low on environmental effectiveness, but also acknowledge that a reporting inventory or register may have somewhat greater effectiveness, as evidenced by the success of the Toxics Release Inventory, provided it is not undermined by moral hazard. Labeling or certification programs can be somewhat more effective but are dependent on consumer willingness to pay for public goods. For the case of groundwater nitrates, the impact of which is mainly a localized health effect, there is no direct evidence on willingness to pay by the broader public. More generally, compared to products like dolphin-safe tuna, bird-friendly coffee, and sustainably-harvested timber, “low nitrogen” farming is arguably less charismatic and suffers from lower levels of public awareness, both of which tend to reduce willingness to pay. Although none of these instruments can be recommended as a cornerstone for mitigating groundwater nitrate, each should be considered in a complementary role to other regulations.[4]

Education programs are generally highly technologically feasible, however we rate labeling/certification programs and reporting inventories/registries lower due to the associated

[4] The literature on effectiveness of public awareness campaigns (diet, exercise, smoking, drugs and alcohol, texting/cell use while driving) may provide relevant insights, but is beyond the scope of this chapter.

emissions monitoring problems. Such programs would have to be based on estimated emissions (or perhaps inputs for labeling/certification programs, which is why they rate slightly better), and thus the feasibility of adequately modeling groundwater nitrate emissions from agricultural operations would need to be addressed. Similarly, the cost-effectiveness of education programs tends to be good, with the same caveat for labeling/certification programs. The labor costs associated with labeling/certification and training/outreach tend to increase the costs of these policies.

Because education programs largely depend on the individual choices made by producers, and sometimes consumers (e.g., use of information in a reporting inventory), they tend to generate an uneven distribution of costs and benefits. Because costs tend to be incurred voluntarily, it is the uneven distribution of pollution reduction benefits (a public good) that is of concern. However, because the environmental effectiveness of these programs is relatively poor, we do not expect there to be significant aggregate differences in benefits. Rather such programs are likely to produce small environmental improvements for the vast majority of the population, and substantial environmental improvements for a small minority (e.g., those who happen to rely on drinking water that is impacted by producers who respond to the education programs), at little cost to producers, which is why we rate them as having small distributional effects.

Institutional compatibility is generally good for education programs since such programs already exist (e.g., cooperative extension). However, the creation of a reporting inventory/register may require new legislation (USEPA 2009). The adaptability of each instrument is generally good, largely because they involve a limited amount of voluntary participation and are based on information transfer; however the incentives for technological innovation tend to be small.

8.5.2.2 Standards

We consider standards for groundwater nitrate management that are compulsory, enforced, and that account for spatial heterogeneity; the last of these implies that input and emission standards would be non-uniform. All types of standards with these properties are potentially effective for groundwater nitrate management. We rate ambient standards somewhat lower because enforcement may be significantly problematic because the regulator knows very little about the contributions of each source.

Input standards that require the use of "best available" technologies should be technologically feasible. Emission standards for groundwater nitrates would be based on estimated emissions, which requires an adequate emissions model; this creates added technical challenges. Ambient standards additionally require an adequate environmental model which further diminishes technological feasibility. It is noteworthy that a TMDL is a type of ambient standard that is being applied to surface water nutrient problems in practice, however implementation typically relies on enforcement of source-specific input or emission standards rather than on penalties for violating the ambient standard; in this sense, TMDLs are hybrid instruments. The same can be said of the National Ambient Air Quality Standards.

The main drawback of using standards is poor cost-effectiveness. However, for nonpoint sources, this conclusion is based on experience with input standards (e.g., NMPs); emission standards have not been used and experience with ambient standards is limited to hybrid applications. Applying emission or pure ambient standards to the nonpoint source groundwater nitrate problem would reduce abatement costs, possibly significantly. However administrative costs could increase significantly. Overall we expect the abatement costs savings, which accrue in perpetuity, to outweigh the administrative costs, some of which would be non-recurring fixed costs and others of which would benefit from economies of scale in implementation; however this is suggested but unproven because of lack of relevant evidence on costs.

Uniform standards would create widespread environmental improvements, implying relatively small distributional effects on the benefit side. However some producers may incur significant abatement costs and some may find compliance to be substantially costlier than others. Small producers also may be disproportionally impacted by fixed costs. Although some of these costs would be passed on to consumers through higher prices, it is difficult to judge these secondary effects. All of this suggests larger distributional effects on the cost side compared to the benefits side. Overall we rate these effects as being larger than for education-based programs, and thus having moderate magnitude.

Although there is precedent for agricultural input standards (e.g. BMPs, nutrient management plans), Porter-Cologne does not permit the state to prescribe abatement technologies, which reduces the institutional compatibility of this approach. However Canada et al. (2012) suggest that this obstacle could be overcome fairly easily with well-designed input regulations, so we rate the institutional compatibility of input standards relatively high. We rate emission standards slightly lower because they would have to be based on estimated emissions which remains a relatively novel concept. Although models are used in other related contexts (e.g., to develop NMPs, establish TMDLs, and set ratios for point-nonpoint surface water trading programs), none of these cases relies on model output to levy a penalty for non-compliance. While this could prove problematic from an institutional perspective, Baerenklau and Wang (2015) argue that a recent legal decision supporting the Chesapeake Bay TMDL bodes well for standards based on estimated emissions. Ambient standards are likely to be more problematic since penalties are not directly related to any source-specific information. Regardless the authority for enforcing emission and ambient standards is inherent in the Porter-Cologne Act.

Compared to institutional compatibility, the relative ranking is reversed for adaptability. Input standards tend to be the most rigid and inflexible. They are furthest removed from ambient groundwater nitrate concentrations so they are least responsive to changes in factors that subsequently combine with the regulated inputs (e.g., other production and abatement technologies, economic and environmental conditions) to produce the damage-causing concentrations. They also provide the smallest incentive for innovation in abatement technologies. Continually revising standards to keep them appropriate and relevant is a costly process. Emission standards perform better in this regard by accommodating changes in farm-level decision-making without undermining their effectiveness; but they do not accommodate changes beyond the farm scale, such as local environmental conditions or industry size. Ambient standards are the most adaptable because they regulate the environmental medium of concern (groundwater quality) directly, thus they accommodate changes in broader environmental and economic conditions as well.

8.5.2.3 Economic incentives: Charges

We focus on positive (non-uniform) charges first before commenting on how the assessment would differ for subsidies. The environmental effectiveness of non-uniform charges for the case of groundwater nitrate is similar to that for standards. Notably water, rather than fertilizer, appears to be the preferable target for an input charge, both in terms of environmental effectiveness and cost to the producer (Helfand and House 1995; Knapp and Schwabe 2008). The ratings for (estimated) emission and ambient charges remain the same as for standards but in practice their effectiveness may be slightly lower due to the additional uncertainty about abatement cost curves.

Technological feasibility of charges is the same as for standards. Cost effectiveness should be similar to, but in practice slightly better than, that for non-uniform standards due to improved allocative efficiency. An efficient reallocation of abatement effort potentially can provide substantial improvements in cost-effectiveness, but due to the spatially heterogeneous nature of the nitrate problem, some of this improvement would need to be sacrificed in order to achieve pollution reduction goals state-wide. Furthermore costs will be higher for producers if charge revenues are not invested back into the industry; here we assume such investment would occur since this should reduce distributional effects, and we rate the distributional effects the same as for the analogous standards. Institutional compatibility for charges is rated similar to that for standards, and for similar reasons; but we note that input charges do not conflict with Porter-Cologne as do input standards (in fact, Porter-Cologne implies that dischargers should incur the costs associated with contaminated drinking water), thus improving their compatibility.

Evaluating the adaptability of charges raises some competing issues. On the one hand, price-based instruments generally provide more flexibility for producers than do standards. On the other hand, such flexibility can directly undermine the effectiveness of charges in a spatially heterogeneous environment. Also, being artificially-set prices, charges may need to be deliberately revised to retain their initial effectiveness when other economic variables change (e.g. inflation, technological innovation). Such revisions can be costly for regulators to promulgate and for producers to respond to, thus we do not expect they would be undertaken as often as they should be. In light of all this, we rate the adaptability of charges about the same as for their analogous standards.

Our assessment of negative charges (subsidies) for pollution-reducing inputs, abatement, or improvements to ambient quality is very similar to that for positive charges. Institutional compatibility would be better due to common past and current experience with agricultural subsidy mechanisms. However the consequence of this approach, as with other similar subsidy policies, is that it artificially increases profitability in the regulated industry, which can increase the size of the industry. This can exacerbate pollution problems rather than mitigate them, even if individual loadings are reduced, unless steps are taken to diminish the benefits received by new entrants due to the subsidy.

8.5.2.4 Economic incentives: Tradable Permits

The logical alternative to a price-based economic incentive (charges) is a quantity-based economic incentive (tradable permits).[5] For the case of groundwater nitrate, a key challenge facing any tradable permit instrument is again related to spatial heterogeneity and the local nature of nitrate pollution. Specifically, local pollution problems require local permit markets. Local markets imply local prices which mirror the non-uniform charges considered in the preceding section, with the additional advantage that local pollution levels are limited by the total number of allocated permits rather than being dependent upon uncertain abatement cost curves. But local markets also tend to be thin which undermines the desirable economic properties of tradable permits. For example, the quantity of trades may be small due to the limited number of potential trading partners, or a small number of firms may develop excessive influence in the market. Either case limits the gains from trade in the market. While the problem of thin markets may not have a solution, regulatory supervision can diminish the potential for market power. However such involvement tends to increase administrative costs. For all these reasons, we rate tradable permits similarly to non-uniform charges but we expect they would be somewhat more costly and also somewhat more adaptable because they are quantity-based rather than price-based.

8.5.2.5 Economic incentives: Auctions

Auction-based abatement contracts are implemented with a reverse auction format in which the producers are the sellers and the regulator is the buyer. The producers submit bids (an abatement plan and corresponding compensation) to the regulator who selects the combination of bids that achieves the environmental goal at least cost. Such auctions exhibit properties that are similar to those for

[5] In a highly cited article, Weitzman (1974) develops a general framework in which quantity-based regulation is preferable to price-based regulation when the slope of the marginal damage curve is steep relative to the slope of an uncertain marginal abatement cost curve. Because we know very little about the marginal damage curve for nitrogen pollution, Weitzman’s often-cited result offers little guidance in this case.

abatement subsidies (negative emission charges), which is why we rate the environmental effectiveness, technological feasibility, and distributional effects the same as for abatement subsidies. However there are also some important differences. First, as with any contract bidding process, there is competition among the sellers to submit bids that are appealing to the buyer. This competition tends to push compensation bids down and thus reduces the surplus of payments received by the producers and the cost incurred by the regulator relative to a standard subsidy mechanism. Second, the regulator is able to deliberately select bids to coordinate abatement efforts across producers and to minimize efforts that are duplicative or even countervailing. So, for example, rather than subsidize many small-scale groundwater capture wells with a uniform subsidy, the regulator might choose to fund a smaller number of large-scale but more cost-effective wells in critical areas of the watershed. This also tends to reduce costs relative to a subsidy mechanism. However the trade-off is that additional administrative costs are needed to run such a program, including development of a nutrient fate and transport model for the affected areas. For these reasons we expect that the cost-effectiveness of auctions would be slightly better than for abatement subsidies, with the cost advantage increasing in the long-run. The institutional compatibility for auction-based abatement contracts is good because both abatement subsidies (which are typically targeted at management practices) and contract auctions are well-established mechanisms that producers should be familiar with. We rate adaptability as moderate because it is possible that producers will favor the relative certitude of longer-term contracts that would be difficult to modify in the short-run.

8.5.2.6 Summary

In order to bring about significant reductions in emissions of nitrate to groundwater, policies must rely on more than just educational efforts. Of the remaining policy instruments, four receive ratings of “moderate” or better for each criterion. These are emission standards, emission charges, abatement subsidies, and auction-based abatement contracts. The first two instruments receive identical ratings, while the latter two receive slightly better ratings than these for institutional compatibility; with the caveat that subsidies potentially increase industry size if steps are not taken to prevent this. Abatement subsidies and contracts are similar policies, which explains their similar ratings, but contracting involves an additional and significant effort to coordinate abatement across sources whereas the standard subsidy approach does not.

This result may seem at odds with the discussion in section 8.2 that recommends moving towards the "polluter-pays-principle" based on the lackluster performance of BMP subsidy programs. However there are noteworthy differences between the subsidy mechanisms we consider and the BMP subsidies that have been implemented in practice. First, the mechanisms we consider are performance-based, rather than input-based, which is consistent with the recommendations in section 8.2. Second, the mechanisms we consider would have subsidy levels high enough to induce adequate levels of participation, robust enforcement of abatement obligations, and contracts that guarantee payment of subsidies over time horizons long enough to justify initial capital investments. If there is insufficient public willingness to pay for such mechanisms, or an inability to achieve these conditions for any reason, then instruments that place the payment burden on producers (i.e., emission standards or charges) would be the leading candidate mechanisms.

8.5.3 Ammonia policy assessment

The challenges associated with mitigating atmospheric ammonia emissions from agricultural operations are similar to those associated with nitrate emissions, but with some noteworthy differences. Ammonia emissions are nonpoint in nature and thus exhibit the same inherent problems of observability and stochasticity that also characterize nitrate emissions. Ammonia emissions also exhibit spatial heterogeneity, but the scale is substantially larger: whereas only a small number of farms (perhaps just one) may be responsible for nitrate contamination of a groundwater well, most air quality issues are regional and some (i.e. greenhouse gases) are global. Therefore policies do not have to be tailored to as many localities. Furthermore it is not the ammonia emissions themselves that cause local air quality problems, but the interaction of those emissions with sulfur or nitrogen oxides (derived primarily from combustion processes) that creates airborne particulate matter. Therefore ammonia regulations ideally should be responsive to whether or not a region is oxide limited. Finally, ammonia derives primarily from animal manure, and management strategies for reducing ammonia emissions typically involve conserving ammonia in manure and/or converting it to nitrate for subsequent field application. Examples include biofiltration, covering animal housing and manure storage facilities, separating solid and liquid waste streams, and applying chemical additives to stored manure; dietary manipulation (i.e. reducing crude protein in feed) is an exception. Although evidence suggests these practices may be relatively cost-effective (Iowa State University 2004; Ndegwa et al. 2008), in order for ammonia control policies to be environmentally effective, the additional nitrate emissions from conserving the ammonia must be mitigated as well. Therefore ammonia control costs should be at least as high as those for nitrate. They also will vary across producers depending on the difficulty of modifying pre-existing housing and manure storage facilities to allow implementation of control strategies.

[Table 8.4]

In light of these similarities and differences, we rate education-based policies for ammonia emissions essentially the same as for nitrate emissions. The noteworthy differences between these two pollution problems do not change the fact that education-based policies rely on voluntary actions and thus have generally poor environmental effectiveness. The other evaluative criteria are similarly dependent on characteristics of the education-based policies, not the empirical problem, and thus remain unchanged as well. However we rate the institutional compatibility of reporting inventories and registries better than for nitrates because ammonia emissions reporting already is required under the Emergency Planning and Community Right-to-Know Act (EPCRA).

For standards we also find broad similarities between nitrate and ammonia policies. However we rate the institutional compatibility of input and emission standards for ammonia slightly better than for nitrates due to the similarities such regulations would have with existing State Implementation Plans for National Ambient Air Quality Standards; but we leave the evaluation for (pure) ambient standards unchanged due to the additional complications that would arise from using the ambient concentration of a final pollutant (particulate matter) to reduce emissions of a precursor pollutant (ammonia). Despite the preceding logic about higher source control costs, we also note that regional (rather than local) spatial heterogeneity should produce administrative cost savings, and so we leave the cost-effectiveness ratings the same as for nitrates.

For the economic incentives, we also perceive there to be broad similarities compared to nitrate policies. However we rate the cost-effectiveness of tradable permits slightly better due to the regional scale of the ammonia problem which should improve market efficiency and reduce administrative costs. Given all of these similarities and again using the “moderate or better” selection criteria as for nitrate policies, five instruments appear to be preferred: emission standards, emission charges, tradable emission permits, abatement subsidies, and auction-based abatement contracts; again with the same caveats as for the case of nitrates.

8.5.4 Additional considerations

The preceding assessment organizes candidate policies and their attributes into a simplified framework for purposes of comparison and evaluation. While useful, there are important additional considerations that should enter any discussion of nitrogen policy. We consider six such issues here: levels of evidence and agreement regarding policy assessment outcomes, emerging abatement technologies, improved monitoring/modeling of nonpoint problems, point-of-use treatment, hybrid policy instruments, and the potential for integrated nitrogen policy.

8.5.4.1 Evidence and agreement

Throughout this assessment, reserved words are used to characterize the levels of evidence and agreement associated with important quantitative and qualitative statements. The policy assessment summary tables contain many such qualitative statements. Generally there is limited empirical evidence to support these statements, particularly for non-traditional policy approaches, as can be seen in the case study matrix in Table 8.2. There are two reasons for this. First, there has not been extensive experimentation with alternative policies for controlling nitrogen pollution. Second, as mentioned in section 8.3, for policies that have been implemented there have been few formal impact assessments. These factors generally undermine the strength and certitude of any policy implications that may be drawn from experience. Therefore the policy assessments must rely more on economic theory and intuition than they do on empirical evidence.

Partly because of this, as well as the inherent scale and heterogeneity exhibited by the pollution problems of concern, it is also difficult to gauge the level of agreement for the assessments. While there may be high agreement among economists regarding the theoretical cost-effectiveness of different policy mechanisms, there may be low agreement among producers regarding abatement costs specifically because of the general lack of evidence and their different operating conditions. Therefore, for purposes of characterizing the evidence and agreement associated with the policy assessments, "tentatively agreed by most" seems appropriate generally.

8.5.4.2 Emerging abatement technologies

The fundamental approach to mitigating nitrate leaching has not changed significantly in the past twenty years. Generally what works is more precise management of water and nitrogen inputs. This includes improved irrigation system uniformity, full accounting of nitrogen sources and sinks, reductions in applied water and N, and proper timing of water and nitrogen applications. Such practices have been called the 4Rs of nutrient stewardship: right amount, right time, right place and right form. Randall et al. (2008) provides a good overview of management practices commonly used to implement the 4Rs. Some of these strategies were used to successfully reduce P loads in the Imperial Valley in 2004 (SWCRB 2010). A full accounting of nitrogen sources and sinks also can bring about changes to cropping patterns that can effectively mitigate nitrate leaching. Some cropping changes, such as fallowing, may create relatively large costs for producers; others, such as the creation of nitrate buffer zones, may not (Mayzelle et al. 2015). Similarly a full accounting can lead to the adoption of improved manure management techniques that reduce volatilization of ammonia from the waste stream and conserve nitrogen on-site for potential use in crop production.

Although the fundamental approach to managing nitrates has not changed significantly, there have been some recent improvements in the ability of producers to implement the 4Rs. For example, nitrification inhibitors, controlled release fertilizers, and precision farming (variable rate) techniques are now commercially available. Recycling of shallow groundwater ("pump and fertilize") also looks effective for both large dairies (Wang and Baerenklau 2014) and crop operations (Dzurella et al. 2012); but so far this practice has had limited field testing. There is some evidence that constructed wetlands may be effective for smaller animal feeding operations (100-200 head) but not for large scale operations (1000+ head) that characterize California's dairy industry (Wang 2012); land costs may be high for large scale operations, as well. There has been substantial interest and effort in designing treatment technologies for animal manure that would function similarly to municipal wastewater treatment plants (i.e., ultimately disposing of waste nitrogen as nitrogen gas), but so far none has emerged as an economically viable option in practice. Other technologies, such as membrane filtration of aqueous ammonia in waste lagoons, (Samani Majd and Mukhtar 2013) and vermiculture-based technology developed in Chile currently remain experimental.

While currently there does not appear to be an obvious technological solution on the horizon, the future is uncertain and therefore policies that incentivize innovation efforts and are flexible enough to accommodate beneficial new technologies are preferable, other things equal.

8.5.4.3 Improved monitoring and modeling of nonpoint source problems

Nonpoint source pollution is characterized by a relative lack of information: regulators are unable to adequately monitor emissions by individual polluters and also cannot accurately infer from observable ambient pollution levels the contributions by individual emission sources. This contrasts with point sources for which monitoring is relatively easy and inexpensive and thus there is relatively little uncertainty about individual loadings. However the distinction between point and nonpoint sources is artificial because monitoring cost is a continuous rather than binary variable. Therefore in reality pollution problems exist along a continuum with some clearly classified as point source problems (very easy to monitor) and others clearly classified as nonpoint problems (very difficult); but many also exist in the middle. As emissions monitoring and modeling technologies improve (i.e., as their accuracies increase and/or their costs fall), more pollution problems can be shifted within the monitoring cost continuum and effectively converted to, and managed as, point source problems.

Examples of recent advances in monitoring technologies include the use of satellite data to estimate evapotranspiration (Idaho DWR 2013), the use of remotely sensed vegetation indices to estimate field-level N application rates (Shaver et al. 2011), and the use of embedded sensor networks to monitor agricultural water quality (Zia et al. 2013). Although such technologies have not yet been applied to agricultural nonpoint source pollution control in practice, they are potentially very useful to the extent they can provide more accurate information to regulators about individual source loadings and thus improve the performance of policy mechanisms that are based on estimated emissions.

While this is good news, some nonpoint source problems—including agricultural nitrate and ammonia emissions—present additional challenges. In both cases it is the ambient concentration of pollution, not the emissions, that is of concern. And again in both cases, the mechanisms that govern the conversion of emissions to ambient concentrations are neither completely observable nor understood. For nitrates, there is also a significant time lag between emissions entering the environment and arriving at a point where they cause damages. For a regulator who is primarily concerned about improving ambient quality by controlling emissions, all of this means that a good model of the environmental fate and transport mechanisms is needed. A poor model will contribute to a misallocation of abatement effort across sources and thus increased costs and/or reduced environmental effectiveness.

Inadequate modeling, or the perception of it, also can increase policy costs by engendering conflicts between regulators and polluters. A recent example is the case brought by the American Farm Bureau Federation and Pennsylvania Farm Bureau against the USEPA (Copeland 2012, p.14). In this case, the plaintiffs argued that the Chesapeake Bay TMDL was “arbitrary and capricious on the basis that EPA used models to support TMDL allocations beyond their predictive capabilities.” (United States District Court for the Middle District of Pennsylvania, p.90).

However in 2013, the U.S. District Court ruled in favor of the USEPA, finding that its use of scientific models and data were reasonable, and deferring to the agency’s expert judgment in relying on those models to promulgate rules (ibid, p.97). The court also emphasized the “heavy burden” of establishing that a model is arbitrary and capricious since it requires establishing “no rational relationship to the realities [it purports] to represent.” (ibid, p.97) While this opinion bodes well for future reliance on models to regulate nonpoint source pollution, it also demonstrates another benefit of implementing defensible modeling techniques.

8.5.4.4 Point-of-use treatment

The timescales for the ammonia and nitrate pollution problems differ significantly. If ammonia emissions were to cease today due to a policy intervention, there would be measurable ambient improvements within days or weeks. But this is not the case for nitrate emissions. Due to the very long time lags that characterize transport of nitrate through the subsurface environment, past emissions of nitrate often will not cause damage at a receptor point for years or decades. In other words, regardless of any nitrate source control policies that might be instituted today, California will be living with the consequences of its past nitrate emissions for a long time.

For this reason, along with the source control policy challenges discussed above, it is worth considering point-of-use treatment as a potential pollution control strategy. That is, rather than controlling sources of nitrate *emissions*, nitrate *pollution* could be controlled at receptor points before causing damages. Such an approach could be implemented, for example, as a type of ambient standard where the standard is applied to *produced* rather than *in situ* groundwater. Although large-scale remediation of California’s groundwater basins would cost billions of dollars over several decades (Harter and Lund 2012), wellhead treatment, blending of available sources, and importing new supplies are potentially cost-effective damage prevention methods(Harter and Lund 2012). Furthermore, by effectively shifting remediation activities into the future, point-of-use treatment costs are further reduced by the long-term effects of discounting.

However this approach presents challenges that are similar to, and in some ways more onerous than, those associated with *in situ* ambient standards. Although long transit times between sources and receptors are beneficial for reducing the present value of abatement costs, long time lags also imply longer distances, greater uncertainties, and thus greater difficulty in adequately modeling the fate and transport mechanisms. This has implications for the cost-effectiveness and technological feasibility of any policy that attempts to regulate current emissions based on estimates of future nitrate concentrations at distant wellheads. An alternative approach would be to charge current producers for current point-of-use treatment costs, effectively passing forward the costs of previous emissions in the same way that some social programs (e.g. Social Security) are structured. While the economic efficiency of such a mechanism is not particularly good (because it fails to internalize the external cost of emissions from each source), it would be a reasonably practical means to fund point-of-use treatment efforts. However it may not have any effect on current emissions of nitrate to groundwater, particularly if a source's charge is unrelated to its emissions. This means that elevated groundwater nitrate concentrations will persist, potentially causing additional unexpected problems in the future.

For these reasons, a dual approach to the problem has substantial merit. In addition to setting new source reduction policies to create a more sustainable future, a separate additional effort would be made to address acute groundwater nitrate contamination problems that otherwise will persist for many years. This would seem particularly important in areas where drinking water supplies are threatened by nitrate concentrations exceeding the MCL. In such areas, point-of-use treatment will be needed until the ambient effects of new source control policies manifest in the future, at which time treatment efforts can begin to ramp down. See Jensen et al. (2013) for an extensive review of treatment options.

8.5.4.5 Hybrid policy instruments

While simple solutions may be adequate for simple problems, more complex problems—such as nitrogen pollution—probably merit more creative and nuanced solutions. For environmental quality problems generally, this often means crafting policies that do not fit neatly into any single category discussed in the preceding evaluations. Hybrid policies that include attributes of multiple policy categories are potentially more effective in practice than a pure policy instrument. Two examples are Total Maximum Daily Loads and the National Ambient Air Quality Standards. Both of these policies appear on the surface to be ambient standards, but each relies primarily on source-specific input and emission standards to achieve the desired ambient goals.

While assessing all such hybrid combinations that might be used for nitrogen mitigation is not feasible here, the preceding evaluations can be used to inform discussion of policies that cut across traditional categories. For example, consider a TMDL-type approach for groundwater nitrates. The loading limit might be applied to each domestic well, and a transport model would be used to calculate the allowable loadings throughout the well capture zone. Input and/or (estimated) emission standards would then be established for nitrogen sources to achieve the desired load. Such a policy mechanism largely would exhibit the characteristics of input and emission standards in Table 8.3, but also should exhibit relatively better adaptability presuming the source requirements are driven by observed concentrations at the wells and are thus easier to modify if/when those concentrations exceed the loading limit.

8.5.4.6 Potential for integrated nitrogen policy

Chapter 7 describes eleven strategic actions that California may take to help solve the nitrogen challenges it faces today. By and large, the actions describe *individual* targets for specific nitrogen sources and impacts. Similarly the present chapter focuses on two critical agricultural nitrogen issues and assesses candidate policy responses for each issue *separately*. While convenient and perhaps even necessary for understanding nitrogen issues and potential responses, such compartmentalization over simplifies both the issues and the necessary responses. Rather, given what we know about the way nitrogen behaves, efforts to deal with excess nitrogen should be organized in a way that reflects the cross-media nature of the problem. Because of their mobility and multiplicative effects, planning for multiple forms of nitrogen in multiple media simultaneously underlies any successful strategy for management. Efforts to control individual nitrogen species alone neglect the synergies and tradeoffs associated with the nitrogen cascade. Poorly designed strategies that fail to account for underlying dynamics of reactive nitrogen in the environment will likely have negative unintended effects. A prime example is the potential tradeoff between the two emissions streams highlighted in this chapter, either of which is likely to increase in response to more stringent policies placed on the other (Aillery et al. 2005, Baerenklau et al. 2008). To address this issue, Yeo and Lin (2014) design a tradable permit system that allows the exchange of nitrogen permits between air and water emissions. With such a permit system, there will be cost savings from trading between air and water since farmers would be able to choose practices that reduce nitrogen emissions to air and water jointly at least cost. There will also be environmental benefits from allowing sources to trade between air and water emissions permits, as a system that accounts for damages to air and water will internalize potential spillovers that would arise if air and water emissions were regulated separately and independently.

Management practices and technologies are already available for virtually every source activity (Chapter 7). Yet concentrations of reactive nitrogen in the environment are increasing and concerns for ecosystems and human health are becoming more severe (Chapter 5). Continued degradation can in part be attributed to increased source activity (Chapter 3) and/or lack of effective regulatory policies (this chapter). In many cases, enhanced nitrogen management is constrained by a lack of information or capital investment. That is, the obstacles to utilization are not technical in nature. Therefore, this assessment concludes that technical solutions to the nitrogen challenge exist for California now and thus integrated practice and policy solutions are needed to transition California to a sustainable nitrogen future.

Design and implementation of an integrative strategy is not without challenges. A multitude of factors constrain such an approach, mostly as a result of actions requiring the crossing of multiple boundaries. Divisions are not only physical but also geographic. An integrated approach would require bridging long standing separation between ideas and institutions, for example breaking down the regulatory silos and identifying conflicting policies. It may require Californians to come to an agreement on the lesser of two pollutants—a calculation which may differ by region. Research would need to change too. It would have to shift its perspective and view whole farming, transportation, and city systems, enabling scientists to look across nitrogen sources and their impacts on society for the greatest potential for nitrogen reductions. Additional boundaries also need to be crossed, including those between science, practice, and policy (e.g., air and water quality policies with tools proven not only to reduce emissions but also to improve the ability to monitor). We also need to consider boundaries between spatiotemporal scales (e.g., from field plots to landscapes and from hours, in the case of ammonia emissions, to centuries, in the case of percolation of NO3- to groundwater); and boundaries existing along supply chains (e.g., from pre-farm to fork to human and solid waste disposal). Finally, in order to effectively bridge the aforementioned boundaries, various stakeholder groups, including farmers, low-income communities, and others need to be able to engage constructively with a range of institutions, including regulatory agencies and research institutions. Currently, data that captures the complexities of nitrogen challenges for both source and impacts and could be used to inform the discussion is only narrowly available and not readily connected (Boxes 7.4, 7.5). Reform, expansion, and integration of the monitoring systems will be fundamental to providing farmers, scientists, policy makers, and citizens the information they need for evidence-based decision-making. In short, and consistent with recent recommendations for nitrogen policy in the EU (Bull et al. 2011) and the US (USEPA 2011), we suggest nothing less than a wholesale transformation of how nitrogen is thought of, monitored, and managed in California; development of technical solutions by themselves (without appropriate supporting policies and institutional mechanisms) will be wholly inadequate.

8.6 Conclusion and relevance of California's nitrogen policy for the rest of the world

This chapter has surveyed a variety of environmental policy instruments, both in theory and practice, and has provided an assessment of several instruments for purposes of controlling agricultural nonpoint source emissions of nitrate and ammonia in California. It should be clear that there are not obvious solutions to these problems. Rather, each candidate solution entails tradeoffs as well as increased costs for at least a subset of stakeholders. However some approaches appear more promising than others, and some kind of action is needed given the size and scope of the problems and the potential damages.

Business as usual is not an appealing option. Business as usual means continuing to add more than 500 Gg of nitrate and ammonia each year to California's already stressed groundwater and air resources. Free disposal of agricultural nitrogen waste may have been acceptable at some time in the past, but the massive scale of California's modern agriculture means that free disposal eventually leads to widespread degradation and the associated impacts on ecosystems and human health that have been documented in this assessment. Absent any regulatory action to mitigate flows of waste nitrogen into the environment, we will continue to experience these impacts in more locations and at elevated levels, and will pass on to future generations a problem that is more difficult, costly, urgent and uncertain than the one we currently face.

A reasonable path forward would mirror the dual policy approach recommended in the preceding section. First, in locations where ambient concentrations pose immediate threats to ecosystems and human health, cost-effective treatment and remediation activities should be undertaken. Second, the process of promulgating long-term source-control policy for nitrogen emissions in California should begin. As is apparent from the empirical cases in section 8.3 and the policy assessment in section 8.5, overall there is relatively limited evidence on the effectiveness of specific policy approaches for purposes of mitigating nitrogen emissions. Similarly, Harter and Lund (2012) conclude that "inconsistency and inaccessibility of data prevent effective and continuous assessment" (p.2) of nitrate in groundwater. Both of these observations suggest that a valuable first step would be a needs assessment to determine which information gaps must be filled before appropriate policy decisions can be made (Rosenstock et al. 2013). This could include a vulnerability assessment to determine priority areas, similar to the Nitrate Vulnerable Zones that were established under the EU Nitrate Directive. A second step would be to determine how to fill the identified gaps. Some will require the relatively simple task of integrating disparate sources of information that already exist at the local agency level; others will require new research that aims to reduce some of the uncertainty that has been documented in this assessment.

Whatever policy approaches are chosen, there will be a need for regular monitoring of ambient conditions, review and evaluation to assess and improve policy outcomes. While the State may undertake some assessment activities itself, given the scale and scope of the policy problem and the anticipated widespread effects of the policy response, assessments can be expected to be undertaken independently by researchers at universities, think tanks, and other concerned organizations—similar to the large body of research that has emerged in response to climate change policies. Regardless, the State can facilitate all of these evaluations through careful documentation and maintenance of policy-relevant data, as well as efforts to make the data available and the methods transparent.

The potential benefits of an integrated nitrogen strategy are difficult to measure, but potentially large. This applies to California agriculture and also to other places intensifying production in irrigated systems. Given the ubiquity of nitrogen species in the economy and environment, the magnitudes of the stocks and flows, and the potential damages to ecosystems and human health, the distortions that have been created by relatively cheap disposal of nitrogen byproducts are manifold. For example, due to cheap disposal, fuel and fertilizer appear to be artificially inexpensive, which reduces the costs of transportation and food production; this increases the demand for land which consumes excessive amounts of natural habitat; it also increases the supply of agricultural commodities which reduces their market prices and leads to both water and air pollution and their attendant effects on ecosystems and human health. Therefore policies that effectively make it more costly to dispose of nitrogen byproducts will have ripple effects throughout this chain, potentially affecting everything from agricultural markets to land use patterns to health outcomes. Similarly, policies that affect other elements in the chain—such as agricultural commodity support programs, for example—will impact the efficacy of nitrogen policies.

Despite distinctive features and crucial details of the nitrogen challenges in California, there is reason for some optimism that insights from future efforts to attain a better balance of benefits and costs of nitrogen flows in the state also can provide useful insights for intensive (and intensifying) agricultural systems in other parts of the world. By the same token, California also can benefit from judicious interpretation of lessons gained elsewhere, often at considerable cost. Europe provides a case in point. Despite major differences between California and the European policy setting across the board -- spanning physical, agricultural, environmental, social, cultural, institutional, and political dimensions, to name a few – features of the European case (described in greater detail in the Chapter 8 Appendix, A8.2.11; also see the European Nitrogen Assessment and van Grinsven et al. 2012) suggest some important common lessons and policy parallels. For example, lessons already learned from the European experience indicate that a coordinated mix of mandatory regulatory instruments, including good agricultural practices and nitrogen rate limits, can produce measurable environmental improvements in water and air quality. However, though there is evidence that the integrated European policies have reduced nitrogen pollution, effects are not uniform and addressing nitrate contamination of groundwater seems particularly recalcitrant. These variable outcomes in Europe also appear to reflect political, economic, and environmental differences across regions, providing further support to the point that “one-size-fits all” strategies are unlikely to be effective or efficient. Thus, policy frameworks should embrace the benefits of locally different approaches; otherwise effective local policies may be undermined. European experience also reinforces the point that outcomes depend not just on policies and practices, but also on trajectories of economic expansion or contraction of polluting sectors. It is highly desirable then that policies are designed to be effective under a variety of uncertain future conditions. More generally, the complexity of the nitrogen cycle presents a formidable challenge, particularly for reducing nitrogen from agricultural systems. This provides additional rationale for coordinated policies and cross-compliance requirements.

Is nitrogen the next carbon? As noted in Chapter 1 and examined in various aspects throughout this assessment, nitrogen exhibits qualities similar to carbon. Carbon also is ubiquitous in the economy and environment, is characterized by large stocks and flows, and, while indispensable to life on Earth, also poses a significant threat in terms of potential damages from unintended leakages and emissions into the environment. Hence there has been, and continues to be, much interest in pricing carbon emissions. Doing so not only could mitigate carbon-derived impacts on climate change, but it will also have similar ripple effects as the economy sheds existing carbon-related distortions and readjusts to a new normal in which disposal of carbon is no longer as cheap as it used to be. Pricing carbon emissions thus provides a more integrated, holistic approach than a large number of more narrowly focused policies that would require substantial effort to coordinate. And yet, at the time of this writing, our continuing inability to agree on the need for concerted global action to mitigate climate change by reducing carbon emissions -- much less on practical steps needed to design and implement a strategy to avert its risks -- raises questions about when (or indeed whether) a global strategy for carbon is politically feasible. As a second-best approach, perhaps some of the nitrogen-oriented practices and policies assessed in chapters 7 and 8 can contribute workable examples of regional efforts to govern these “common pool resources” (Ostrom 1990, extended in Dietz et al 2003 and Stern 2011) as an inspiration for a more decentralized approach to carbon emissions too. So, in this sense, perhaps nitrogen is not only the next big global concern, but also may hold some practical lessons for breakthroughs in addressing our carbon-based challenges as well.

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Table 8.1 Policy assessment matrix [Navigate back to text]

	EVALUATION CRITERIA					
SELECTED POLICY INSTRUMENTS	*Environmental effectiveness*	*Technological feasibility*	*Cost effectiveness*	*Distributional effects*	*Institutional compatibility*	*Adaptability*
Education						
Labeling or certification programs						
Training and outreach						
Reporting inventories or registers						
Awards or recognition						
Standards						
Input standards						
Emission standards						
Ambient standards						
Economic Incentives						
Input charges						
Emission charges						
Ambient charges						
Tradable input permits						
Tradable emission permits						
Tradable ambient permits						
Auction-based abatement contracts						

Table 8.2 Case Study Coverage Matrix [Navigate back to text]

	CASE STUDIES											
SELECTED POLICY INSTRUMENTS	*California Nonpoint Source Program*	*California Agricultural Water Quality Grants Program*	*California Central Coast Agricultural Waiver Program*	*California Dairy Regulations*	*California Air Regulations*	*Neuse River Basin*	*Gulf of Mexico*	*Maryland's Nutrient Management Program*	*Florida Everglades*	*Conestoga River Watershed*	*European Experience*	*State-EPA Nutrient Innovations Task Group*
Education												
Labeling or certification programs												X
Training and outreach	X	X	X	X	X	X	X	X	X	X	X	X
Reporting inventories or registers			X		X						X	X
Awards or recognition												
Standards												
Input standards	X	X	X	X	X	X	X	X	X	X	X	X
Emission standards				X	X						X	
Ambient standards	X	X	X	X	X	X	X	X	X	X	X	X
Economic Incentives												
Input charges	X	X					X		X		X	X
Emission charges											X	
Ambient charges												
Tradable input permits								X				X
Tradable emission permits								X			X	X
Tradable ambient permits												
Auction-based abatement contracts										X		

Table 8.3 Groundwater nitrate policy assessment matrix [Navigate back to text]

	EVALUATION CRITERIA					
SELECTED POLICY INSTRUMENTS	*Environmental effectiveness*	*Technological feasibility*	*Cost effectiveness*	*Distributional effects*	*Institutional compatibility*	*Adaptability*
Education						
Labeling or certification programs	Poor	Good/ Moderate	Good/ Moderate	Small	Good	Good
Training and outreach	Poor	Good	Good/ Moderate	Small	Good	Good
Reporting inventories or registers	Poor/ Moderate	Moderate	Good	Small	Good/ Moderate	Good
Awards or recognition	Poor	Good	Good	Small	Good	Good
Standards						
Input standards	Good	Good	Poor	Moderate	Good/ Moderate	Poor
Emission standards	Good	Good/ Moderate	Moderate	Moderate	Moderate	Moderate
Ambient standards	Good/ Moderate	Moderate	Moderate	Moderate	Poor/ Moderate	Good
Economic Incentives						
Input charges	Good	Good	Poor/ Moderate	Moderate	Good	Poor
Emission charges	Good	Good/ Moderate	Moderate	Moderate	Moderate	Moderate
Ambient charges	Good/ Moderate	Moderate	Moderate	Moderate	Poor/ Moderate	Good/ Moderate
Tradable input permits	Moderate	Good	Poor/ Moderate	Moderate	Good	Poor
Tradable emission permits	Good	Good/ Moderate	Poor/ Moderate	Moderate	Moderate	Moderate
Tradable ambient permits	Good/ Moderate	Moderate	Poor/ Moderate	Moderate	Poor/ Moderate	Good
Auction-based abatement contracts	Good	Good/ Moderate	Moderate	Moderate	Good	Moderate

Table 8.4 Ammonia policy assessment matrix [Navigate back to text]

	EVALUATION CRITERIA					
SELECTED POLICY INSTRUMENTS	*Environmental effectiveness*	*Technological feasibility*	*Cost effectiveness*	*Distributional effects*	*Institutional compatibility*	*Adaptability*
Education						
Labeling or certification programs	Poor	Good/ Moderate	Good/ Moderate	Small	Good	Good
Training and outreach	Poor	Good	Good/ Moderate	Small	Good	Good
Reporting inventories or registers	Poor/ Moderate	Moderate	Good	Small	Good	Good
Awards or recognition	Poor	Good	Good	Small	Good	Good
Standards						
Input standards	Good	Good	Poor	Moderate	Good	Poor
Emission standards	Good	Good/ Moderate	Poor/ Moderate	Moderate	Good/ Moderate	Moderate
Ambient standards	Good/ Moderate	Moderate	Poor/ Moderate	Moderate	Poor/ Moderate	Good
Economic Incentives						
Input charges	Moderate	Good	Poor/ Moderate	Moderate	Good	Poor
Emission charges	Good	Good/ Moderate	Moderate	Moderate	Moderate	Moderate
Ambient charges	Good/ Moderate	Moderate	Moderate	Moderate	Poor/ Moderate	Good/ Moderate
Tradable input permits	Moderate	Good	Moderate	Moderate	Good	Poor
Tradable emission permits	Good	Good/ Moderate	Moderate	Moderate	Moderate	Moderate
Tradable ambient permits	Good/ Moderate	Moderate	Moderate	Moderate	Poor/ Moderate	Good
Auction-based abatement contracts	Good	Good/ Moderate	Moderate	Moderate	Good	Moderate