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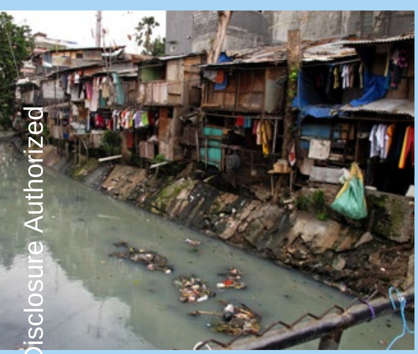
Policy Instruments for Water Pollution Control in Developing Countries



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Contents

1. Introduction

This paper surveys the economic theory behind regulatory and other solutions to the stark ambient water pollution problems that exist in many developing countries, and what is known from the empirical economics literature about the effectiveness of these solutions. Severe water pollution problems are widespread in developing countries. Within the monitored areas of China's main river systems, for example, only 28 percent have water suitable for drinking, and about one-third do not meet the country's lowest ambient water quality standards, making these rivers unsuitable even for irrigation (World Bank 2006). Dissolved oxygen levels - an important indicator of healthy aquatic ecosystems - in China, Brazil, India, and Indonesia (the four most populous developing countries) are well below average levels in industrialized countries (Greenstone and Jack 2015). Major river systems in developing countries are highly impaired; India's Ganga River, alone, receives point source pollution amounting to more than 1.3 billion liters of untreated domestic waste and 260 million liters of untreated industrial waste, in addition to agricultural and urban runoff (Dakkak 2016).[1]

Ambient water pollution due to untreated or insufficiently treated effluent from municipal, agricultural and industrial sources has received much less attention from economists than have unclean drinking water and poor sanitation.[2] Economists have produced convincing empirical estimates of both the negative impacts on morbidity, premature mortality and other aspects of social welfare of inadequate access to drinking water and sanitation, and the positive impacts of interventions designed to expand access.

Several factors may explain the lesser attention paid to ambient water pollution in the economics literature. First, direct consumption of unclean water and exposure to human waste from inadequate sanitation have acute and chronic health impacts causing significant mortality and detriments to human capital formation; thus, the economic impacts of reducing such exposure may be large, making this a potentially high-impact topic of analysis for economists.[3]Ambient water pollution control, in contrast, has less direct impacts on human morbidity and mortality. Recent papers have demonstrated clear connections between some of the most severe ambient water pollution problems in developing countries and human health (Ebenstein 2012, Do et al. 2018). However, demand for regulation may be lower than for air pollution regulation, if wealthy households in developing countries can avoid the effects of water pollution more easily (Dixon 1991). For example, they may use private groundwater wells, access piped water systems, filter water, purchase bottled drinking water, live in areas far enough from polluted water to avoid any local disamenities, and even travel for aquatic recreation.

Second, links between ambient pollution concentrations and the economic value of water resources related to impacts other than human health – commercial fishing, recreation, property values, and non-use values are uncertain and difficult to measure (Keiser et al. 2018, Keeler et al. 2012). Even where they are well-measured, economic impacts may in most cases be highly variable across geographic space, even for a single pollutant (Keeler et al. 2016).

A third reason that ambient water pollution has received somewhat low attention from economists is that collecting ambient water pollution data is challenging. Measurement and monitoring, themselves, are more difficult than for air pollution. While emissions from point sources may be easily monitored, modeling water pollution's fate and transport to surface and groundwater resources, as well as its ultimate impact on ambient concentrations (sensitive to precipitation, runoff, flow, temperature and other factors) is complicated, and even more so for pollution from nonpoint sources such as agriculture and urbanization – an increasing share of water pollution loads (Briggs 2003, Fisher-Vanden and Olmstead 2013). Water quality measurements within individual countries may be taken by many different organizations, including national, state and local governments, research institutions, and non-governmental organizations, leaving researchers to sort through data collected in many different formats and processes, using inconsistent measurement methods, and stored in diverse locations (Keiser et al. 2018). Use of satellite data to measure water quality provides a potential improvement in this regard, though important questions remain to be resolved before the scientific community is confident in the accuracy of this approach (Zheng and DiGiacomo 2017).

In some cases, analyses of large-scale ambient water pollution control policies in industrialized countries suggest that their estimated benefits may not exceed their estimated costs (Keiser and Shapiro 2018). Given the less significant remaining water pollution problems in most industrialized countries, water pollution control in developing country settings may contain a good deal more low-hanging fruit. In addition, human health impacts are only one category of potential damages from severe ambient water pollution. Poor water quality may also damage agricultural output (Hagerty 2018), educational outcomes (Zhang and Xu 2016), and labor productivity (Meeks 2017) in developing countries. Losses to recreational opportunities and other ecosystem services in polluted surface water in developing countries have also been monetized (Beharry-Borg and Scarpa 2010, Choe et al. 1996, Day and Mourato 2002, Mishra 2017).

Given that the literature suggests that ambient water pollution control in developing countries may generate significant economic benefits, how should reductions in municipal, agricultural and industrial water pollution be achieved? This question is the primary focus of this paper. The standard economic prescription would be to harness market forces in any such effort, using approaches such as pollution taxes, tradable permits, and information disclosure to reduce pollution cost-effectively. In industrialized countries, economists can point to many examples of successful market-based approaches to pollution control, from the U.S. sulfur dioxide trading program (Keohane 2006) intended to reduce acid rain (with unexpected, large co-benefits to human health) to the European Union's Emissions Trading System aimed at reducing the greenhouse gas emissions that are changing the global climate (Ellerman et al. 2010). Even in industrialized countries, however, this standard prescription for market-based policy instruments has proven more difficult to fill for water pollution than for air pollution (Fisher-Vanden and Olmstead 2013).

The paper proceeds as follows. In Section 2, we survey the menu of policy instruments used to control ambient water pollution: prescriptive approaches such as technology standards and performance standards; market-based approaches such as pollution taxes, water quality trading, and information disclosure; and voluntary environmental agreements. For each policy instrument, we first offer a brief treatment of the theory behind the approach, and then discuss empirical assessments of these approaches in practice, in both industrialized and developing countries (where such examples are available). Section 3 considers challenges for water pollution control policies that may be specific to developing country settings: sparse and/or poor quality data, and threats to the effectiveness or efficiency of water pollution policies related to weak monitoring and enforcement, rent-seeking, and other issues. We examine what the literature has to say about these questions in terms of both theory and empirical evidence. Section 4 summarizes the gaps in the empirical literature on water pollution policy impacts, and Section 5 concludes.

Notes

1. A 30-hour fire on the Ganga related to pollution from a pharmaceutical plant occurred in 1984 (Do et al. 2018); such fires were not uncommon on major U.S. rivers prior to the Clean Water Act (Olmstead 2010).
2. Ambient water pollution in rivers, streams, lakes, estuaries and other coastal waters are related to drinking water and sanitation problems in two ways. First, nutrients and pathogens from inadequate sanitation enter waterways, creating or worsening ambient water quality problems. Second, where raw water is a drinking water source, even if partially treated, poor ambient water quality contributes to welfare losses from inadequate drinking water access.
3. For two recent examples, see Geruso and Spears (2018), and Galiani et al. (2005).

2. Survey of Approaches Used to Control Water Pollution

2.1. Prescriptive Policies

Until recently, the standard approach to environmental regulation included almost exclusively an array of policy instruments that economists refer to as "command-and-control" (CAC) or prescriptive approaches, which regulate the behavior or performance of individual factories, power plants, and other commercial and industrial facilities. We can divide these approaches into two general classes: technology standards and performance standards. A technology standard requires firms to use a particular pollution abatement technology. For example, the 1977 Clean Air Act Amendments required new power plants to install large flue-gas desulfurization devices ("scrubbers") to remove sulfur dioxide from stack gases. A performance standard allows polluters more leeway in the choice of control technology, imposing a ceiling on total emissions in a period (for example, tons per year), or a maximum allowable emissions rate (for example, pounds of pollution per unit of output produced, or per unit of fuel consumed). Hybrid approaches are also common. For example, in the United States, the Clean Water Act requires individual point sources of pollution to meet emissions limitations based on "best available" technology, which tends to include a single technology, or a small number of related technologies. In theory, regulators can vary technology or performance standards across regulated firms, though in practice, they have tended to implement uniform standards.

CAC policy instruments are not all equal in economic terms. For example, performance standards are generally better than technology standards at minimizing the sum of emissions control costs and pollution damages (Besanko 1987). Even within the category of performance standards, some are better than others in terms of their effectiveness and cost-effectiveness (Helfand 1991). For reasons discussed below, however, economic theory strongly favors market-based over CAC policy instruments.

2.1.1. Applications of prescriptive water pollution policies in industrialized countries

Most existing water quality regulations in industrialized countries are collections of command-and-control approaches. For example, the Clean Water Act (CWA) - the central set of water pollution regulations in the United States — relies primarily on prescriptive policy instruments to achieve ambient water quality improvements. The CWA's main tool is a set of effluent standards, implemented through point-source permitting. The National Pollutant Discharge Elimination System (NPDES) specifies quantitative effluent limits by pollutant, for each point source, based on available control technologies. For the most part, industrial point source compliance with these permits has been high (Freeman, 2000). Municipal sewage treatment has also expanded dramatically, in part due to substantial federal investments, resulting in impressive improvements in urban water quality in some locations. Until recently, no statistical analyses had convincingly established a causal impact of the CWA's prescriptive regulations on U.S. water quality; new results suggest a fairly small positive impact, on average, in U.S. waterbodies (Keiser and Shapiro 2018). When the benefits of the CWA are compared with the costs, however, it appears that the CWA may have had significant net benefits when early water quality gains were achieved, between

1972 and the late 1980s, but that at some point, incremental costs began to exceed incremental benefits (Carson and Mitchell 1993, Lyon and Farrow 1995, Keiser and Shapiro 2018). More work is needed to fully understand the economics of the CWA, particularly to monetize its benefits. However, it is likely that the results of economic analyses thus far – ranging from small net benefits to potentially significant net costs – are due in part to the relatively costly (non-market-based) means by which pollution reductions have been achieved under the CWA, and the fact that standard-setting is not governed by benefit-cost analysis (Olmstead 2010). For example, the CWA's stated goal was the elimination of all point-source pollution to U.S. waterbodies by the mid-1980s.

While the U.S. CWA mandates specific technologies for specific sectors and industries, the EU Water Framework Directive (WFD) takes a different approach, setting up a structure through which individual member states set and implement coordinated but self-selected objectives for ambient water quality and pollution control, and allowing significant discretion in how those objectives are met (Bourblanc et al. 2013). Another aspect of the WFD that sets it apart from the CWA is the way in which it allows economic analysis to enter the standard-setting process within member states. States may demonstrate, for example, through economic valuation of costs and benefits, that the costs of meeting an ambient water quality target in a particular area exceed the benefits (Artell and Huhtala 2017). Because the WFD's ecological goals for surface waters were to be obtained beginning in 2015, it may be possible to estimate the benefits and costs of the WFD, but we know of no published studies that have performed such an analysis.[1]

2.1.2. Applications of prescriptive water pollution policies in developing countries

Like industrialized countries, developing countries regulate water pollution primarily through the use of standards and direct infrastructure investments (e.g. sewage treatment facilities), rather than market-based policy instruments. This could be due to developing countries' having modeled their own pollution control regulations on statutes like the U.S. CWA, or it could be due to the fact that some of the same challenges to market-based approaches to water pollution control that exist in industrialized settings also exist or are even intensified in developing countries. We consider some of the challenges to water pollution regulation that may be specific to the developing country context in Section 3.

India's National River Conservation Plan (NRCP) is an example of a prescriptive approach to water pollution control. Like the U.S. CWA, India's NRCP, starting in 1985, established a set of designated uses for surface waters and prescribed a set of approaches for achieving levels of water quality appropriate to those designated uses. However, while the NRCP prescribes the construction of sewage treatment plants and other capital investments to reduce water pollution, it does not provide a dedicated source of revenues to fund those investments (in contrast, the U.S. CWA, in early decades, established state revolving funds that effectively subsidized local treatment plant construction and upgrades with federal dollars). Greenstone and Hanna (2014) find that India's NRCP has not reduced water pollution concentrations in river segments covered by the Plan. They attribute this failure to weak institutional support for the NRCP's goals, and low public demand for ambient water quality improvements (supported by their finding of low rates of reference to the term "water pollution" in India's most widely-read national newspaper).

In China, the current primary policy on water pollution reduction is the Water Pollution Prevention and Action Plan, also named as the "Water Ten Plan" (State Council 2015). The Water Ten Plan establishes a sets of targets for water pollution reduction and lays out approaches to achieve the requirements, such as setting pollution reduction targets for small factories in 10 polluting industries and shutting down the small factories that fail to meet targets. The Water Ten Plan also requires plants in 10 major polluting industries to install particular abatement technologies. For instance, all paper and pulp factories in China are required to switch to Elemental Chlorine Free (ECF) or even Total Chlorine Free (TCF) bleaching technologies. Similar to the NRCP, China aims to install sewage treatment plants for 95 percent of cities and 85 percent of villages by 2020. According to the *People's Daily* (an official newspaper in China), the Chinese government estimates a budget of 2 trillion yuan ($330 billion) to tackle water pollution, and a large portion of the investment will come from the private sector (Huan 2015). Because the Water Ten Plan was passed recently, it may be possible to estimate the impacts, benefits and costs of it in the future as more actions have been implemented.

2.2. Market-Based Policies

In contrast to the prescriptive approaches described above, market-based policy instruments (MBIs) are decentralized, focusing on aggregate or market-level outcomes, such as total pollution levels or total emissions, rather than the activities of individual facilities. The classic economic prescription is to tax negative externalities, and to subsidize positive externalities, with the efficient tax (subsidy) equal to the marginal damages (benefits) at the efficient level of the externality (Pigou 1920, Baumol 1972, Sandmo 1976)[2]. In response to a tax, regulated firms have two choices for each unit of pollution they would have emitted in the absence of regulation – they can continue to emit that unit, paying the tax, or they can abate that unit, incurring costs to do so. Thus, each firm will reduce emissions just to the point at which the marginal cost of emissions abatement is equal to the unit tax on emissions. Since each firm equates the cost of abatement with the tax, marginal abatement costs are equal across firms, generating the least-cost allocation of emissions reductions. In the water pollution context, marginal damages from water pollution usually vary with the location of the source. In this context, an efficient water pollution tax would be differentiated by damages, either at the source level, or through the creation of tax "zones" where heterogeneity in damages is sufficiently coarse (Boyd 2003).

The standard Pigouvian results in economic theory address pollution from point sources; taxing nonpoint source (NPS) pollution is considerably more complicated.[3] However, this is an important set of pollution sources in the water context. Estimating firm-level emissions (let alone the damages from emissions) may be prohibitively costly, due to the presence of multiple small polluters to a single water body, information asymmetries, complex pollutant fate and transport processes, and stochastic environmental factors, such as weather (Suter et al. 2008). In this context, taxes can focus on emissions proxies for some sources (such as estimates of field losses of fertilizer or pesticide residuals), inputs to polluting processes (such as fertilizers, pesticides, urban impervious surfaces, and particular farming or forestry practices that affect runoff), and ambient concentrations of water pollutants (Shortle and Horan 2001). Taxes on inputs and other NPS emissions proxies may be relatively straightforward, but somewhat removed from the actual damages from pollution (through concentrations and exposure). Segerson (1988) developed a theoretical policy instrument for NPS pollution control which assigns penalties for pollution damages (or rewards for abatement) to firms based on ambient water quality.[4]

While environmental taxation approaches had been proposed since the early part of the last century, a seminal but much later contribution by Ronald Coase recognized the fundamental symmetry of externalities. Coase (1960) noted that the direction of compensation was not prescribed by the efficiency criterion, and that the mere existence of externalities in a market could, under certain very restrictive conditions, induce private negotiation of efficient outcomes.[5] Such outcomes are hindered in the real world by transaction (bargaining) costs and other barriers, especially with a large number of parties involved (Coase 1960). Nonetheless, this provided a new touchstone for the theory of efficient environmental regulation and highlighted the importance of property rights in the question of how private markets will tend to resolve environmental problems, if at all.

Coase's focus on property rights is linked to the development of systems of marketable pollution permits, known as "cap-and-trade" systems (though there are other variations on the same theme). In order for Coasian bargaining to occur, the government must assign and enforce property rights to pollution (Montgomery 1972; Dales 2002). The conceptual framework of emissions trading programs arising from these early contributions is well described in Tietenberg (2006). The regulator sets an aggregate cap on pollution and allocates the implied number of pollution permits to the regulated community, either by auction or a system of free allocation or "grandfathering," often based on past emissions. The pollution permits are transferable, and each firm will buy and sell permits based upon a comparison of market permit prices with its own marginal abatement costs (in the same way that a firm reacts to an emissions tax, except that the price is set by the market, rather than the regulator, who focuses, instead, on the quantity of allowable emissions). When the permit market clears, each firm has equated its own marginal pollution abatement cost with the prevailing permit price, resulting in equal marginal costs across firms – the least-cost allocation of control responsibility to meet the aggregate cap.

One of the most appealing aspects of cap-and-trade approaches is that, in theory, the ability of such a policy to arrive at the least-cost allocation of responsibility for emissions abatement across regulated firms is independent of the initial allocation of permits. This means, for example, that initial permit allocations may be manipulated to accomplish distributional outcomes that build sufficient political support for new or more stringent environmental regulations, or to meet income redistribution goals. There are some important exceptions to this general rule. First, market power in the permit market (a small number of permit sellers) can lead permit sellers to withhold permits and drive up permit prices, establishing a correlation between initial permit allocation and the ability of the cap-and-trade system to achieve least-cost aggregate abatement (Hahn 1984).[6] Marginal transaction costs that either increase or decrease with the size of permit trades may also establish such a correlation (Stavins 1995). In most cases, however, permits can be freely distributed in any fashion without interfering with the efficient functioning of emissions markets.

As noted in our discussion of pollution taxes, the marginal damages from water pollution can vary dramatically with the location of discharge, depending on the characteristics of receiving waters and other factors that influence the effectiveness of reductions made in different locations in a watershed. Establishing location-based trading ratios for each pair of polluters, which function like exchange rates, is an efficient approach in a cap-and-trade system for water pollution (Rodríguez 2000, Hung and Shaw 2005, Farrow et al. 2005, Konishi et al. 2015). Trading ratios reduce the theoretical cost savings from trading (relative to a uniform standard) — they are, after all, constraints on trade. However, getting trading ratios right can also increase the benefits of regulation, if high-damage sources also have high abatement costs (Boyd 2003). Thus, this is an important tradable permit policy design concern not just for cost-effectiveness, but for efficiency in the water pollution context (Lungarska and Jayet 2018).

Payments for ecosystem services (PES) are another class of market-based approach that can draw a direct line to the theory developed by Coase. Recent work counts 550 active PES programs around the world (Salzman et al. 2018). Some existing PES programs follow closely the bargaining model introduced by Coase, in which private individuals and/or firms (where property rights are well-defined and transaction costs low) negotiate solutions to externalities without government intervention - for example, Nestlé Waters' payments to farmers in the catchment area of its mineral water springs in France's Vittel area to change their land management practices and reduce nitrate transport to groundwater (Depres et al. 2008). Other PES programs involve an important deviation from Coase; governments execute contracts with private individuals or firms to pay for environmentally-friendly changes in behavior (similar to the structure of conditional cash transfer programs aimed at increasing other beneficial behaviors such as investments in health and education), as in Mexico's federal conservation payments program (Alix-Garcia et al. 2018). In either case, the basic theory is the same. Where externalities create a divergence between the party or parties who bear the costs of pollution control, and those who enjoy the benefits, incentives provided by contract can potentially resolve the problem. Where governments are the purchasers, they are providing public goods through negotiated contracts with private parties.

Cost-effective PES policies are not prescriptive; contracts should be based to the extent possible on the water pollution outcome of interest (for example, ambient concentrations of particular pollutants at particular locations), leaving the method of pollution reductions up to the recipient (Jack et al. 2008). In practice, this may be difficult to achieve in the water pollution context, especially in developing countries, given sparse models and data linking specific activities (such as agricultural land management practices) to ambient water pollution concentrations.

When applied to water pollution control, PES approaches are known as payments for watershed services (PWS). There are several theoretical and practical challenges to this approach. As in other contexts, efficient PWS systems must account for spatial and intertemporal heterogeneity (Jack et al. 2008). That is, just as differential environmental taxes would be required for contexts in which the marginal damages from pollution vary over space and time, differential PWS would be required where the marginal benefits of pollution abatement vary over space and time. Also, in strictly private Coasian bargaining, policymakers may be essentially indifferent about distributional outcomes; when governments are the parties contracting with providers of ecosystem services, however, it is desirable to reduce the rents received by service providers that are due to information asymmetry (regulators' lack of information about the service costs). Policymakers and regulators may have limited budgets, and a cost-effective PWS program maximizes the "bang for the buck" from expenditures and avoids paying for abatement that would take place even without a PWS incentive. The literature suggests a range of approaches to reducing adverse selection, including auctions (Ferraro 2008).

PWS programs share an important theoretical downside to environmental subsidies, more generally, in that they may increase the returns to polluting activities; for example, paying farmers to preseve or install buffers between pastures and water courses may unintentionally increase returns to farming, leading to additional agricultural development in the long run (Jack et al. 2008). In addition, implementing a PWS program may simply push polluting behavior into areas not under contract; this "leakage" has been a concern for PES programs focusing on deforestation (Pattanayak et al. 2010). Unintended effects like these should be assessed when estimating the benefits and costs of PWS approaches. Like all environmental subsidies, PWS raise concerns about additionality – ensuring to the degree possible that payments purchase water-quality-improving behavior that would not have taken place in the absence of the incentive, and ideally (from a fiscal perspective) that payments do not exceed recipients' willingness to accept (Cason and Gangadharan 2004, Kirwan et al. 2005).[7] In addition, compared to ecosystem services like carbon sequestration – a common focus of PES programs – contracting for watershed protection and other water-oriented ecosystem services is more challenging due to what is often weaker evidence regarding the complex biophysical processes that mediate the influence of land management practices and other contractual activities on water quality (Bruijnzeel 2004). Finally, where PWS programs have dual purposes – for example, watershed protection and poverty alleviation or rural development – this may reduce both their effectiveness and cost-effectiveness in achieving water quality goals (Pattanayak et al. 2010).

Mandatory information disclosure policies, another market-based approach, may correct a type of market failure relevant to the environment – information asymmetry – or may simply provide information that allows consumers to more effectively express their preferences over how their consumption choices affect the provision of public goods in the marketplace. The literature identifies some hypothetical mechanisms through which information disclosure may improve environmental performance (Tietenberg 1998, Powers et al. 2011). For example, disclosure may affect consumers' demand for polluting firms' goods; firms' stock prices and their ability to hire and retain employees; private citizens' incentive to sue polluters; political support for more stringent pollution control standards or enforcement; and pressure from community groups and nongovernmental organizations. It may also provide new information to managers about plants' discharges and options for reducing them.

The set of market-based approaches described above: taxes, tradable permits, payments for watershed services, and mandatory information disclosure, comprise the set of policy instruments discussed in the sections that follow. Other market-based approaches exist, though they are outside the scope of this paper.[8]

2.3. Applications of Market-Based Water Pollution Policies in Industrialized Countries

2.3.1. Pollution taxes

According to the Organization for Economic Cooperation and Development (OECD), hundreds of environmental taxes had been applied in OECD countries by the mid-1990s (OECD 1995). However, most of these have focused on air pollution. For example, Sweden, Norway, Denmark, Italy, France, Switzerland, Spain, Poland and Finland either currently tax sulfur emissions or have done so in the past, either directly or through differentiated fuel taxes (Sterner 2003). Sweden, France, Italy, Spain and other industrialized countries tax nitrogen oxide emissions from industrial sources (Sterner 2003). In some cases, these European air pollution taxes have been causally linked with decreases in emissions (Sterner 2003).

Water pollution taxes are less common in industrialized countries. France has assessed fees on water pollution since at least the 1970s, with revenues earmarked for the construction of wastewater treatment plants and other water pollution abatement investments. The fees are primarily designed to raise revenue, not to induce pollution abatement, and they are set at a level so low that they may have had little effect on pollution (Glachant 2002, Boyd 2003).[9] Germany also has a system of water pollution fees, established in the 1970s, also characterized by low fees (Boyd 2003). The Netherlands assesses fees on heavy metals discharged to water from large enterprises, as well as organic discharges from urban and agricultural households, and industrial enterprises of all sizes (Stavins 2003). The revenues in the Dutch case, as in France and Germany, are earmarked for subsidizing improved wastewater treatment (Boyd 2003). An analysis of the Dutch water pollution fee system suggests that the fees are statistically associated with a steep decline in industrial water pollution emissions; econometric methods used in this relatively old study did not establish causality (Bressers 1988). In fact, to our knowledge, none of these multi-decadal European water pollution fee systems have been analyzed using modern econometric techniques. It may be useful to revisit these systems to develop robust estimates of their effectiveness.

2.3.2. Tradable permits

The experience with water quality trading in other industrialized countries is not much more extensive than experience with water pollution taxes, though there are active programs in Australia, Canada, and the United States (Selman et al. 2009). In the United States, while nearly three dozen water pollution trading programs have been established, many have seen no trading at all, and few are operating on a scale that could be considered economically significant (Fisher-Vanden and Olmstead 2013). While water quality trading holds substantial promise, many challenges remain to be worked out by economists and by environmental managers. These challenges involve both physical aspects of water pollution problems that require modifications to the typical structure of pollution trading as practiced for air quality, as well as constraints imposed by current regulatory approaches to water pollution control that limit market function, including the implied assignment of rights to pollute. For example, in the United States, agricultural sources of water pollution are essentially unregulated under the Clean Water Act, and these sources tend to have the least-cost abatement options for many remaining water pollution problems. Markets that do not incorporate agricultural sources are, thus, thin, and their potential gains from trade may be fairly small (Fisher-Vanden and Olmstead 2013).

We discuss a few specific, active water quality markets in industrialized countries in the paragraphs that follow. Unfortunately, none have been rigorously evaluated for either effectiveness or cost-effectiveness.[10] Since 2004, the Australian state of New South Wales has operated a trading program to control salinity in the Hunter River, where sources include irrigation return flows, brine disposal from coal mining, and water diversions for cooling in electricity generation. The trading scheme restricts saline discharges by coal mines and power plants to periods when river flows are high, and to amounts less than or equal to a facility's salinity credit allocation. If discharges exceed credits, participants may purchase credits from other facilities. We know of no comprehensive analysis of the impact of this permit trading program on water quality, though local regulators consider it successful, because exceedances of salinity caps in each river segment are uncommon (Environment Protection Authority, State of New South Wales 2018).

The State of Connecticut in the U.S. established the Long Island Sound Nitrogen Credit Exchange in 2002, with 79 municipal sewage treatment plants participating. Because source location affects the environmental impact of a unit of nitrogen discharged, the program uses a system of trading ratios based on geographic trading zones. Abatement cost differentials are generally driven by plant size, due to economies of scale in municipal sewage treatment. Though we know of no study establishing a causal link between this trading program in nutrient concentrations in Long Island Sound, the state environmental regulatory agency claims that it reduced the nitrogen load entering the Sound by 65 percent between 2002 and 2014 (Connecticut Department of Energy and Environmental Protection 2016).

Phosphorous trading on the Minnesota River in the United States is another example of an active water quality trading program. To address low dissolved oxygen levels caused by algae blooms related to high phosphorus concentrations, the Minnesota Pollution Control Agency issued in 2005 a single National Pollutant Discharge Elimination System permit (updated in 2009) for phosphorus discharges to the Minnesota River, applicable to 47 permitted sources – mostly municipal sewage treatment plants and a few industrial point sources (Minnesota Pollution Control Agency 2009). The general permit allows facilities to trade phosphorus abatement allocations, using a system of facility-specific trading ratios to account for spatial heterogeneity in marginal damages. We know of no analysis that establishes a causal link between this market-based program and water quality outcomes in the Minnesota River.

Though these programs do provide examples of water quality trading programs in industrialized countries, note that even these relatively successful markets are very thin relative to the counterparts in air pollution regulation. While it operated, the U.S. sulfur dioxide trading program mentioned earlier comprised approximately 2500 sources. In contrast, the numbers of trading participants in the examples above are: Hunter River (23 point sources), Long Island Sound (79 point sources), and Minnesota River (45 point sources) (Fisher-Vanden and Olmstead 2013). That is not to say that mutually beneficial trades between a small number of participants in a market do not benefit those participants. However, the potential welfare gains from both trading and ambient water quality improvements made possible by trading would likely be much larger where such markets operate at scale.

2.3.3. Payments for watershed services

PWS approaches have been implemented for water pollution control in many industrialized country contexts. Pure private negotiations over PWS may be uncommon; the Vittel example discussed in section 2.2 is a good example (Depres et al. 2008), but we have not identified others in the literature.[11]

In contrast, industrialized countries have experimented with PWS in which governments pay private individuals and firms for watershed services, at both small and large scales. On the smaller end of the spectrum, since the 1990s, New York City's water system has paid farmers and other private landowners in the Catskill-Delaware watershed west of the Hudson River for onsite pollution control measures to avoid very expensive filtration that would otherwise have been required for compliance with federal drinking water quality regulation (Appleton 2002). At a much larger scale, the U.S. Conservation Reserve Program (CRP), administered by the U.S. Department of Agriculture, in 2018 paid more than $1.8 billion to more than 300,000 U.S. farms for environmentally-beneficial practices on 22 million acres (USDA 2018). Given the scale of annual expenditures, the CRP is a major U.S. environmental policy. The CRP's long-term contracts do result in lasting agricultural land retirement (Roberts and Lubowski 2007), which likely generates water quality benefits. Water quality improvement is an important CRP goal, and the likelihood that enrollment of an acre of land will reduce water pollution (for example, by reducing erosion) contributes to the index that determines the annual per-acre payment. Causal estimates of the CRP's water quality impact are not available, however. Empirical work by the implementing agency suggests that the CRP generates tens of millions of dollars in recreational water quality benefits in the United States each year (Feather et al. 1999).

Other industrialized countries have also implemented PWS policies. For example, Australia's 2004 Catchment Care conservation auction paid landowners for riparian zone restoration (Connor et al. 2008). The UK has also implemented PES programs: the Environmentally Sensitive Areas Scheme, the Conservation Stewardship Scheme, and the Nitrate Sensitive Areas Scheme, some of which have water quality objectives (Dobbs and Pretty 2008). We know of no empirical estimates of the impacts of these programs on water quality.[12]

2.3.4. Information disclosure

Many information disclosure policies have been established and evaluated in industrialized countries. One of the most well-studied is the U.S. Toxics Release Inventory (TRI) program, established in the aftermath of a deadly chemical accident in Bhopal, India in 1984 at a Union Carbide facility. The TRI requires manufacturing firms to report annual chemical releases to air, water, and land to the U.S. EPA, which then publicly releases the information (U.S. Environmental Protection Agency 2003).

From the TRI's inception in 1986 to the mid-2000s, total annual releases of reportable toxic chemicals fell by nearly 50 percent, but the observed decrease in toxic releases has not been causally attributed to the TRI. Data are not available on chemical releases before the program began, or for unregulated facilities, making it difficult to determine whether the TRI actually decreased toxic releases (Bennear and Coglianese 2005). Event studies on outcomes other than environmental performance have found that firms whose TRI releases receive media coverage experience abnormal stock returns (Hamilton 1995), and that these firms reduce pollution in response (Khanna et al. 1998). Other researchers have tried to address the causality questions surrounding the TRI by testing for alternative explanations for the observed decreases in releases of listed chemicals. For example, facilities may have shifted to unlisted chemicals, or to chemicals with lower volume but higher toxicity (Greenstone 2003, Gamper-Rabindran and Swoboda 2006), or they may have reduced their use of listed chemicals below reporting thresholds (Bennear 2008). The accessibility of TRI information to the interested public varies across states, as some states have attempted to increase availability and salience of this information; state governments' production of reports analyzing and presenting the TRI data appear to lower the amount of TRI chemicals discharged (Bae et al. 2010). One event study finds no significant impact of TRI-reported emission reductions on local housing prices, casting some doubt on the public's awareness and understanding of emissions disclosure (Bui and Mayer 2003).

Another U.S. information disclosure program is the consumer confidence reporting requirement under the Safe Drinking Water Act (SDWA), which requires public drinking water systems to provide an annual report to their end-users. The report includes average concentrations of key drinking water contaminants, reports any violations of the 90+ maximum contaminant level standards under the SDWA, and explains any potential health implications of such violations. This reporting requirement appears to have reduced total violations by public drinking water systems in the U.S. state of Massachusetts by 30–44 percent (Bennear and Olmstead 2008), and had an even stronger effect on microbial violations. Preliminary results from a study of the disclosure program's national impacts indicate similar nationwide effects (Baker et al. 2018).[13]

Information disclosure policies also appear to have significant impacts in industrialized countries outside of the U.S. For example, the disclosure of environmental incidents appears to have strong negative effects on the market value of Canadian firms (Laplante and Lanoie 1994), though this analysis did not examine disclosure's impacts on pollution. The Republic of Korea has published monthly lists of facilities violating environmental regulations since the mid-1980s; one study suggests that investors on the Korean Stock Exchange react strongly to these disclosures — perhaps even more strongly than for similar events in Canada and the United States (Dasgupta et al. 2006).

2.4. Applications of Market-Based Water Pollution Policies in Developing Countries

2.4.1. Pollution taxes

China's pollution levy system is an example of a pollution tax in a developing country. The system has been in place since the early 1980s and generates a large amount of revenue, some of which funds the operation of local Environmental Protection Bureaus (EPBs) (Bluffstone 2003). Through 2003, the levy system initially required industrial plants to pay a fee only on the single pollutant for which the plant exceeded its standard by the greatest amount. Since a 2003 reform of the pollution levy system, plants are required to pay levies on the three pollutants with the greatest amount of exceedance, and the levy rates were increased dramatically. Even before the reform, studies used general equilibrium models to show that the pollution levy system has been effective in reducing emissions from Chinese industry (Jiang and McKibbin 2002; Wang and Wheeler 2003). Wang (2002) uses data on plant-level pollution expenditures and finds that industries are strongly responsive to pollution charges but are not responsive to CAC requirements. Recent analysis supports earlier conclusions that China's pollution levy system is effective at reducing water pollution, and suggests that doubling China's levy for wastewater dumping would avert 17,000 premature deaths from digestive cancers per year at a cost of about $500 million per year — a per-life cost that is a fraction of VSL estimates for China (Ebenstein 2012).

However, one study suggests that inspections may play a more significant role than the pollution levies, themselves, in reducing emissions (Dasgupta et al. 2001b). Since the levy system relies on self-reporting by firms, the local EPB performs inspections of polluting firms every few years and charges the firms a substantial fine if they are found underreporting. Lin (2013) uses data on plant-level pollution to show that these inspections increase plants' self-reported pollution by more than 3 percent. He also suggests that plants generally under-report their pollution and, more importantly, that inspections may be effective at verifying plants' reported pollution but may not be effective in reducing pollution in China.

A separate environmental tax, the "Pay for Permit" policy, has been applied to industrial COD emissions in the Lake Tai Basin, Jiangsu, China, since 2009. Participating firms must purchase a permit from the local government for each unit of expected COD emissions, with penalties for violations. A difference-in-differences study examining pollution discharges and abatement by participating vs. non-participating firms, before vs. after the policy, demonstrates that treated plants reduced emissions by about 40 percent in the first two years post-policy (He and Zhang 2018). This pilot program is interesting because it charges firms for every unit of pollution, a classic environmental tax, while the older pollution levy system charges penalties for emissions over maximum standards.

In 1993, the Colombian government adopted Law 99 to reform the country's legal basis for a national discharge fee system. The law mandated that CARs, a set of government-determined regional environmental regulatory authorities based on their ecological characteristics, develop emission inventories of all organic waste facilities within their borders. CARs must charge all polluters a fee per unit of biological oxygen demand (BOD) and total suspended solids (TSS). CARs monitor facilities' discharges and invoice them every six months. According to Colombia's environment ministry, nationwide BOD discharges from point sources covered in the program fell 27 percent and TSS discharges fell by 45 percent under the program (Blackman 2006). However, due to data limitations, no empirical tests have confirmed a causal link between these reductions and the emission discharge fees. In addition, the new discharge fees were accompanied by more effective permitting, monitoring, and enforcement, as well as increased transparency and accountability of CARs, which may be partially responsible for observed declines in water pollution (Blackman 2006). Note that Law 99 allows CARs to retain water pollution fee revenues, which creates an economic incentive to enforce the discharge fee program.

In the 1980s, Malaysia began charging a fee on BOD emissions from the palm oil industry; emissions below national standards are charged at a basic level, and those above the standards are taxed at ten times the basic rate (Bluffstone 2003). Unpublished studies suggest that the implementation of these fees reduced the BOD load very dramatically, even while palm oil production increased (Vincent 1993, Vincent et al. 1997).

2.4.2. Tradable permit policies

Given the very small number of well-functioning water quality trading programs in industrialized countries, and the relatively small scale of those programs, it is, perhaps, not surprising that we have found no examples of such a program in a developing country.

Tradable permit policies for air pollution are also uncommon in developing countries. China's emerging cap-and-trade system for carbon dioxide emissions is one example, though it is too early to evaluate that program's effectiveness. The City of Santiago, Chile established a tradable permit program to control air pollution (total suspended particulates) in the 1990s. Assessments of that program's effectiveness do not suggest that it actually reduced pollution, and it has been described as underdeveloped (Coria and Sterner 2010). For example: the cap may have been set too low by regulators, who lacked information about actual emissions prior to the program, reducing demand for permits; each bilateral trade has had to be reviewed by the government, increasing transaction costs and decreasing trading volumes; and trades are permanent, rather than annual, reducing permit supply (Montero et al. 2002).

2.4.3. Payments for watershed services

The literature contains many summaries of applications of PES in developing countries (see, for example, Pagiola et al. 2005, Bulte et al. 2008, Pattanayak et al. 2010, and the May 2008 special issue of *Ecological Economics*, vol. 65, issue 4). A full review of these programs is beyond the scope of this paper. Here, we focus on a few programs focusing on water quality for which available empirical assessments allow us to draw some conclusions about the effectiveness and cost-effectiveness of PWS.

In 2006, the government of Beijing, China began paying farmers upstream of Miyun Reservoir (its only surface water reservoir) to convert land from rice cultivation to dryland crops, with the dual goals of increasing water yield in the catchment and reducing nutrient flows into the reservoir.[14] A recent assessment of this PWS program suggests that it has been very successful, with an estimated benefit-cost ratio of 1.5, and net benefits flowing to both upstream service providers and downstream payees (Zheng et al. 2013). This program, known as Paddy Land-to-Dryland, involves annual payments from Beijing to farmers that amount to about 1.2 times the annual opportunity cost of conversion. By 2010, all rice fields upstream had converted to dryland crops (mostly corn). The estimated increase in water yield attributable to the program is equivalent to 5 percent of annual runoff in the reservoir between 2000 and 2009, and total nitrogen and total phosphorus concentrations have also been reduced significantly (Zheng et al. 2013).[15] Another PES program in China is the Grain for Green program, which pays rural farmers to replace cultivated crops on sloped land with tree seedlings to prevent erosion and lessen flooding. While this is the largest PES program in the developing world, it does not focus on water quality (Uchida et al. 2009).

Mexico's federal conservation payments program has many goals, including watershed protection and aquifer recharge; protecting forests to maintain their "hydrological services" is the program's main stated goal, and the program is financed by water user fees (Muñoz-Piña et al. 2008). The program has significantly reduced deforestation (Alix-Garcia et al. 2015, 2018), increased land-cover management activities (Alix-Garcia et al. 2018), and does not appear to have crowded out private environmental stewardship not incentivized by the program (Alix-Garcia et al. 2018). While increased forest cover attributable to the program may have improved water quality and/or availability, to our knowledge, there has been no rigorous assessment of these impacts. Eligibility is based, in part, on landowners' residence within areas where aquifers are over-exploited, and also focuses on cloud forest, which may play an important role in capturing water from fog during the dry season (Muñoz-Piña et al. 2008). However, the literature suggests that forests, generally, can be either net suppliers or net demanders of water (Pattanayak and Kramer 2001). Though it is well-studied relative to other PES programs, we cannot say whether Mexico's program has achieved its water-related goals.

Many smaller-scale PWS systems have been established in Latin America. For example, in Colombia's Chaina watershed in the eastern Andes, downstream water users pay upland farmers for changes in land-management practices that reduce soil compaction and erosion. Though no causal estimates of this small program's impacts are available, one study suggests that it has both reduced deforestation and regenerated riparian vegetation, which could both improve water quality (Moreno-Sanchez et al. 2012). PWS programs in Bolivia's Upper Los Negros watershed and Ecuador's Palahurco watershed have also been described in the literature (Pattanayak et al. 2010). Kosoy et al. (2007) provide case studies of three additional small-scale PWS programs in Honduras, Costa Rica and Nicaragua. To our knowledge, the water quality and quantity impacts of these Central and South American programs have not been rigorously evaluated.

A PWS program has been piloted in Tanzania's Urugulu Mountains, the upland catchment area for the basin that provides water for most of Dar es Salaam and surrounding regions. This is an interesting example due to its private origins; two international NGOs facilitated negotiations between upstream farmers and downstream local communities and firms, without a major national government role. While we discuss two examples of such arrangements in industrialized countries (the New York City and Vittel cases discussed earlier), we have not identified other examples of these arrangements in developing countries. The Equitable Payment for Watershed Services program engages upland farmers with downstream water utilities, beverage companies (including Coca-Cola) and breweries (Mussa and Mwakaje 2013). Studies of this program have examined its mostly positive impacts on farmers' livelihoods (Mussa and Mwakaje 2013), and anecdotal reports suggest that water quality has improved (Branca et al. 2011).[16] However, to our knowledge, no causal study on this PWS program has been published.[17,18]

2.4.4. Information disclosure policies

Information disclosure programs have been used in developing countries as alternative environmental policy instruments in the past few decades, though only a few have been rigorously evaluated. Empirical evidence suggests that disclosure can significantly reduce pollution from firms. More generally, citizen complaints targeted at specific firms do appear to decrease firms' value (and positive environmental news appears to increase firms' value) in capital markets in some developing countries – for example, Argentina, Chile, Mexico and the Philippines (Dasgupta et al. 2001a).

Indonesia's Program for Pollution Control, Evaluation and Rating (PROPER) was established by Indonesia's National Pollution Control Agency (BAPEDAL) in 1995 to rate and disclose the environmental performance of factories (Tietenberg 1998). Factories are color-rated (black, red, blue, green and gold, where black labels the worst performers and gold labels the best performers) based on BAPEDAL's evaluation of their performance. Evaluations are based on pre-existing survey data, plant-level self-reported data, and inspections conducted by BAPEDAL. Evaluations of PROPER have concluded that the program has reduced pollution. Using descriptive summary statistics of PROPER ratings for plants, past studies find that PROPER has a short-term impact on improving below-average firms' performance but does not increase the number of firms that use more than required environmental management technologies (Tietenberg 1998, Blackman et al. 2004). Garcia et al. (2007, 2009) use panel data on BOD and

COD emissions from about 140 firms, estimating firm fixed-effects models to evaluate the PROPER's effectiveness. They find that PROPER does reduce firm pollution emissions, especially for low-compliance firms.[19]

India's Green Rating Project (GRP) is another information disclosure program in a developing country. It began in 1997 and is run by one of India's most influential environmental NGOs, the Centre for Science and Environment (CSE). CSE evaluates the environmental performance of large industrial plants in India, assigns numeric ratings to these plants and awards them "leaves" depending on their score. It also informs the public about the leaves rating and offers plants information about their pollution abatement options. Powers et al. (2011) use plant-level survey data on COD and TSS concentrations to evaluate the impact of the GRP on discharges from India's largest pulp and paper plants. They find that the GRP significantly reduced pollution among dirty plants.

Overall, information disclosure policies in developing countries seem to be effective in reducing firms' water pollution discharges. The effectiveness of information disclosure policies should be interpreted with caution, however, since empirical studies are mostly performed on firm-reported discharge data. The success of disclosure programs at achieving emission reductions in developing countries may be puzzling given the presumption that the demand for environmental quality in poor countries is relatively low (García et al. 2009). Some mechanisms tested in the literature include the environmental audit mechanism and community demand. Blackman et al. (2004) conducted a plant-level survey of Indonesian firms covered by PROPER. The survey results suggest that an important channel through which PROPER improves firms' performance is by improving managers' understanding of their plants' emissions and abatement opportunities (the "environmental audit" effect). However, providing managers information without making that information public may not be sufficient to induce pollution reduction (Blackman et al. 2004). Garcia et al. (2007, 2009) and Powers et al. (2011) all suggest that plants in wealthier communities are more responsive to information disclosure – results consistent with findings in developed country contexts. However, the ways in which wealthy communities in developing countries may exert pressure on firms to reduce pollution is an area needing additional research.

Environmental information disclosed to households in developing countries also appears to facilitate better household decisions. For example, field experiments in Bangladesh and India find significant responses by households to information about drinking water wells' arsenic levels (Madajewicz et al. 2007, Barnwal et al. 2017). Knowledge that a well has an unsafe arsenic concentration increases the probability that a household switches to another well within one year by 0.37 (Madajewicz et al. 2007). However, if the information is only presented to respondents as one-time verbal information, it does not induce households to switch to safe water sources (Bennear et al. 2013).

2.5. Voluntary Approaches and Informal Regulation

Voluntary approaches (VAs), also referred to as voluntary environmental programs, are alternative policy tools that do not fall into either the market-based category or the prescriptive category. Regulators either offer polluters incentives (cost-sharing programs, or environmental leadership programs) to reduce pollution or induce participation by threatening stricter regulation if a VA is not adopted

(Borck et al. 2008; Borck and Coglianese 2009). The literature provides several explanations for how VAs might reduce pollution (McGuire et al. 2018).

Advantages of voluntary approaches over traditional regulations include: (1) potential cost savings to achieve environmental targets, since polluters have the flexibility to choose abatement techniques (as under a market-based approach), and they will choose a cost-effective technology; (2) increased cooperation and communication between polluters and regulators (Alberini and Segerson 2002). However, the obvious potential downside is that firms may not engage in costly pollution reduction without requirements, monitoring and enforcement. In fact, empirical analyses of such programs must worry about selection, since the firms for whom it is least costly are most likely to join a voluntary environmental program.

The literature contains theoretical models on the effectiveness of VAs in comparison with mandatory policies (Segerson and Miceli 1998; Wu and Babcock 1999; Segerson and Wu 2006). Assuming voluntary regulations are less costly than mandatory regulations, Segerson and Miceli (1998) use a model with one polluter and one regulator to show that VAs are always socially desirable and can improve environmental quality when: (1) government has more bargaining power than firms, (2) the magnitude of background threat is high, and (3) social cost of government funds is small. Manzini and Mariotti (2003) adapt Segerson and Miceli (1998)'s model to a group of heterogeneous polluters. Their model yields an interesting result that the equilibrium outcome is determined by the bilateral negotiation between the regulator and the firm with the toughest attitude towards abatement.

In theory, VAs can be useful in reducing ambient water pollution. In the US, 60 percent of nonpoint source pollution comes from agriculture and various voluntary programs have been established control nonpoint source pollution (Bosch et al. 1995, U.S. Environmental Protection Agency 1996, Segerson and Wu 2006, Fleming 2017). Segerson and Wu (2006) develop a theoretical model that analyzes the use of VAs in conjunction with an ambient tax to control nonpoint source pollution. The tax is used if the voluntary approach fails to meet a pre-specified goal. They find with the threat of a tax, the voluntary approach may be sufficient to induce compliance.The theory literature discussed above assumes that industries are motivated to negotiate with regulators due to background legislative threats. Since developing countries often have weak regulatory capacities, existing theoretical models may not be as applicable to VAs in developing countries. Blackman et al. (2006) develop a game-theoretic model to dynamically examine the effect of VAs on pollution abatement when regulatory capacity is weak. Their model assumes that no penalties are applied to industries for failing to abate pollution in the short term given weak enforcement at both the national and local levels. But due to the probability that national environmental regulatory capacity will develop, polluting firms may face more stringent penalties in the long term. If VAs allow for a significant increase in penalties in the long run, this outweighs the low probability of enforcement, so firms are likely to adopt abatement technologies under VAs.

2.5.1. Applications of VAs in industrialized countries

Voluntary approaches have been applied in several cases in the United States, Europe, and Japan at both the federal and state level (Daley 2007, Morgenstern and Pizer 2007).[20] In the United States, VAs are used to improve environmental quality in areas as climate change, air quality, transportation, and water pollution reduction (Lyon and Maxwell 2007, U.S. Environmental Protection Agency 2015). Relatively little is known about the actual effectiveness of VA programs (Koehler 2007). One well-studied VA is the U.S. 33/50 program initiated in 1988 to reduce emissions of 17 chemicals to air, soil, and water by 33 percent by 1992 and 50 percent by 1995. Evidence of 33/50's effectiveness is mixed. Vidovic and Khanna (2012) compare TRI emissions (to avoid using the voluntarily-reported 33/50 emissions data as the outcome variable) by firms enrolled in 33/50 with those of firms not enrolled; they find no statistically significant decrease in pollution attributable to 33/50. In contrast, Khanna and Damon (1999) and Innes and Sam (2008) attribute significant reductions in 33/50 releases to participation in the program. Overall, it is not clear if VAs make positive contributions to water quality improvements, or how they compare with mandatory regulations (Khanna 2001, Borck and Coglianese 2009).

2.5.2 Applications of VAs in developing countries

Blackman et al. (2010) provide a review of the empirical evidence of VAs' impacts in developing countries. Several past studies have studied VAs in developing countries including Chile, Mexico, Colombia, China, the Czech Republic and Brazil (Blackman et al. 2006, Blackman and Sisto 2006, Jiménez 2007, Hu 2007, Blackman et al. 2010, Blackman et al. 2013). Only the study on Chile provides credible evidence of the effectiveness of a VA in improving environmental outcomes (Jiménez 2007).

Unlike in developed countries, where VAs are used to encourage firms to over-comply with mandatory regulations, regulators in developing countries tend to use voluntary programs to boost compliance with existing ill-defined, incomplete and infrequently-enforced command-and-control environmental policies (Blackman and Sisto 2006). Blackman et al. (2010) use plant-level data on more than 100,000 facilities to analyze the effectiveness of the Clean Industry Program in Mexico. Their analysis suggests that dirty firms recently punished by the government are more likely to participate in the Clean Industry Program. But after industries graduate from the program, participants do not have substantially less pollution than matched nonparticipants. Blackman et al. (2013) find a similar result that VAs had minimal immediate effects on firms' environmental performance in Colombia.

Though empirical studies often fail to establish immediate effects of VAs on environmental performance in developing countries, the literature suggests that VAs may facilitate capacity-building in both government institutions and the private sector (Blackman et al. 2013). For instance, in the Colombian case, regulators lack sector-specific technology information to implement traditional regulations. The government obtains such data from firms that participate in VAs, establishing baseline data that can be used to develop regulations.

Similar to VAs, informal regulations imposed by local communities can also supplement formal policy instruments to boost industries' compliance with existing mandatory policies. In developing countries where formal regulations are weak or absent, informal regulations can be achieved when communities bargain with local industries and supply environmental resources at raised prices as pollution damages increase (Pargal and Wheeler 1996). In Indonesia, because of informal regulation, water pollution-intensive plants are less common in new facilities, private enterprises, and high-income, well-educated areas.

2.6. Funding, Subsidizing and Incentivizing Infrastructure Investment

An extensive literature suggests that provision of safe drinking water and sanitation benefits households in many different ways. Reduced incidence of diarrheal disease, with associated reductions in infant and early childhood mortality, is one key benefit (Galiani et al. 2005, Geruso and Spears 2018), as are increased early human capital development and potential long-run labor-market outcomes (Zhang 2012, Spears 2012). Having access to water at home also saves time collecting water; studies show the time-use change significantly increases female literacy (Sekhri 2013), social integration and self-reported well-being (Devoto et al. 2012). Many other benefits have been empirically demonstrated.[21]

Inadequate sanitation is also linked with ambient water pollution. That is, major investments in sanitation infrastructure can also impact human and ecosystem health by improving ambient water quality. Untreated waste is a classic negative externality, and centralized wastewater collection and treatment is a classic public good. Most positive externalities of piped water are likely to be internalized within a region, but wastewater treatment generates spillover benefits to downstream regions (Chiang 2016). There is a strong economic justification for government provision, and large-scale investments in sanitation infrastructure are common. For example, since 1972, the cost of wastewater treatment projects funded by U.S. CWA grants has been about $680 billion ($2014) (Keiser and Shapiro 2018).

2.6.1 Industrialized country empirical applications

Past literature has emphasized the role of water treatment and sewerage systems to the decline in mortality rates in U.S. and European cities in the early 20th century. For example, Alsan and Goldin (2018) find clean water and sewage infrastructure investments are complementary interventions and together accounted for one-third of the decline in log child mortality in the United States from 1880-1920. The provision of piped, treated drinking water in major American cities during the early 20th century resulted in large reductions in urban mortality, with an estimated social rate of return to infrastructure investments of 23 to 1 (Cutler and Miller 2005). Delaney et al. (2011) find similarly impressive reductions in infant mortality in Ireland post-WWII as a result of improved sanitation. In 1959, the US Sanitation Facilities Construction (SFC) Act funded the provision of water and sanitation infrastructure on Native American reservations. From 1960 to 1998, the infant mortality gap between Native Americans and White Americans decreased from 27 to 3 per 1000. Watson (2006) attributes 40 percent of the convergence since 1970 to sanitation investments.

However, sanitation investments targeting ambient water quality improvements (where they do not directly affect health) are less likely to be net beneficial. Keiser and Shapiro (2018) use project funding, water quality and average housing price data in the US to estimate the benefits and costs of the CWA. They find that the average municipal sewage treatment grant project cost is about $35 million while the estimated effect of a grant on housing prices near affected rivers is about $9 million (in 2014 US dollars). Since the hedonic property model they use only captures part of the benefits of CWA projects, failing to account for recreational and other values, a more comprehensive estimation of municipal sewage treatment project benefits may be useful. Note, also, that municipal sewage treatment in the United States and other industrialized countries is relatively advanced. By the time of the CWA, for example, many federal investments in improving wastewater treatment would have raised treatment from primary to secondary, or secondary to tertiary treatment — upgrades with higher marginal abatement costs and lower marginal benefits than establishing primary treatment (a situation perhaps more relevant to many developing country contexts). There is some evidence that in industrialized settings, further reducing remaining water quality contaminants in relatively advanced municipal sewage treatment may not be as cost-effective as other means of improving ambient water quality, such as reducing agricultural non-point source pollution (van der Veeren and Tol 2001, Shortle 2017).

2.6.2 Developing country empirical applications

For developing countries, the literature also shows positive impacts of toilets, latrines, and safe drinking water on health and educational outcomes. Improvements in access to piped drinking water and sanitation in Brazil between 1970 and 2000 resulted in a combined welfare gain of $10,300 per capita, explaining 22 percent of within-municipality variation in life expectancy over the period (Soares 2007). Duflo et al. (2015) estimate the impact of an integrated water and sanitation improvement program in rural India that provided household-level water connections, latrines, and bathing facilities to all households in approximately 100 villages. They find that the intervention reduced diarrhea episodes by 30-50 percent. Drinking water provision also has significant impacts on the health of both adults and children (Zhang 2012) and individuals' educational attainment in China (Zhang and Xu 2016).

As in industrialized countries, existing evidence of the impacts of large-scale infrastructure investments on ambient water quality is much harder to find. As noted earlier, Greenstone and Hanna (2014) find the National River Conservation Program (NRCP), which focuses on investments in wastewater collection and treatment (as well as community toilets, crematoria and public education), has had no significant impact on ambient water quality.[22]

2.7. Environmental Liability Policies

Liability rules that internalize the external costs of ambient water pollution are also quite common. However, transaction costs may be high relative to administrative regulations such as standards, taxes, and permits (Acton and Dixon 1992; Dixon, Drezner, and Hammitt 1993). Liability rules are often designed to support a "polluter pays principle," though due to the economic incidence of liability funding, polluters may not really pay (Probst et al. 1995). In theory, a mix of administrative regulation and liability rules for water pollution control may be optimal (Shavell 1984; Kolstad, Ulen, and Johnson 1990). Joint-and-several liability rules for NPS pollution, with implications similar to those of ambient pollution taxes, have been considered in theory (Miceli and Segerson 1991).[23]

Studies also suggest environmental liability can induce abatement technology diffusion and adoption, and there exists a reversed U-shape relationship between the stringency of policies and technology diffusion (Endres and Fiehe 2011, Perino and Requate 2012, Friehe and Langlais 2017).

2.7.1 Industrialized country examples

The 1977 Amendments to the CWA established strict liability of polluters for the discharge of oil and other hazardous substances into U.S. navigable waters. The liability limits were subsequently raised and responsibility for natural resource damages expanded by the 1980 Comprehensive Environmental

Response, Compensation, and Liability Act (known as Superfund), the 1990 Oil Pollution Act, and the 2000 National Marine Sanctuaries Act (Boyd 2004). Liability for oil spills in U.S. waters is potentially unlimited under the Oil Pollution Act, which was passed in the aftermath of the Exxon Valdez oil spill off the Alaskan coast (Kim 2002). This unlimited liability has resulted in avoidance behavior among regulated firms, such as the formation of single-vessel corporations, so as to reduce the potential costs of a spill. Much work has been done in the area of law and economics on the effects of liability rules under Superfund and related state laws, though not with respect to water resources per se (Kornhauser and Revesz 1994; Sigman 1998; Chang and Sigman 2000, 2007; Alberini and Austin 2002). There is some evidence that strict liability regimes reduce unexpected pollution releases to the environment. Firms have, in some cases, developed behavioral responses to avoid liability, though at least one study has demonstrated that, unlike under the Oil Pollution Act, divestiture into smaller firms is not the mechanism through which this is achieved (Alberini and Austin 2002).

This research on environmental liability regimes became even more policy-relevant with the European Union's adoption of the Environmental Liability Directive (ELD) in 2004 (with member states incorporating the directive into national law in subsequent years). The ELD holds polluters strictly responsible for the environmental damage they cause to water, soil,and protected species and habitats and requires public authorities to ensure that polluters restore damaged natural resources. The focus on natural resource damage remediation was new to many EU member states (Winter et al. 2008). Repairs may be done either by the polluter, or by public agencies that can then attempt to recover costs from polluters. Whether a polluter or the public sector actually does the repairs may be an important distinction; in the United States, the EPA tends to choose less extensive environmental remedies under Super fund when firms are expected to bear a greater share of the costs (Sigman 1998). We know of no empirical analysis of the impacts of the ELD on ambient water pollution or other outcomes.[24]

Notes

1. There are some papers that examine, prospectively, the relative cost-effectiveness of various compliance approaches in particular sectors, in particular watersheds. For example, see Fezzi et al. (2010). The WFD has been criticized for ineffectiveness, since the number of surface water bodies in "good" condition increased by only 10 percent in the first WFD cycle (2009-2015) (van Rijswick and Backes 2015). However, careful econometric analysis would be required to determine whether the WFD, itself, has reduced water pollution in member states, and by how much.

2. Note that, in the dynamic context, subsidies can lead to excessive entry into the subsidized industry, and are, thus, not truly equivalent to taxes in their ability to achieve efficient externality management (Baumol and Oates 1988).

3. Nonpoint source water pollution is the largest source of water pollution in the U.S., causing 40 percent of remaining impairment in lakes, rivers, and estuaries (U.S. Environmental Protection Agency 1996). It occurs when rainfall and melting snow create urban and agricultural runoff that deposits pollution into water bodies.

4. This work has motivated many extensions (Xepapadeas 1991, Xepapadeas 1992, Herriges *et al*. 1994, Horan *et al*. 1998, Hansen 1998).

5. Of course, the outcome of Coasian bargaining - who compensates whom - has important distributional implications, even if it does not affect efficiency.

6. Small trading volume in at least one permit market, that for total suspended particulates in Santiago, Chile, has been attributed to market concentration (Montero *et al*. 2002).

7. Where poverty alleviation is an important objective of a PWS (or other PES) program, as is the case in some developing countries, this last concern may be less pressing.

8. For example, additional market-based approaches include the reduction or elimination of environmentally-damaging subsidies, and deposit-refund systems (Stavins 2003).

9. One study suggests that French water pollution fees from 1986-1992 were approximately one-half the level of estimated social damages (Thomas 1995).

10. For a more comprehensive review of such programs, see Fisher-Vanden and Olmstead (2013).

11. There are other examples of pure private PES for non-water environmental services in industrialized countries. For example, the NGO Defenders of Wildlife pays ranchers near Yellowstone National Park in the United States for livestock predation by wolves, reducing local landowners' opposition to wolf reintroduction in national parks. See Anderson and Libecap (2014) for many other examples.

12. Prior research has noted that the CRP and other U.S. agri-environmental programs are more targeted to environmental goals than their European counterparts, and that European programs are more focused on income transfers to farmers than on reducing agriculture's environmental impacts (Baylis et al. 2008).

13. Disclosure of violations under the SDWA may also have caused households to engage in averting behavior, substituting bottled water for tap water in public water systems with violations (Graff Zivin et al. 2011).

14. Beijing's Guangting Reservoir closed in 1997 due to heavy pollution from wastewater discharges, fertilizers and pesticides (Zheng et al. 2013).

15. Fertilizer applications actually increased due to the program, but because of the removal of flood irrigation, transport of nutrients to the reservoir decreased (Zheng et al. 2013). However, this unintended side effect has diminished the program's benefits and may have other external impacts, for example as a threat to groundwater upstream of the reservoir, and as a source of additional greenhouse gas emissions.

16. Note that water quality monitoring is taking place for this program, suggesting that its impacts are at least potentially measurable (Branca et al. 2011).

17. It is not clear whether this program is still operating; one source notes that it was initiated in 2006 and designed to end in 2012 (Branca et al. 2011). A more recent source suggests that it began in 2009 and does not mention an end date (Mussa and Mwakaje 2013).

18. An additional African PWS program is South Africa's Working for Water program, established in 1995 (Turpie et al. 2008). This is a large program with a unique design; private individuals (not landowners) are paid to restore public and private lands in water supply catchments. However, the program focuses on increasing water supply in water-scarce regions (by eliminating invasive species) - not on water pollution control.

19. However, firms seem to underreport actual pollutant concentrations, so results should be interpreted with caution (García et al. 2007).

20. Khanna (2001), Alberini and Sergerson (2002) and Borck and Coglianese (2009) all give good reviews on the empirical literature evaluating the effectiveness of VAs.

21. A review of the full literature on the economic benefits of clean drinking water and sanitation is beyond the scope of this study. See Ahuja et al. (2010) for a useful review of studies on drinking water.

22. Compensating behavior has been observed in the context of household drinking water and sanitation investments (Bennett 2012, Jessoe 2013). It is possible that similar behavior might exist among polluting firms and households near new sanitation infrastructure, especially because they do not bear the full costs of pollution that will move downstream. This could be one reason for the null finding in Greenstone and Hanna (2014). More spatially-disaggregated analysis may be needed to assess whether compensating behavior and spillovers interact to reduce the ambient water quality impacts of sanitation infrastructure investments.

23. Under joint-and-several liability, losses may be pursued from any single party. Polluters held liable (de-fendants in such cases) can pursue payment from other defendants, but a harmed party may recover all damages from any single defendant, regardless of their individual share of the liability.

24. The primary compensating regime used in ELD favors physical remediation of resources or services but does not allow estimation of social values. Past research has demonstrated a role for more comprehensive benefit valuation within the ELD. For example, using a case study of a toxic spill damaging Doñana National Park in Spain in 1998, Martin-Ortega et al. (2011) show that under the ELD, in-kind compensation may be insufficient to offset welfare losses without accounting for non-use values. Similarly, Bullock and O'Shea (2016) use the case of pollution in the River Suir, Ireland and suggest that ignoring social values associated with water, such as amenity values, will result in inadequate estimation of costs and benefits.

3. Challenges of Water Pollution Control in Developing Countries

3.1. Challenge of Data Availability in Developing Countries

One primary challenge to implementing and assessing the effectiveness of water pollution control policies in developing countries is data availability. However, this challenge is increasingly being overcome by the availability of satellite data measuring pollution, though more often for environmental problems such as air pollution, deforestation and groundwater depletion (Foster et al. 2009, Jayachandran 2009, Alix-Garcia et al. 2013, van der Meer et al. 2012) than for water pollution (Schaeffer et al. 2012).

A dearth of data on ambient water pollution concentrations can also be partly overcome by exploiting variation in water pollution that results from exogenous shocks (policy interventions, natural events) that have direct connections to ambient pollution concentrations, but do not require their direct measurement. An example of this approach examining water pollution impacts is Do et al. (2018), in which the authors exploit variation in industrial pollution on India's Ganga River due to Supreme Court rulings that mandated pollution reductions from the tanning industry to estimate the impacts of pollution reductions on neonatal mortality, without using any direct measures of river water quality.[1]

Field experiments in which researchers collect their own data on environmental or health outcomes in the context of a randomized control trial have proven to be another very valuable tool for overcoming the problem of inadequate observational data to examine the impacts of drinking water and sanitation interventions (e.g., Kremer et al. 2011). Randomized control trials are unlikely to be widely applicable to examining the impacts of ambient water pollution, however. Ambient pollution concentrations may not be as easily or cost-effectively reduced on a scale that is large enough to support statistical analysis of the impacts of such interventions. However, randomized experiments can be and have been used to examine the impacts of specific regulatory interventions on the emissions behavior of individual firms or households, which certainly contributes to ambient water pollution concentrations (e.g., Duflo et al. 2013). Connecting the dots between these relatively small-scale causal estimates and large-scale pollution reductions (and their impacts) would be challenging.[2]

Innovative use of available technologies, increased monitoring frequency, and third-party monitoring could improve data collection in developing countries. A recent example is China's automatic water quality monitoring system. Prior work has found evidence of environmental data manipulation by local governments in China, thus data quality is a significant concern for studies of pollution China (Ghanem and Zhang 2014). Beginning in 1999, China's Ministry of Ecology and Environment (MEE) (formerly known as the Bureau of Environmental Protection) has collected data on major rivers and lakes using an automatic water quality monitoring system. Monitoring stations operated by the central government collect and publish real-time water quality data online automatically.[3] Both central and local governments have limited capability to manipulate the data collected, compared to prior monitoring regimes. The automatic monitoring system also significantly improves the frequency of water quality monitoring. Traditional water quality data in China are collected once a month. In contrast, the automatic reporting system can collect and report data every 4 hours. Recently, China's central government expanded the automatic water quality monitoring system from 100 to 2,050 stations across the country to improve data availability and credibility (Xinhua Net, 2018).

Third-party monitoring can also improve water quality data in developing countries. Duflo et al. (2013) provide empirical evidence on the importance of third-party monitoring and corruption in water pollution data collection. This study shows that in India, having a third-party auditor who is externally selected and paid causes more truthful reporting of firms' water pollution emissions.

Citizen science can also increase the scope of water quality data collection by governments that have limited resources. In the United States, the EPA supports multiple volunteer water quality monitoring programs throughout the country, including the Equipment Loan Program for Citizen Science Water Monitoring Program (U.S. Environmental Protection Agency 2016). We know of no empirical papers that use water quality data collected through citizen science initiatives, however, Kolstoe and Cameron (2017) use the eBird dataset (a citizen science project from Cornell University) to evaluate willingness to pay for preservation of bird species. Moreover, citizen science and other crowdsourcing approaches may also provide opportunities to educate, engage and empower the public to protect the environment. The literature provides theoretical background on crowdsourcing water quality data (see Borden et al. (2016) for a review). In developing countries, with the increasing availability of mobile phones and internet access, citizen scientists could make a significant contribution to future water quality data collection.

3.2. Imperfect Monitoring, Enforcement, and Compliance

There is a rich literature in environmental economics on the monitoring and enforcement of pollution control regulations, and the role of stringency in affecting outcomes (such as emissions, or air and water quality). A full review of this literature is beyond the scope of this paper, but is available in Shimshack (2014). Theoretical models of firms' response to monitoring and enforcement derive from an older, broader literature on the economics of crime and law enforcement (Becker 1968). Firms consider environmental regulatory compliance in a benefit-cost framework, minimizing the sum of abatement costs and expected enforcement penalties, which are related to both the probability of detection and the magnitude of the penalty for noncompliance (Shimshack 2014, Dasgupta et al. 2000). From the government's perspective, the regulator must balance the cost of monitoring against the benefit of detecting environmental harms from noncompliance.

3.2.1. Industrialized country empirical applications

Dozens of studies in the economics literature establish that monitoring and enforcement of environmental regulations reduces pollution, deters future violations, and even encourages over-compliance by regulated entities (Shimshack 2014). Most of the studies that demonstrate these impacts examine monitoring and enforcement of U.S. environmental regulations, at the state and federal level, perhaps due to both data availability and a long history of pollution control regulation. Below, we summarize a few relevant examples with respect to water pollution from Shimshack (2014).

The pulp and paper industry is one of the most heavily-regulated industries in North America, as it creates a significant amount of air and water pollution. Thus, many studies have examined the impacts of regulatory monitoring and enforcement on behavior in this industry. During the 1980s, U.S. and Canadian pulp and paper firms appear to have reduced water pollution in response to both inspections and increased threats of inspections (Magat & Viscusi 1990, Laplante & Rilstone 1996). More recently, pulp and paper plants increased compliance and decreased water pollution emissions in response to formal enforcement actions with monetary penalties during the 1990s and 2000s (Shimshack & Ward 2005, 2008). The responsiveness of U.S. municipal wastewater treatment plants to monitoring and enforcement has also been examined extensively. Even though these are generally public entities, they also appear to reduce pollution in response to sanctions and federal fines during the 1990s (Earnhart 2004a, 2004b). Chemical facilities also increased compliance in response to sanctions and fines during the late 1990s and early 2000s (Glicksman & Earnhart 2007).

The studies summarized above consider the impacts of *specific* deterrence, investigating the impacts of increased monitoring or enforcement on the directly-affected facilities. Monitoring and enforcement also have *general* deterrence effects; their impacts spill over to other firms in an industry or region, affecting their compliance behavior indirectly. For example, Shimshack and Ward (2005) find that the general deterrence effects of monetary penalties for U.S. CWA violations by U.S. pulp and paper firms in the 1990s were at least as large as the specific deterrence effects. Notably, economists have also found that private enforcement actions, such as citizen suits, have general deterrence effects. For example, CWA violations at wastewater treatment facilities fall after a facility in the same state is sued by private citizens (Langpap and Shimshack 2010).

3.2.2. Developing country empirical applications

A small number of studies on monitoring and enforcement of environmental regulations in developing countries demonstrate empirically their important role in increasing compliance. For example, increased government inspections appear to reduce air and water pollution by Chinese manufacturing firms, while increases in pollution levies do not appear to do so (Dasgupta et al. 2001b). Wang (2002) shows that Chinese pollution levies (penalties for noncompliance with water pollution standards) increase expenditures on end-of-pipe wastewater treatment by industrial polluters. The financial penalties for noncompliance in China's pollution levy system also appear to have significant positive effects in Wang and Wheeler (2003), who demonstrate that the levies' impacts are higher in areas of the country where regulatory institutions are stronger.

Given the extensive evidence on the importance of monitoring and enforcement in determining the effectiveness of environmental regulation, many studies have examined the challenges that developing countries face when their capacity for stringent monitoring and enforcement is weak (Afsah and Makarim 1999; Wang and Wheeler 1999; Dasgupta et al. 2000). It is not unusual for developing countries to have environmental standards that are actually quite stringent, but that are simply not met due to weak enforcement and compliance (Greenstone and Jack 2015). For example, Duflo et al. (2013) use a field experiment to demonstrate that corruption dramatically reduces the effectiveness of industrial emissions standards in Gujarat, India and that improving the incentives of firms and third-party auditors in this system reduces emissions.

Zhang et al. (2018) assess a recent Chinese effort to better link the incentives of decentralized environmental managers with national objectives — the National Specially Monitored Firms (NSMF) pilot program, established in 2007. This program identified specific firms that were major emitters of air and water pollution (as well as hazardous solid waste) and provided centralized oversight of the local monitoring of these firms; all NSMFs are required to install automatic monitoring systems and transmit emissions information in real time to the central government, which verifies the accuracy of the data through monthly inspections. Local governments receive the information from the central government, and regulated firms' pollution levies are determined jointly by the central and local authorities. Using a regression discontinuity design, Zhang et al. (2018) find that the additional central supervision of local authorities in the NSMF program reduces industrial COD emissions by 26.8 percent in the first year, with continuing reductions in later years.

The literature has also empirically examined the determinants of imperfect monitoring and enforcement in developing countries. For instance, Wang et al. (2003) investigate the bargaining power of industries with local authorities pertaining to the enforcement of pollution levies in China. They show that state-owned enterprises and firms with more adverse financial situations have more bargaining power, but firms with higher social impacts from their emissions have less bargaining power with local authorities.

Even in developing countries where government monitoring and enforcement is weak, the literature finds that some firms comply with environmental regulations for extra-legal reasons, such as subsidized environmental management training (Dasgupta et al. 2000) and community pressures (Wang 2000). Note that with some market-based approaches to water pollution control, the necessity for sufficient capacity in the public sector may extend to tax collection, as well as environmental regulation (Besley and Persson 2013).

3.3. Rent-Seeking and Environmental Regulation

Rent-seeking in regulatory systems can drive a wedge between what the designers of environmental policy intend, and what is actually achieved. For example, in the case of corruption, if a bureaucrat's own utility maximization affects the processes of standard-setting, monitoring, or enforcement, the level of water pollution abatement may fall below what is socially desirable (Wilson and Damania 2002, Greenstone and Jack 2015). In theory, corruption can also cause more polluting firms to enter a market than would be the case if it were not possible to bribe regulatory officials (Biswas and Thum 2017). More generally, relationships between citizens or firms and policymakers may distort policy outcomes relative to what society, in general, may want to achieve (Wilson and Damania 2002). For example, firms may simply lobby for less stringent environmental regulation, or regulators may be "captured" by the industries they monitor.[4] A comprehensive review of the literature on rent-seeking and regulation is beyond the scope of this paper, and is available in Dal Bó (2006). Below, we discuss the empirical evidence regarding the impacts of these phenomena on regulation in industrialized and developing countries.

3.3.1. Industrialized country empirical applications

While the empirical literature demonstrating regulatory capture in industrialized countries is fairly thin, there are a few examples. We know of no examples specific to water pollution and its regulation.

Cicala (2015) examines U.S. power plants' procurement behavior before vs. after the end of cost-of-service regulation of electric utilities, and he shows that a significant share of the decrease in coal expenditures by the coal-fired plants post-deregulation can be attributed to pre-deregulation regulatory capture. There is also some evidence that influence groups cause distortions in environmental policy outcomes, specifically, in the United States and other industrialized countries, though the evidence does not extend to rent-seeking, *per se*.[5] For example, Cropper et al. (1992) show that the U.S. EPA's decisions regarding pesticide registration cancellation or extension in the 1970s and 1980s were influenced by public comments from both industry and environmental groups. In contrast, Muehlenbachs et al. (2011) look for evidence that regulated firms influence the timing of U.S. EPA press releases that could be potentially damaging to firms' reputations, and find no evidence of such influence.

3.3.2. Developing country empirical applications

To our knowledge, only one study in the literature directly tests for the effects of corruption on water pollution outcomes in a developing country. In India, aligning environmental auditors' incentives with those of regulators significantly reduces under-reporting of water pollution emissions from audited plants, though the effects for water pollution are somewhat smaller than those for air pollution (Duflo et al. 2013). Interestingly, the firms in this experiment actually reduced water pollution emissions in response to better auditing, though air emissions showed no statistically significant impact (Duflo et al. 2013).

Reductions in environmental quality due to rent-seeking in developing country regulatory processes have also been demonstrated for air quality in Mexico (Oliva 2015) and deforestation in Indonesia (Burgess et al. 2012). Oliva (2015) uses data on vehicle emission testing in Mexico City and finds that 9.6% of car owners in Mexico City paid about $20 bribes to circumvent testing. Her simulation results show that eliminating cheating and increasing retest costs would reduce vehicle emissions in Mexico City. Burgess et al. (2012) use satellite data of land cover change to investigate how local officials' incentives affect deforestation in Indonesia. Their results suggest that corruption increases deforestation in Indonesia and officials' rent-seeking behaviors vary with their degree of market power.

Much of the empirical evidence regarding regulatory capture comes from analyses of regulated utilities in developing countries (Dal Bó 2006). For example, Dal Bo and Rossi (2007) attribute large efficiency losses to corruption of public officials in regulated utilities in Latin American countries, estimated in terms of increases in the number of employees in the sector, as well as increased operation and maintenance costs.[6] Regulatory capture in a non-environmental setting can have effects that spill over to environmental quality. For example, groundwater depletion is closely tied to electricity prices, given that the cost of pumping water to the surface is an important factor in the rate of depletion. Sekhri and

Nagavarapu (2013) show that the imposition of electricity price regulation in India reduced groundwater extraction more in jurisdictions where politicians had smaller incentives to capture the electricity tariff-setting process than in jurisdictions with higher such incentives.

It is not clear whether market-based approaches to water pollution control are more or less subject to distortions from rent-seeking than prescriptive regulations. The empirical examples discussed above are examples of rent-seeking in the context of command-and-control policies, and we know of no examples in the context of market-based policies in developing countries. However, taxation provides an opportunity for rent seeking in developing countries (Olken and Pande 2012), so this may be an additional concern with respect to environmental taxes or auctioned tradable permit systems.

3.4. Decentralized Regulation and Multi-Jurisdictional Spillovers

Decentralization can be an efficient way to provide public goods such as water pollution control, as decentralized regulation allows local jurisdictions to set standards in line with local preferences for environmental quality, and individuals sort across jurisdictions on this basis, among others (Tiebout 1956, Oates 1972, Oates and Schwab 1988). On the other hand, river pollution across political and geographic boundaries poses classic negative externalities and free-riding problems. Regulating water pollution then is difficult due to such displaced environmental damages, where downstream residents are impacted by pollution from upstream industries. Since polluting entities emitting to shared water resources cannot internalize the full benefits of pollution reductions, they may have fewer incentives to constrain emissions. A related explanation of the spillover effect is that local regulators make policies to maximize political support (Magat et al. 2013). When the benefits of an environmental policy are mostly external, regulators have less incentive to act. Ultimately, inter-jurisdictional spillovers can contribute to a “race to the bottom” in which jurisdictions compete for industries by promulgating less stringent regulations (Kunce and Shogren 2005). Polluting facilities may also move in response to policy changes within a jurisdiction, counterbalancing any water quality improvement within the more stringent jurisdiction with increases in pollution elsewhere, known as the “pollution havens hypothesis.” The evidence for this behavior globally suggests that while it does occur, effects are generally small and focused in industries that are particularly dirty, have high regulatory compliance costs, and can be easily moved (Decheleprêtre and Sato 2017); one recent paper provides evidence of this kind of unintended impact of regulatory asymmetry on water pollution in China (Chen et al. 2018). Centralized regulation, while less responsive to local preferences for environmental quality, can internalize spillovers. Which approach is most efficient depends on whether the efficiency loss from centralized standard-setting that ignores local conditions exceeds the efficiency gain from eliminating spillovers (Banzhaf and Chupp 2012).[7]

3.4.1. Industrialized country empirical applications

The literature provides some empirical evidence of inter-jurisdictional water pollution spillovers both within and across country borders. Using TRI data, Helland and Whitford (2003) find that facilities near US-Canada/US-Mexico borders systematically emit more air and water pollution than other facilities; these systematic differences do not exist for off-site in-state transfers. Sigman (2002) uses biochemical oxygen demand (BOD) data to look at the extent of free-riding across international borders and finds that international spillovers significantly impair water quality in rivers. When excluding European Union (EU) upstream stations, stations upstream of international borders have about 40 percent higher BOD levels than other stations. Stations upstream within the EU have less free-riding problems than stations upstream of non-EU borders; the EU seems to offer an institutional setting that constrains free-riding.

Spillovers are relevant across jurisdictions within a country, as well. Environmental policy in the US is a hybrid of federal standard setting and state implementation and enforcement. Sigman (2005) investigates free riding by U.S. states authorized to issue and enforce pollution permits under the Clean Water Act, in comparison to states for which the U.S. federal government is in charge of CWA monitoring and enforcement. She finds that stations downstream from authorized states have water quality index (WQI) values about 4 percent lower than other stations, and estimates significant welfare costs from this free-riding problem. Gray and Shadbegian (2004) use air and water pollution data from US pulp and paper mills and find that plants near other states emit more pollution. These transboundary spillover effects are reduced if the affected states have more pro-environment Congressional representatives.

3.4.2. Developing country empirical applications

Empirical studies suggest that transboundary spillover and free-riding problems also exist in developing countries. In 2001, the Chinese government included improving environmental quality in its Five-Year Plan for the first time and mandated pollution reduction targets for all provinces. However, the central government did not describe how the local Bureau of Environmental Protection (BEP) should coordinate to achieve these targets. Because local BEPs are controlled by local governments, they were incentivized to strategically place polluting industries in border counties. Cai et al. (2016) assess the impacts of this approach using data on firm production and entries. They find that water-polluting production and new entry into water-polluting industries are significantly higher downstream of county borders. With pollution levy fees collected by the provincial government as an indicator of enforcement, provincial governments appear to have allocated the most lenient enforcement to downstream border counties in reaction to pressures from the 2001 policy reform (Cai et al. 2016). Chiang (2016) examines the impacts of central government-provided incentives for local Chinese government officials to expand the fraction of households within their jurisdictions who are covered by clean piped water and sanitation systems. She finds that these incentives do more to expand access to piped water than to expand access to sanitation, because sanitation expansion creates spillover benefits to downstream jurisdictions that are not captured locally.

Lipscomb and Mobarak (2017) take advantage of the exogenous redrawing of county borders in Brazil to identify the potential water pollution spillovers in rivers as they approach borders. They find that pollution increases as rivers travel towards downstream borders, and that it increases at an increasing rate as rivers approach a border.

Abundant evidence demonstrates the potential importance of multi-jurisdiction spillovers in designing water pollution policy, but solutions to the problem are still not clear. The Coase theorem suggests private negotiation among actors can, under certain restrictive circumstances, provide efficient solutions to such negative externalities (Coase 1960). There are many examples, in practice, of Coasian solutions to pollution problems (Anderson and Libecap 2014). However, private solutions are unlikely in an international context where no binding legal framework exists to facilitate negotiation and enforce standards (Sigman 2002; Wolf 2007). Even within individual countries, transaction costs and poorly-defined property rights are important barriers to private resolution of spillovers.

Another possible solution is to provide local regulators stronger incentives to improve water quality, where regulation is decentralized. The idea is to motivate local officials to take action to reduce pollution (by internalizing some of the benefits from water quality improvements) and to discourage free-riding by upstream jurisdictions. This approach might also help drive local regulators of shared water resources to the bargaining table, if downstream regulators can reduce the costs of achieving a water quality goal by inducing upstream regulators to reduce the burden of pollution entering their jurisdictions.

Several studies have examined the impacts of such approaches in China. In 2005, China's central government began to evaluate government officials for promotion based on certain water quality measures. Kahn et al. (2015) find that while chemical oxygen demand (COD)[8] decreased significantly at boundaries after the regime shift, more harmful pollution measures such as petroleum and mercury were not affected (Kahn et al. 2015). In a follow-up study, Chen et al. (2018) show that the policy targeting at reducing water pollution has had unintended perverse consequences. After the policy change in 2005, water quality in the Yangtze River deteriorated in 2011 despite all provinces having achieved their COD reduction goals. Because upstream provinces of the Yangtze River are less economically developed and have relatively better water quality, they also have less stringent COD reduction targets. As local officials' regulatory intensity has varied over space and time, water-polluting industries have shifted to less-regulated areas upstream (Chen et al. 2018). Because water pollution is displaced to upstream areas, a larger proportion of the river and a larger population are exposed to pollution.

Zhang et al. (2018) assess a more recent Chinese effort to better link the incentives of decentralized environmental managers with national objectives — the National Specially Monitored Firms (NSMF) pilot program, established in 2007. This program identified specific firms that were major emitters of air and water pollution (as well as hazardous solid waste) and provided centralized oversight of the local monitoring of these firms; all NSMFs are required to install automatic monitoring systems and transmit emissions information in real time to the central government, which verifies the accuracy of the data through monthly inspections. Local governments receive the information from the central government, and regulated firms' pollution levies are determined jointly by the central and local authorities. Using a regression discontinuity design, Zhang et al. (2018) find that the additional central supervision of local authorities in the NSMF program reduces industrial COD emissions by 26.8 percent in the first year, with continuing reductions in later years.

Notes

1. There are many examples of these approaches in the literature looking at environmental questions other than water pollution; for example, Ebenstein et al. (2017) focus on air pollution and health in China, and Davis et al. (2014), examine appliance subsidies and energy efficiency in Mexico.

2. Researchers may want to give some thought to creative experimental approaches in this regard. For example, if a large-scale sanitation intervention is conducted in a location near a waterway, in addition to collecting information on health outcomes among local households,

researchers could collect water quality samples downstream, and examining whether the sanitation intervention creates positive externalities in terms of reduced ambient concentrations (and subsequent human health or other impacts) downstream.

3. See http://www.cnemc.cn for access to the China Environmental Monitoring Center, and http://123.127.175.45:8082/ for the link to the system.

4. The literature also identifies incentives for firms to lobby for *more* stringent (rather than less stringent) regulatory standards so as to increase rivals' costs (Puller 2006; Grey 2018).

5. Evidence of regulatory capture in the United States has also been demonstrated in sectors such as professional sports (DeAngelo et al. 2018), and commercial casinos (Walker and Calcagno 2013).

6. Corruption in other sectors in developing countries has important social costs, as well. Distortion of loan allocations to politically-connected firms by public banks in Pakistan generates social costs on the order of 0.3 to 1.9 percent of Pakistan's gross domestic product (Kwhaja and Mian 2005).

7. Several papers in the literature address this topic from the perspective of the theory of common pool resources, using game theory and drawing upon specific case studies of shared rivers, though the focus is primarily on water quantity, rather than water quality (Rogers 1969; Frisvold and Caswell 2000). In theory, the incentive to free-ride in international surface water allocation can sometimes be overcome. Becker and Easter (1999) consider the U.S. states and Canadian provinces sharing the Great Lakes and show that a relatively small coalition can provide a stable cooperative outcome, given the distribution of gains and losses in the region from cooperating over water diversions.

8. This is one of two environment indicators the central government in China uses to evaluate local officials' promotion opportunity. The other indicator is sulfur dioxide.

4. Identifying the Evidence Gaps

Tables 1 and 2 summarize the literature's coverage of water pollution control policies, focusing only on empirical studies of policy impacts. Table 1 contains studies of policies in industrialized countries, and Table 2 focuses on developing country studies. In both tables, studies are color-coded, with those containing plausibly causal estimates of policy impacts in black, descriptive or non-causal empirical studies in blue, review articles in orange, and qualitative studies in green. The tables do not mention every study in the literature, though we have tried to highlight those that are most informative regarding the issues covered in this paper. Together, Tables 1 and 2 provide a map of the remaining gaps in empirical evidence of water pollution policy impacts, which are quite substantial.

Notice, for example, that in industrialized countries, which have created and piloted market-based approaches and promoted them internationally as effective and cost-effective solutions to pollution problems, there are no causal estimates (to our knowledge) of the impacts of taxes and tradable permits on water quality. This is mostly due to the fact that industrialized countries have seldom implemented taxes and tradable permits as water quality regulations (most such policy instruments have targeted air pollution in industrialized countries). And there are now high-quality estimates of the impacts of prescriptive water pollution control policies in these countries. However, the evidence gap in market-based approaches to water pollution control is striking, given the general availability of water quality data in these settings. For example, European countries have used water pollution taxes for many decades, and the U.S. CRP (a PES program with water quality goals) was established in 1985, yet we have no plausibly causal estimates of these policies' impacts on water quality or (secondarily) their cost-effectiveness relative to alternative approaches. In contrast, several studies have examined the impacts of voluntary pollution control policies, perhaps due to an inherent skepticism of these programs' potential.

The historical impacts of major water pollution control investments in the United States and other industrialized countries are well-studied. Not surprisingly, we identified no studies that empirically estimate health impacts from ambient water pollution control policies more recently in these settings. Significant health impacts, where raw water quality is already fairly high, are unlikely and may not be an important area for further research, though recent work suggests that nutrient runoff from agricultural sources – a significant ambient water quality problem not formally regulated in many industrialized countries – may be a greater health concern than previously known (Ward et al. 2018).

While the analysis of water pollution policy impacts in developing countries started somewhat later than in industrialized countries, our review of the literature identified more plausibly-causal studies of water quality policy impacts in developing countries than in industrialized countries (Table 2). In fact, the convincing studies of the beneficial impacts of water pollution taxes in developing countries, especially in China (both the general pollution levy system and the "Pay for Permit" system in the Lake Tai Basin), are really the only such estimates in the literature. In contrast to industrialized countries, economists have recently demonstrated very significant gains in human health (especially reductions in premature mortality) from ambient water quality policies in China, India and Bangladesh (focusing on groundwater in this latter case). Note that while the empirical evidence regarding the effectiveness of market-based approaches to water pollution control is thicker in developing countries than in industrialized countries, there are still a very small number of causal estimates available.

While the PWS evaluation literature is also still thin, here again, the best empirical evidence for the impacts of these programs are in developing countries, not in industrialized countries. Note that in most cases, even where water quality is an important goal of the program, many of the best assessments of PWS programs do not actually evaluate impacts on emissions or ambient water quality; Zheng et al. (2013) is the exception and provides convincing evidence of beneficial and cost-effective impacts in China.

Unlike in the industrialized-country setting, there is a pressing need for more research on the health impacts of ambient water pollution control in developing countries, given the demonstrated health damages from exposure to the high levels of surface-water pollution prevalent in many countries (Do et al. 2018, Ebenstein 2012). More research in this area is essential to monetizing the benefits of additional pollution control, which would be a key input to further regulation and enforcement of existing regulations.

Finally, as Table 2 indicates, the water pollution policy approach with the strongest plausibly-causal evidence regarding impacts on health and other aspects of human capital is infrastructure investment (primarily investment in large-scale sewage collection and treatment). Notably, none of the papers we identified in this part of the literature measure impacts on either water pollution emissions or ambient water quality; they jump to estimating impacts on health outcomes directly.

5. Conclusions

The theory of market-based approaches to environmental policy is well-established, and many high-profile experiments with taxes, tradable permits, and information disclosure policies in industrialized and developing countries have demonstrated that market-based approaches can achieve pollution reduction goals cost-effectively. In addition, theory and empirical evidence suggest that taxes and tradable permits provide a stronger incentive for long-run, abatement-cost-reducing technological change. Given the scale of water pollution problems in developing country rivers, lakes, streams and coastal waters, market-based pollution control policies hold significant promise within the menu of solutions countries may consider.

Unfortunately, developing countries have few concrete examples on which to draw when designing new market-based water pollution control policies. China's pollution levy system is a good example, as most analyses suggest that it has reduced water pollution emissions. However, China's levies are assessed only on emissions above regulatory standards, not on all units of emissions (and levels are likely well below marginal damages), making this different from a true Pigouvian tax. The newer pilot tax policy in China's Lake Tai Basin may be a better example, given that the tax is assessed on all units of pollution, and it appears to have dramatically reduced water pollution.

Perhaps it should not be surprising that we have uncovered so few clear examples of successful market-based policies for water pollution control in developing countries; the record in industrialized countries is also thin. In fact, the only plausibly-causal estimates of the impacts of water pollution taxes on water quality focus on developing countries. While there are a number of successful tradable permit programs for water pollution control in the United States, Australia and Canada, all of these operate on small scales compared with the most well-known examples for air pollution control. The design of these programs can serve as a model for future such systems in developing countries – particularly the careful development of trading ratios to account for spatially heterogeneous marginal damages. However, water quality trading programs would have to operate at a much larger level to make progress on the severe water pollution problems in major developing country rivers; countries like China and India would truly be breaking new ground were they to design significant water pollution cap-and-trade policies at scale.

Information disclosure programs appear to be more common than either taxes or tradable permit programs for water pollution control, in both industrialized and developing countries. At least two information disclosure programs in developing countries (Indonesia's PROPER and India's GRP) may have resulted in significant water pollution emissions decreases. In each of these cases, the best available studies establish causal links between emissions reductions and disclosure, though little is known about *ambient* water quality improvements these policies may have achieved. Voluntary agreements are difficult to assess, given the problem of selection. Though many VAs have been implemented in developing countries, there is little, if any evidence that they have actually reduced pollution.

Major public infrastructure investments, particularly in municipal wastewater treatment, clearly paid off for industrialized countries during earlier periods of their development. The estimated benefit-cost ratios and estimated cost-per-life-saved of these investments are truly impressive in comparison to current environmental regulations, on the margin. If one pairs these historical analyses for the United

States and Europe, with current estimates of the human health and human capital impacts of investments in drinking water, sanitation, and (in a small number of cases) ambient water pollution control in developing countries, it seems likely that many major urban wastewater treatment investments would be net beneficial. However, projects and investments would need to be evaluated individually. In addition, given that the literature suggests that such investments in the United States may at some point have crossed the threshold to net costs, it would be useful to examine how much treatment can be achieved before reaching that point.

The empirical literature also finds a good deal of support for the hypothesis that where regulatory institutions are weak, water pollution control policies are unlikely to have much effect. The literature on monitoring and enforcement, in particular, shows that this is as true in industrialized settings as it is in developing countries. However, the average strength of regulatory institutions in many developing countries is simply much lower, and corruption and rent-seeking are more pervasive. The challenges of decentralization and water pollution spillovers also must be considered in developing country settings. On a positive note, recent experiments with industrial pollution auditors in India, and econometric analysis of policies that increase central oversight of local pollution monitors in China, provide evidence that fairly simple, low-cost interventions may be able to overcome the some of these particular challenges for water pollution policy in developing countries.

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