POLICY INSTRUMENTS FOR WATER POLLUTION CONTROL IN DEVELOPING COUNTRIES

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Abstract

Severe ambient water pollution is common in many developing countries. A broad array of regulatory and other policy approaches can improve water quality, but some approaches are more well-studied than others, and there are many additional challenges specific to the developing country setting. This paper describes a set of prescriptive, market-based, voluntary and other policy instruments to control water plllution and reviews empirical assessments of these approaches in practice, focusing primarily on developing countries. We also examine additional challenges such as data availability, monitoring and enforcement, rent-seeking, and the issue of decentralization. The paper highlights important gaps in published empirical research on these issues, which increase in importance as the evidence accumulates regarding water pollution's impacts on outcomes such as health, human capital, and productivity.

Keywords: water pollution, market-based, prescriptive, information disclosure, payments for watershed services, infrastructure

JEL Codes: Q53, Q58

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1. INTRODUCTION

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). This paper examines a range of 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.

Water pollution control in developing country settings may generate substantial benefits. Ambient water pollution has human health impacts in these settings (Ebenstein 2012, Do et al. 2018). Poor water quality may also reduce agricultural output (Hagerty 2019), educational outcomes (Zhang and Xu 2016), and labor productivity (Meeks 2017). Damages 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). If they are to be net beneficial, how should reductions in municipal, agricultural and industrial water pollution be achieved in developing countries? The standard economic prescription would be to use market-based policy instruments to reduce pollution cost-effectively. 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.

This paper proceeds as follows. In Section 2, we describe the menu of policy instruments used to control ambient water pollution: prescriptive approaches, market-based approaches, information disclosure, voluntary approaches, and infrastructure investments. We focus primarily on developing countries, mentioning research from industrialized countries where appropriate. Section 3 considers challenges for water pollution control policies in 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. Section 4 summarizes the gaps in the empirical literature on water pollution policy impacts, and Section 5 concludes.

2. WATER POLLUTION POLICIES AND EMPIRICAL EVIDENCE OF THEIR EFFECTIVENESS

2.1 Prescriptive policies and their application

The standard approach to environmental regulation tends toward an array of policy instruments known 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. For example, a technology standard requires firms to use a particular pollution abatement technology. A performance standard allows polluters more leeway in the choice of control technology, imposing a ceiling on total emissions in a period, or a maximum allowable emissions rate. Most existing water quality regulations in industrialized countries use these approaches. For example, the Clean Water Act (CWA) -- the central set of water pollution regulations in the United States -- relies primarily on a set of effluent standards, implemented through point-source permitting, in addition to substantial federal funding for the expansion and improvement of municipal wastewater treatment.

Developing countries, likewise, regulate water pollution primarily through the use of standards. For example, India's National River Conservation Plan (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.

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.

In China, the current primary policy on water pollution reduction is the Water Pollution Prevention and Action Plan, or 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 those that fail to meet targets. The Water Ten Plan also requires plants in ten 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. Because the Water Ten Plan was passed recently, it may be possible to estimate the impacts, benefits and costs of it in the future.

2.2 Market-based policies

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. One such approach is to tax negative externalities, and to subsidize positive externalities.[3] Under a uniform 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, so that an efficient tax is differentiated by source, or using tax "zones" recognizing heterogeneity in damages (Boyd 2003). The standard Pigouvian results in economic theory address pollution from point sources; taxing nonpoint source (NPS) pollution – an important share of total pollution in the water context – is considerably more complicated, and has prompted much discussion in the literature (Shortle and Horan 2001).

Tradable pollution permits, or "cap-and-trade" systems, are another market-based approach. 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.

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[3] 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 manage externalities efficiently (Baumol and Oates 1988).

The permits are transferable, and 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. When the marginal damages from water pollution vary with the location of discharge, depending on the characteristics of receiving waters and other factors, establishing location-based trading ratios for each pair of polluters is an efficient approach (Konishi et al. 2015).

Payments for ecosystem services (PES) are another class of market-based approach. Recent work counts 550 active PES programs around the world (Salzman et al. 2018). 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. When applied to water pollution control, PES approaches are known as payments for watershed services (PWS). As with taxes and tradable permits, efficient PWS systems must account for spatial and intertemporal heterogeneity (Jack et al. 2008). 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, for example by using auctions (Ferraro 2008).

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 (Tietenberg 1998, Powers et al. 2011). 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.

2.3 Applications of market-based water pollution policies

2.3.1 Pollution taxes. Water pollution taxes are fairly uncommon in industrialized countries. France, Germany and the Netherlands all have systems of water pollution taxes – some dating to the 1970s (Boyd 2003). Of these, only the Dutch water pollution fee system has been statistically associated with a decline in pollution (Bressers 1988).

In contrast, developing countries have experimented with more robust water pollution taxation, and some of these policies have been evaluated econometrically. China's pollution levy system, in place since the early 1980s, 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, plants must pay levies on the three pollutants with the largest exceedance, and levy rates have increased dramatically. Even before the reform, general equilibrium analyses showed that the pollution levy system reduced emissions (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 not to CAC requirements. Recent analysis 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).

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. This pilot program charges firms for every unit of pollution, a classic Pigouvian tax. 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 demonstrates that treated plants reduced emissions by about 40 percent in the first two years post-policy (He and Zhang 2018).

In 1993, Colombia's Law 99 implemented a national discharge fee system. The law mandated that CARs, a set of government-determined regional environmental regulatory authorities charge all polluters a fee per unit of biological oxygen demand (BOD) and total suspended solids (TSS). 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, these effects may not be causal. 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).

In the 1980s, Malaysia began charging a fee on BOD emissions from the palm oil industry. Unpublished studies suggest that the implementation of these fees reduced the BOD load very dramatically, even while palm oil production increased (Vincent et al. 1997).

2.3.2 Tradable permits. The experience with water quality trading in 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. 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). To our knowledge, none have been rigorously evaluated for either effectiveness or cost-effectiveness.

We can identify no examples of an active water quality trading program in a developing country. The lack of water quality trading programs in these settings is also consistent with the small number of tradable permit policies for air pollution 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 reduced pollution (Coria and Sterner 2010).

2.3.3. Payments for watershed services. PWS approaches have been implemented for water pollution control in many industrialized country contexts. For example, 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). The CRP's long-term contracts do result in lasting agricultural land retirement (Roberts and Lubowski 2007). 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, though empirical work by the implementing agency suggests that the CRP generates tens of millions of dollars in recreational water quality benefits each year (Feather et al. 1999).

The literature contains many summaries of applications of PES in developing countries, which include some robust PWS programs (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). For example, in 2006, the government of Beijing, China began paying farmers upstream of Miyun 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. A recent assessment of this PWS program, known as Paddy Land-to-Dryland, 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). By 2010, all rice fields upstream had converted to dryland crops (mostly corn). Total nitrogen and total phosphorus concentrations have been reduced significantly (Zheng et al. 2013).[4]

Watershed protection and aquifer recharge are among the many goals of Mexico's federal conservation payments program. Protecting forests to maintain their "hydrological services" is the program's main stated goal, and the program is financed by water user fees. 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 (Alix-Garcia et al. 2018). While these impacts may have improved water quality, to our knowledge, there is no rigorous assessment of this question.

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 program's impacts are available, one study suggests that it has both reduced deforestation and regenerated riparian vegetation, which could 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. The water quality impacts of these programs have not been rigorously evaluated.

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[4] 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.

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. 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).[5] Water quality monitoring is taking place for this program, suggesting that its impacts are at least potentially measurable, and anecdotal reports suggest that water quality has improved (Branca et al. 2011).

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, which requires manufacturing firms to report annual chemical releases to air, water, and land to the U.S. EPA, which then publicly releases the information. From the TRI's inception in 1986 to the mid-2000s, total annual releases of reportable chemicals fell by nearly 50 percent, but the observed decrease has not been causally attributed to the TRI (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, Khanna et al. 1998). Similarly, the disclosure of environmental incidents and violations appears to have strong negative effects on the market value of firms in Canada (Laplante and Lanoie 1994) and Korea (Dasgupta et al. 2006).

Information disclosure programs have been used often in developing countries as environmental policy instruments, though only a few have been rigorously evaluated. 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). Early studies found 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). More recently, Garcia et al. (2007, 2009) use panel data on BOD and COD emissions from about 140 firms, estimating firm fixed-effects models to

[5] 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).

evaluate PROPER's effectiveness. They find that PROPER does reduce pollution emissions, especially for low-compliance firms.

India's Green Rating Project (GRP) began in 1997 and is run by an Indian environmental NGO, 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 ratings 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, and they find that it significantly reduced pollution from dirty plants.

Overall, the information disclosure policies that have been evaluated in developing countries seem to reduce firms' water pollution discharges. These results should be interpreted with caution, however, since empirical studies are mostly performed on firm-reported discharge data (García et al. 2007). 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 among consumers who might react to disclosure may be relatively low (García et al. 2009). Blackman et al. (2004) conducted a plant-level survey of Indonesian firms covered by PROPER. Their survey suggests that an important channel through which PROPER improves firms' performance is by improving managers' understanding of emissions and abatement opportunities. Garcia et al. (2007, 2009) and Powers et al. (2011) all suggest that plants in wealthier communities are more responsive to information disclosure.

2.4 Voluntary approaches

Voluntary approaches (VAs) 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 and Coglianese 2009). Advantages of VAs 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 (2) increased cooperation and communication between polluters and regulators (Alberini and Segerson 2002). However, an obvious potential downside is that firms may not engage in costly pollution reduction without requirements, monitoring and enforcement. Empirical analyses of such programs must worry about selection, since the firms for whom it is least costly are most likely to join.

VAs have been applied in several cases in the United States, Europe, and Japan at both the federal and state level. In the United States, one well-studied VA is the 33/50 program. Evidence of 33/50's effectiveness is mixed. Vidovic and Khanna (2012) 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. Zhou et al. (2019) show that even firms that do not participate may reduce emissions, due to spillovers from participating firms. Overall, it is not clear if VAs make positive contributions to water quality improvements in industrialized settings, or how they compare with mandatory regulations (Borck and Coglianese 2009).

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, 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).

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 firms graduate from the program, participants do not have substantially less pollution than matched nonparticipants. Blackman et al. (2013) find 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, VAs may facilitate capacity-building in both government institutions and the private sector (Blackman et al. 2013).

2.5 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 and improves welfare in other ways (Devoto et al. 2012).

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 provides public goods. 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 strong economic justification for government provision, and large-scale investments in sanitation infrastructure are common.

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, which generated very substantial net economic benefits (Alsan and Goldin 2018, Cutler and Miller 2005, Delaney et al. 2011). For developing countries, the literature also shows positive impacts of toilets, latrines, and safe drinking water on health and educational outcomes, with attendant net benefits (Soares 2007, Duflo et al. 2015, Zhang 2012, Zhang and Xu 2016). 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. Given the prevalence of this approach, evaluating the impacts of large-scale infrastructure investments to reduce ambient water pollution is an important area for futher research.

3. CHALLENGES OF WATER POLLUTION CONTROL IN DEVELOPING COUNTRIES

3.1 Data availability

One primary challenge to implementing and assessing the effectiveness of water pollution control policies in developing countries is data availability. This challenge may increasingly be overcome with satellite data measuring pollution (Schaeffer et al. 2012). Researchers can also exploit 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. For example, Do et al. (2018) 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.

Field experiments are another tool for overcoming the problem of inadequate observational data to examine the impacts of drinking water and sanitation interventions (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 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.

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. Data quality is a significant concern for studies of pollution China (Ghanem and Zhang 2014). Since 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.[6] Both central and local governments have limited capability to manipulate these data, compared to prior monitoring regimes. The automatic monitoring system also increases the frequency of water quality monitoring, from once a month to as often as every four hours. China's central government recently expanded the automatic water quality monitoring system from 100 to 2,050 stations across the country (Xinhua Net 2018).

[6] 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.

Third-party monitoring can also improve water quality data in developing countries. Duflo et al. (2013) show 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 function as one form of third-party monitoring. The literature provides theoretical background on crowdsourcing water quality data (Borden et al. 2016), however. 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

A rich literature in environmental economics examines the role of monitoring and enforcement of pollution control regulations in affecting outcomes (Shimshack 2014). Dozens of studies establish that monitoring and enforcement of environmental regulations reduce pollution, deter future violations, and encourage over-compliance by regulated entities (Shimshack 2014). Most 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. Studies on U.S. water pollution include Shimshack and Ward (2005, 2008), and Earnhart (2004a, 2004b). 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 (Greenstone and Jack 2015). 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).

A few empirical studies demonstrate the importance of monitoring and enforcement of water pollution policies in developing countries. In China, inspections may play a more significant role than pollution levies, themselves, in reducing emissions (Dasgupta et al. 2001). 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. Wang and Wheeler (2003) demonstrate that the impacts of China's pollution levies are higher in areas of the country where regulatory institutions are stronger. Lin (2013) uses data on plant-level pollution to show that inspections increase plants' self-reported pollution by more than 3 percent but may not be effective in reducing pollution.

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 provides centralized oversight of the local monitoring of firms that are major emitters of air and water pollution and hazardous waste. 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.

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 (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. A comprehensive review of the literature on rent-seeking and regulation is available in Dal Bó (2006).

A small number of studies directly test for the effects of corruption and related behavior on water pollution outcomes in developing countries. 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). 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).[7] 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

[7] 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).

bargaining power, but firms with higher social impacts from their emissions have less bargaining power with local authorities.

3.4. Decentralized regulation and multi-jurisdictional spillovers

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 and Schwab 1988). On the other hand, water pollution across political and geographic boundaries creates negative externalities and free-riding problems. Ultimately, inter-jurisdictional spillovers can contribute to a "race to the bottom" in which jurisdictions compete for industries through 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 (Decheleprêtre and Sato 2017). 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).

The literature provides some empirical evidence of inter-jurisdictional water pollution spillovers both within and across country borders in industrialized settings (e.g., Sigman 2005). 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 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. They find that water-polluting production and new entry into water-polluting industries are significantly higher downstream of county borders. 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 positive spillovers to downstream jurisdictions (via reduced ambient water pollution) that are not captured locally.

Lipscomb and Mobarak (2017) exploit 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. Theory suggests private negotiation among actors can, under certain restrictive circumstances, provide efficient solutions to such negative externalities (Coase 1960). There are 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 contracts. 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. Two recent 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 water quality. Kahn et al. (2015) find that while chemical oxygen demand (COD) 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 has had unintended consequences. After the policy change, water quality in the Yangtze River deteriorated 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, so 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.

4. IDENTIFYING THE EVIDENCE GAPS

Table 1 summarizes the literature's coverage of water pollution control policies in developing countries, focusing only on empirical studies of policy impacts and providing a map of the remaining gaps in published knowledge.[8] It is striking that the studies of the beneficial impacts of water pollution taxes in developing countries, especially in China, are really the only such estimates in the literature. Economists have also 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 comes from developing 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.

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. Finally, as Table 1 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, and the mechanisms for those impacts are not always clear.

5. CONCLUSIONS

The theory of market-based approaches to environmental policy is well-established, and many experiments with taxes, tradable permits, and information disclosure policies in industrialized and developing countries have demonstrated that market-based approaches can

[8] Table A in the online Appendix focuses on industrialized country studies.

achieve pollution reduction goals cost-effectively. 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. Recent benefit-cost analyses of primarily CAC approaches in the U.S. suggest that costs may exceed benefits (Keiser and Shapiro 2019), providing greater impetus to focus on cost-effective approaches, especially as regulatory approaches are still evolving in developing countries, providing some flexibility to use empirical evidence to design their systems.

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 multiple analyses suggest that it has reduced water pollution emissions. The newer pilot tax policy in China's Lake Tai Basin is an even better example, given that the tax is assessed on all units of pollution, and it appears to have dramatically reduced water pollution.

Perhaps it is not 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. Successful tradable permit programs for water pollution control exist in the United States, Australia and Canada, but all of them 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, but water quality trading programs may need to operate at a much larger scale to reduce the severe water pollution problems in major developing country rivers.

Information disclosure programs are 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 decreased water pollution emissions. 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.

Major public infrastructure investments, particularly in municipal wastewater treatment, clearly paid off for industrialized countries during earlier periods of their development. Given the evidence, it seems likely that major urban wastewater treatment investments would be net beneficial, but 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, monitoring and enforcement are weak, water pollution control policies likely have weaker effects. 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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Table 1. Empirical evidence on the impacts of water pollution control policies in developing countries

		Environmental impacts		Health impacts		Socioeconomic impacts					Cost-effectiveness	Unintended consequences	
Water pollution control policies		***Reduction in emissions/effluent***	***Ambient water quality improvement***	***Reduced morbidity***	***Reduced premature mortality***	***Water supply behavior***	***Capacity building***	***Capital market***	***Labor market and poverty alleviation***	***Education and cognitive development***	***Cost-effectiveness***	***Strategic regulatory avoidance***	***Rent-seeking***
Prescriptive policies	***Technology standards***	Greenstone and Hanna 2014	Greenstone and Hanna 2014										
	Performance standards	Afsah and Makarim 1999; Dasgupta et al. 2000; Duflo et al 2013; Zhang et al. 2018			Do et al. 2018							Kahn et al. 2015; Chen et al. 2018	Duflo et al. 2013
Market-based policies	***Pollution taxes***	Jiang and McKibbin 2002; Wang and Wheeler 2003; Wang and Wheeler 1999; Wang 2000; He and Zhang 2018; Ebenstein 2012; Blackman 2006 ; Vincent 1993; Vincent et al. 1997; Bluffstone 2003	Ebenstein 2012; Bluffstone 2003		Ebenstein 2012		Besley and Persson 2013				Ebenstein 2012	Cai et al. 2016, Lin 2013	
	Tradable permits												
	Information disclosure	Tietenberg 1998; Blackman et al. 2004; Garcia et al. 2007, 2009; Powers et al. 2011				Madajewicz et al. 2007; Barnwal et al. 2017; Bennear et al. 2013		Dasgupta et al. 2001a					
	Payments for ecosystem services	Zheng et al. 2013	Moreno-Sanchez et al. 2012, Branca et al. 2011						Pagiola et al. 2005; Bulte et al. 2008; Zheng et al. 2013; Alix-Garcia et al. 2015, 2018; Branca et al. 2011; Pattanayak et al. 2010, Mussa & Mwakaje 2013; Kosoy et al. 2007		Zheng et al. 2013	Pattanayak et al. 2010	
Voluntary approaches		Pargal and Wheeler 1996; Blackman et al. 2006; Blackman and Sisto 2006; Blackman et al. 2010; Jiménez 2007; Hu 2007; Blackman et al. 2013	Blackman et al. 2010										
Infrastructure investments				Zhang 2012; Duflo et al. 2015	Galiani et al. 2005; Soares 2007; Geruso and Spears 2018				Zhang 2012; Spears 2012	Sekhri 2013; Devoto et al. 2012; Zhang and Xu 2016			

Color code:
black: causal analysis
blue: descriptive and non-causal empirical studies
orange: review article
green: qualitative study

A. Online Appendix

Table A summarizes the literature's coverage of water pollution control policies in industrialized countries. Ironically, these countries have created and piloted market-based approaches and promoted them internationally as effective and cost-effective solutions to pollution problems, but we find no causal estimates of the impacts of taxes and tradable permits on water quality in such settings. This is mostly due to the fact that industrialized countries have infrequently implemented such policies. 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. 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 identify 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, may be unlikely.

Table A. Empirical evidence on the impacts of water pollution control policies in developed countries

		Environmental impacts			Health impacts			Socioeconomic impacts				Cost-effectiveness	Unintended consequences	
Water pollution control policies		*Reduction in emissions/effluent*	*Ambient water quality improvement*	*Land use change*	*Reduced morbidity*	*Reduced premature mortality*	*Water supply behavior*	*Capacity building*	*Capital markets*	*Rural development & income transfers*	*Education and cognitive development*	*Cost-effectiveness*	*Strategic regulatory avoidance*	*Rent-seeking*
Prescriptive policies														
	Technology standards		Keiser and Shapiro 2018									Carson and Mitchell 1993; Lyon and Farrow 1995; Keiser and Shapiro 2018		
	Performance standards	Magat & Viscusi 1990; Laplante & Rilstone 1996; Shimshack & Ward 2005, 2008; Earnhart 2004a, 2004b; Glicksman & Earnhart 2007; Shimshack 2014	Keiser and Shapiro 2018									Carson and Mitchell 1993; Lyon and Farrow 1995; Keiser and Shapiro 2018		
Market-based policies	***Pollution taxes***	Bressers 1998; Glachant 2002; Boyd 2003												
	Tradable permits	Olmstead 2013; Environment Protection Authority, State of New South Wales 2018; Connecticut Department of Energy & Environmental Protection 2016	Fisher-Vanden and Olmstead 2013											
	Information disclosure	Khanna et al. 1998; Bae et al. 2010							Laplante & Lanoie 1994; Hamilton 1995; Dasgupta et al. 2006; Khanna et al. 1998				Greenstone 2003; Gamper-Rabindran and Swoboda 2006; Bennear 2008	
	Payments for ecosystem services	Appleton 2002; Depres et al. 2008; Salzman et al. 2018	Appleton 2002; Depres et al. 2008; Salzman et al. 2018	Feather et al. 1999; Roberts and Lubowski 2007; Connor et al. 2008						Baylis et al. 2008		Appleton 2002; Connor et al. 2008; Claassen et al. 2008		
Voluntary approaches		Khanna and Damon 1999; Khanna 2001; Koehler 2007; Innes and Sam 2008; Borck and Coglianese 2009; Vidovic and Khanna 2012												
Infrastructure investments			Keiser and Shapiro 2018			Cutler and Miller 2005; Watson 2006; Delaney et al. 2011; Alsan and Goldin 2018						van der Veeren and Tol 2001		

Color code:
black: causal relations
blue: descriptive and non-causal empirical studies
orange: review article
green: qualitative study