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**ABSTRACT**

We present a lobby model to explain the adoption and persistence of seemingly costly environmental policies relative to the likely benefits generated. The arguments of the model are illustrated by water trade restrictions for mining firms in the Atacama Desert of northern Chile. The area is one of the driest in the world but also the world's top copper producer. Due to regulation of access to local water in the region, firms have begun using desalinated water at a cost of up to $19,542 per m3/day while agricultural water trades at median price of $343 per m3/day. We explore how governmental maintenance of environmental and indigenous water supplies through restrictions on water trades causes these large price differentials. We provide a simple framework that explains how this type of policy can be supported under reasonable assumptions about lobbying. Interest group lobbying, limited information to unorganized general citizens about policy costs and benefits, and their associated distribution can lead to strong regulation, even when the protected environmental areas and agricultural populations are small and isolated. Difference-in-difference modeling of sector prices indicates that after an abrupt increase in regulatory denials, prices diverged in a manner consistent with the lobbying model. Using market price and desalination cost data, policy costs are estimated at $6.15 billion dollars or approximately $350 per citizen, which may or may not equate to perceived general benefits.

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*"Demonstrating that a set of government decisions would improve matters is not the same as demonstrating that actual government decisions would do so. This kind of inference is logically equivalent to identifying the actual workings of the market sector with its ideal workings."*- Gary Becker (1958)

"*We are in a position more and more completely to say how far the waste and destruction of natural resources are to be allowed to go on and where they are to stop. It is curious that the effort to stop waste, like the effort to stop forest fires has often been considered as a matter controlled wholly by economic law. I think there could be no greater mistake."*-Gifford Pinchot (1910) quoted in Roger Congleton (1996, 3)

## I. Introduction

We examine a case in Chile where environmental regulations appear to have costs of $6.15 billion for a country of 17.6 million people by restricting an otherwise influential interest group's access to water, the industrial mining industry, a major contributor to Chilean GDP. These restrictions are designed primarily to safeguard water flows for a very small population of indigenous farmers with extremely low-valued agriculture in the Atacama Desert, some 740 miles or 1,200 km from the major population center in Santiago. About 2,300 hectares are farmed in 17 small indigenous communities with a farming population of 937 (Instituto Censo Agropecuario y Forestal 2007a; 2007b). Inflicting costs on an entrenched group to provide limited and targeted environmental benefits appears to counter the prediction of the regulatory-capture hypothesis that holds that concentrated economic interests dominate policy outcomes (Stigler 1971; Posner 1974). Moreover because the policy costs seem to be so high, relative to likely benefits, the result appears to counter the efficient regulation hypothesis that holds that competition among interest groups will minimize deadweight losses in policy design (Becker 1983, 1985; Wittman 1989).

This case may not be a unique example of the lobbying outcomes in environmental regulation, and the question arises, what lobbying model could lead to such results? In this paper we provide a simple interest-group framework that follows from Olson (1965), Peltzman (1976), Becker (1983), Aidt (1998, 2003), and Yu (2005). Implications from the model are drawn to analyze the costs and benefits of water access restrictions for mining firms in Chile. The case study illustrates the arguments of the model but does not provide a test of its predictions.

The paper is motivated by a relative lack of attention in the literature to the role of interest groups in lobbying for environmental goods provided by government. Given that industry groups are concentrated interests and potentially powerful lobbyists relative to the general population, their role and those of other interest groups in molding environmental policy warrants investigation. That firms capture regulators to secure benefits is well documented in the literature (Laffont and Tirole 1991; Dal Bó 2006). In some settings, however, where regulators choose the level of environmental quality to be provided by policy, the capture hypothesis appears to be rejected by the seemingly high costs imposed on regulated firms. For instance, consider the costs assigned to firms from the Clean Air Act (Greenstone et al 2012); from the closure of the Arctic National Wildlife Refuge to oil exploration and production (Kotchen and Burger 2007); from protection of the Spotted Owl (Montgomery et al 1994); and from EPA pesticide regulation (Cropper et al 1992). There is little attention in the literature on regulatory capture to the role of environmental advocacy groups in countering the influence of industry and how the preferences of those groups may or may not reflect those of general citizens who have higher costs of mobilization (Olson 1965). The conditions when advocacy groups become relatively more influential lobbyists and how their policy preferences can inflict costs both on other organized interests and on the general, unorganized population with societal welfare implications are examined here.

Our framework suggests how high-cost environmental policies might be supported in a lobbying setting.

In the case at hand, mining firms have expanded production in northern Chile, the world's largest copper producer (31% of world production). In the extremely dry Antofagasta region, which provides 17% world production (Cochilco 2014), firms traditionally have purchased water rights from the local agricultural sector to support ore milling and processing. Water rights exchanges are regulated by the Chilean water authority, the Dirección General de Aguas (DGA). Water market transfers typically were approved until 2003, when the DGA, citing environmental concerns, began to deny most transfer applications by mining firms. In response, firms have turned to desalinization at much higher cost.

To assess the effects of water market regulation, a dataset of water right transfers is constructed. Water sales data are not publicly available in many countries, but are in Chile due to a unique reporting requirement equivalent to that required for real estate sales. From 2004-09, water for agricultural irrigation, previously the principal source of water for mining firms, traded at a median sale price of around \$343 per $m^3$/day.[1] The alternative supply, desalinated sea water, must be pumped as far as 150 km (over 93 miles) to altitudes as high as 3,000 meters (9,800 feet) above sea level. Our calculations show the region's largest copper mine, Escondida, presently pays upwards of \$19,542 per $m^3$/day for its desalinated water. Due to these price differentials, the cost of this policy is estimated at around \$6.15 billion, spread between mining firms and the Chilean public, through lost mining royalties. An additional social cost in the form of $CO_2$ emissions from energy generation for desalinization is estimated at $198.5M per year. For the water-trading regulatory policy to be welfare enhancing, its benefits must exceed these costs.

As we describe, however, the environmental areas and agricultural populations protected by such policies are small and isolated, implying that this condition may not be met. Our lobbying framework, however, generates insights as to how such high-cost policies might emerge and persist. Unless relevant marginal costs and benefits are internalized by advocacy groups, the corresponding signals they relay to political decision makers will be unreflective of actual social costs and benefits. When environmental policies are highly valued by a small segment of the population, as may be the case for small-scale indigenous agriculture and isolated environmental sites in northern Chile, the potential for strong environmental protection increases. When the costs of these policies are spread among general citizens and firms, large numbers of heterogeneous citizens are at a disadvantage in organizing to counter an advocacy group that favors more narrowly-valued actions, so long as per-capita costs are relatively low. While firms may lobby to counter environmental interests in some cases, their incentives and ability to do so are often limited, as we suggest is the case in Chile.

## II. Background

### *A. Lobbying and Efficiency*

Interest groups play a critical and complex role in government provision of public goods and regulation, given limited information available to politicians and agency officials regarding citizen demands, policy costs, and distributions. Politicians and agency officials rely on interest groups as sources of information on marginal benefits and costs and to generate political support for taking action. Interest groups form to direct policies in ways their members desire, and their objectives may or may not mesh with those of the broader citizenry.

Becker (1983; 1985) and Wittman (1989) suggested that competition among interest groups could lead to a more efficient political provision of public goods. When this is the case, the process has been viewed as a political market (Peltzman 1976, Becker 1983, Wittman 1989). Like an efficient market, there is competition for government transfers with the political response constrained by the deadweight costs borne by general citizens. In this competitive process no party gains all that it desires as politicians balance group demands and weigh the tax costs facing the broad electorate. Groups that gain most under an efficiency-enhancing policy are those that secure transfers with lower deadweight losses.

The costs of organizing interest groups and of securing relevant benefit and cost information, however, play no direct role in the political-market framework described above. These transaction costs, however, are critical factors in the ability of groups to solve collective action and free-rider problems in communicating with politicians and in describing policy benefits and costs to general voters (Buchanan and Tullock 1962; Olson 1965). Concentrated economic interests, like firms that bear direct costs from environmental policy or benefit from particular transfer policies will have lower transaction costs of organizing and greater incentive to do so than will the general public. Such interests can be expected to organize to exert pressure for favorable government action. This is the so-called capture hypothesis (Stigler 1971; Posner 1974).

Aidt (1998, 5-12) utilizes a model first developed by Grossman and Helpman (1994) to account for both general public and special interest group concerns. He shows that if all societal interest groups are represented, politicians arrive at an efficient allocation in policy decisions. The presence of unorganized citizens, who have a lower value for policy benefits and bear higher policy costs than do interest-group members, introduces inefficiency into the competitive lobbying outcome. Unorganized citizens face higher costs in promoting their concerns relative to specialized interest groups. Hence, citizen demands are not communicated as effectively as are those of more organized parties, and policy makers are therefore more responsive to the demands of the latter. Indeed, Johnson and Libecap (2001) argue that politicians have an incentive to raise the organization costs facing general voters by distorting information on the size and distribution of program benefits and costs when providing preferential programs to specialized interests. If a politician can manipulate the information available to voters by obscuring the benefits transferred to an interest group and overstating broad benefits while understating program costs, overall political support for narrowly-based policies can be increased.

Because of the public goods tied to their objectives, environmental groups may be even more effective in lobbying to advance their interests than are industry groups pursuing ostensibly purely private goods.

*B. Environmental Interest Groups*

Environmental advocates organize to influence the political process. Yu (2005) argues that rather than providing campaign contributions and directly lobbying politicians and agencies, actions for which specialized industry interests have a relative advantage, environmental advocacy groups mobilize broader political support by emphasizing public goods provision. Effective public persuasion by environmental advocates broadens the environmental policies adopted. Environmental advocacy groups that compete with industry groups in influencing government provision of environmental goods generate information about policy benefits and costs to counter the claims of industry. This may lead politicians and agency officials to respond in a manner consistent with broad demand and with reducing deadweight losses. Even so, in the presence of transaction costs in organization not all voters will be included in an advocacy group's campaign and information revelation will be incomplete and potentially distorted. This situation results in a signal of stronger preferences for specific environmental goods, relative to that supplied to politicians and agency officials if all citizens were fully informed and active in lobbying.

Indeed, this skewed outcome of narrow environmental interest group lobbying is more likely when the value of environmental goods and provision costs vary sharply across the population. Heterogeneous preferences among the citizenry raise the transaction costs of mobilization to achieve a broad consensus and hence, reduce signals of general citizen concerns to politicians relative to those of industry and environmental lobbyists (Congleton 1996, 15). Moreover, industry and environmental advocates are not necessarily direct competitors in the political process. Apparently-competing parties can actually lobby for similar positions if they both benefit. For instance, in environmental pollutant regulation, firms may use rules as an entry barrier for competing firms (Buchanan and Tullock 1975). This in turn leads to a cartel-like control over prices and increased producer profits desired by firms, along with the pollution reductions desired by environmental lobbyists (Oates and Portney 2003).[2]

When environmental groups and industry, however, compete in lobbying for environmental protection, environmental advocates are seemingly at a disadvantage because advocacy for environmental public goods is hindered by free riding. Still, these groups appear to be successful relative to industry interests in some cases (Kuzmiak 1991).

For instance, environmental interest groups reduce free riding by lobbying for policies highly valued by their members and by providing club benefits that are denied to nonmembers (Buchanan and Tullock 1962; Olson 1965; Smith 1985, 136-43).

Empirical analysis of this complex lobbying process reveals that environmental policies are responsive to interest-group pressure, but the mechanisms through which this influence manifests vary (Oates and Portney 2003, 331, 339-7; Daley and Layton 2004, 384-90; Daley and Garand 2005, 630-4; Daley 2007, 352-63). For instance, lobby groups are more effective in conveying their positions if they testify before legislative committees that have similar ideologies and policy preferences (Kollman 1997, 529-39). Further, interest group lobbying has been shown to affect administrative agency actions. For example, Ando (1999) finds the speed at which species are listed under the Endangered Species Act is a function, in part, of interest group lobbing of regulatory agencies. Additionally, the magnitude of public health risk and interest group actions have been influential in determining outcomes of EPA pesticide regulation of cancer-causing pesticides (Cropper et al 1992). Finally, organization and lobbying by local environmental groups are associated with EPA superfund-site designation and cleanup (Daley and Layton 2004, 386-9; Daley 2007, 361-3).

## III. Lobbying Model

To formalize the role of pressure groups in the political process, we present a common agency model of political decision making via a simplification of the framework used by Grossman and Helpman (1994). The political process is modeled as a two-stage game where blocks of citizens are represented by pressure groups that influence the decisions of politicians, who determine environmental goods provision. Pressure groups act in the first stage by presenting a payoff schedule specifying the level of the environmental good and their payment for that provision. In the second stage, politicians choose environmental provision and collect payments, ranging from direct campaign contributions to the financing of reports and other activities that generate grassroots support for the policy and the politicians who support it.

*A. The General Case of Many Lobby Groups*

Let $Z$ be the level of environmental damage or costs generated when citizen $j \in \mathcal{J}, \mathcal{J} = 1,2, \ldots, J$ receives benefits from private goods produced by industry. Individual utility functions are $u_j(Z)$ and individuals may be heterogeneous in the value they place on private goods or associated environmental damages. Preferences over $Z$ are assumed to be articulated in the political process individually through voting. Citizens may also be members of lobby groups, and these groups influence policy. Absent pressure groups, it is assumed politicians maximize votes by maximizing total welfare. This assumption differs from the median voter model that concludes that politicians would choose $Z$ such that half of voters would prefer more and half less. While the model here abstracts from the median voter model, this difference allows for a clearer understanding of the effect of interest groups on policy, independent of a particular voting assumption.

Pressure groups are denoted as $i \in \mathcal{L}, \mathcal{L} = 1,2, \ldots, I$, and the set of citizens in pressure group $i$ is denoted as $L_i$. Each citizen is represented by at most one group, but there is no requirement that a citizen be represented. The benefit to an interest group $i$ at a given level of environmental damage is given by the summed utility functions of its members:

$$g_i(Z) = \sum_{j \in L_i} u_j(Z) \qquad (1)$$

When there is heterogeneity among the preferences of groups, there will be competition over the level of desired environmental damage, $Z$. Interest groups seek to maximize the combined utility of their members by choosing a payment schedule, $w_i(Z)$. This function maps the level of environmental damage, Z, into a level of political influence.

In order to exert influence, an interest group must overcome a collective action problem of organizing members and preventing non-members from free-riding on their efforts. In addition, there may be institutional or cultural restrictions placed on the ability of lobby groups to exert influence. In the model, $\gamma_i \in [0,1]$, represents the efficiency of influence for group i.[3] When all γ's are equal, all groups are equally able to convey their utility from a public good allocation to politicians. When group efficiency is not equal, utility is linearly transferable, and a higher γ means a group is better able to communicate the desires of its members to politicians. Factors affecting political efficiency, γ include group size, heterogeneity, wealth, whether there are legal or political limits on certain groups, availability of exclusive club goods that limit free-riding, and whether a preexisting organizational framework exists to reduce mobilization costs.

The politicians' utility function is a combination of total social welfare and campaign contributions:

$$G = \theta \sum_{\mathcal{J}} u_j(Z) + \sum_{\mathcal{L}} \gamma_i \cdot w_i(Z) \qquad (2)$$

The parameter $\theta \in [0, \infty)$, is the social welfare weighting parameter. The first summation term is the influence of voters and is independent of interest group pressure: as θ increases, the politicians' decision approaches one of a social planner who maximizes the sum of all citizen utility. The second summation term describes interest group support and lobby effectiveness for particular policies. Groups with higher efficiency of influence will cause their members' preferences to be weighted more in the political utility function G, all else equal.

The first order condition where the sum of all citizen welfare is maximized is:

$$\frac{\partial G}{\partial Z} = \sum_{\mathcal{J}} \frac{\partial u_j}{\partial Z} = 0 \qquad (3)$$

This equation implicitly defines $Z^*$, the efficient provision of environmental damage.[4] At $Z^*$ the summed marginal gains of citizens who benefit from an increase in environmental damage from private goods production are equal to the marginal losses of those injured by the increase. We now explore how the outcomes of the modeled political process with interest group lobbying can differ from $Z^*$.

First, we must identify the set of strategies interest groups take in specifying their political contribution schedules in exchange for desired policies. Nash equilibrium occurs where the offer of every group $i$ maximizes $g_i$, taking the other groups' strategies as given. Bernheim and Whinston (1986) show that the set of Truthful Nash Equilibrium (TNE) is a compelling refinement of group strategies for these types of political games. Under this restriction, pressure groups truthfully convey their willingness to pay politicians for a change in $Z$. Although this places restrictions on the type of strategy a group uses, there is always a truthful strategy in a group's best-response set; a priori each group can limit itself to a truthful strategy set without an expected loss relative to other strategies. This allows us to reasonably focus on the subset of truthful strategies. The optimal campaign or lobby contribution by a pressure group under TNE is:

$$w_i(Z) = g_i(Z) + K_i \qquad (4)$$

where $K_i$ is a constant that allows lobby groups to capture some of the surplus of an allocation of environmental damage *Z*, while allowing *w* to truthfully convey marginal changes in the welfare of group members. The first order condition for politicians in deciding on environmental policies in the presence of lobbying is:

$$\frac{\partial G}{\partial Z} = \theta \sum_{\mathcal{J}} \frac{\partial g_j}{\partial Z} + \sum_{\mathcal{I}} \gamma_i \frac{\partial w_i}{\partial Z} = 0 \tag{5}$$

Equation 5 demonstrates the influence of lobby groups relative to general citizens. The second summation term shows the marginal lobby group influence, which is proportional to lobbying efficiency γ. As the social welfare parameter $\theta \rightarrow \infty$, lobby groups no longer matter and the outcome approaches that given in equation 3 where the provision of environmental damage approaches $Z^*$. We are interested in characterizing the outcome of the political process, however, where both general welfare and the influence of pressure groups enter into the politician utility function—when θ is finite and policies may stray from the social welfare maximizing provision level.

*B. The Case of Industry and Environmental Advocacy Groups*

To incorporate competitive lobbying, we simplify to a setting with two lobby groups, an industry group and an environmental group, to clearly demonstrate how choice of environmental good provision is affected by lobbying, and show how this choice may deviate from optimal even in a simple setting. The left panel of Figure 1 shows environmental damage increasing along the x-axis. Consumer demand is the marginal private benefit curve MPB(Z), the industry supply function is MPC(Z), and the environmental cost, MSC(Z), is defined such that $\frac{\partial MSC}{\partial Z} \geq 0$. $Z^0$ is the outcome of a competitive market for the private good only, with no consideration for additional environmental social costs. We write the industry and environmental lobby contribution schedules, respectively, as follows:

$$\begin{aligned} w_I(Z) &= Z \cdot P(Z) - PC(Z) + K_I \\ w_E(Z) &= -SC(Z) + K_E \end{aligned} \tag{6}$$

where $P(Z)$ is the market price for a given level of environmental damage, $w_I$ and $w_E$ are the industry and environmental lobby contribution schedules, respectively, and $K_I$ and $K_E$ are constants. Note that the marginal industry lobbying schedule is marginal profit, not MPB because consumer surplus is not captured by the firm. The politician's FOC is:

$$\theta[MPB - MPC - MSC] + \gamma_I[P(Z) + ZP_Z - MPC] + \gamma_E[-MSC] = 0 \tag{7}$$

where $P_Z$ is the derivative of the inverse demand function. The parameters $\gamma_E$ and $\gamma_I$ represent, respectively, the lobbying efficiency of the environmental and industry groups. Equation (7) implicitly defines $Z'$, the lobbying outcome. Whereas $Z^*$, the optimal level of environmental damage from private goods production, occurs where[5]:

$$MPB = MPC + MSC \tag{8}$$

If we assume interest groups are equally efficient lobbyists, $\gamma_E = \gamma_I = \gamma$, we can examine the policy outcome without distortions due to lobbying ability:

$$\frac{\theta}{\theta + \gamma} MPB + \frac{\gamma}{\theta + \gamma} [P + ZP_Z] = MPC + MSC \tag{9}$$

Equation (9) suggests that when reduction in output translates into higher prices for firms, environmental advocates and firms will work together for regulations to reduce output, thereby increasing prices and reducing environmental damage.[6]

The remainder of the paper focuses on the case where industry faces a competitive market with a constant price where no cartelization is possible. This setting is used to outline a series of propositions that we can examine within the framework of our empirical analysis. The right panel of Figure 1 shows firms facing a single price, $MPB = P_0$. The industry again captures the producer surplus, the area between the price and MPC curve. Since the firms are price takers, however, they are best off when producing at $Z^0$, which leads to our first proposition:

*(i) When there is an industry lobby, the absence of an environmental lobby will lead politicians to provide more than the socially-optimal level of environmental damage.*

If, however, an environmental lobby is introduced and group members capture some portion of the environmental benefit, their marginal contribution schedule to politicians for reductions in environmental damage, *Z*, is proportional to MSC. Thus, the introduction of an environmental lobby will serve to push politicians to adopt policies with a lower *Z,* leading to proposition 2:

*(ii) Participation by an environmental interest previously absent decreases the level of environmental damage.*

This proposition does not say whether the reduction in environmental damage that occurs with both an industry and environmental lobby, $Z''$, will be more or less than the socially-optimal level, $Z^*$. That outcome depends upon the relative lobbying efficiency of each group. Explicitly, the government provision satisfies:

$$MPC = P_0 - \frac{(\theta + \gamma_E)}{(\theta + \gamma_I)} MSC \tag{10}$$

where the proportion $\frac{(\theta+\gamma_E)}{(\theta+\gamma_I)}$ determines the weighting of MSC by the lobby efficiency of the two groups.

An interesting complication in the lobby process occurs if citizens also benefit from industry production by sharing in profits through a per-unit tax. This tax, however, reduces the efficiency of the industry lobby:

*(iii) The existence of a per-unit tax decreases the level of environmental damage allowed by politicians.*

A tax, τ, is charged for each unit sold onto the global market. Lobby contribution schedules for industry and environmental interest groups are:

$$\begin{gathered} w_I(Z) = Z(P_0 - \tau) - PC(Z) + K_I \\ w_E(Z) = -SC(Z) + K_E \end{gathered} \tag{11}$$

The politician's FOC simplifies to:

$$MPC = P_0 - \tau - \frac{(\theta + \gamma_E)[MSC]}{(\theta + \gamma_I)} \tag{12}$$

which implicitly defines $Z'''$, the lobbying outcome with a tax. Because MPC is an increasing function and the RHS of (12) is less than the RHS of 10, $Z''' < Z''$: there is less environmental damage than in the case of competitive lobbying without the tax.

Other factors may strengthen the lobby effectiveness of environmental groups relative to industry groups. For example, an environmental group made up of a relatively small number of members that bear most of the environmental cost will be a more efficient lobbyist than a group advocating for regulation of a broad environmental problem.

*(iv) A more efficient environmental lobby group will cause politicians to decrease the level of environmental damage.*

Consider equation 10, with $\gamma_E$ and alternatively with $\gamma_E'$, where $\gamma_E' > \gamma_E$: $\gamma_E'$ represents a more effective environmental lobbyist. Because MPC is increasing in the allocation of damage, Z, is lower with the more effective environmental lobbyist.

We use propositions (i)-(iv) to analyze how apparently costly environmental policies can be introduced and persist even if broad net benefits might be small. Our model demonstrates that because the general public does not lobby, an efficient outcome relies on interest groups balanced in such a way that the preferences of the general public are represented in aggregate, although the groups are not directly reflective of the public's preferences. Because this is unlikely in reality, the specifics of how costs and benefits of regulation are allocated, how lobby groups are able to mobilize, and how politicians and administrative agencies respond is an empirical question, and critical to understanding political provision of environmental regulation. To gain further insight into this process we turn to a case of environmental regulation in Chile.

## IV. Chilean Water Regulation

This case study illustrates the arguments of the model. It focuses on water trading restrictions in Chile's Antofagasta region in the arid northern part of the country that includes the Atacama Desert. The Loa River system provides the primary water source for agricultural and mining production, as well as feeding natural springs and salty wetlands. Although the Loa is relatively small, it is economically significant as the primary surface water source for some of the largest copper mines in the world. Mining is the main economic activity in the area, accounting for 65% of regional GDP. Fresh water is extremely scarce, and the Loa Basin has been declared fully allocated, meaning no new water rights can be granted. New users of water, primarily copper mines that require substantial water for much of the production process must acquire it from other users or pump it from the ocean.[7] Prior to regulatory restrictions an active water market in the region had been driven by increased demand for water in the mining sector. Table 1 shows the estimates of water use by sector, and clearly, mining uses the largest amount of water. The agricultural sector, however, retains a significant portion of the region's water despite producing less than 1% of regional GDP. The importance of the approximately 2,000 hectares of irrigated land out of a total of 12.6 million hectares in the region lies in its link to traditional agriculture and to indigenous communities in the area. The indigenous population represents around 5% of total regional population, although much of the indigenous population is not involved in economically-marginal agriculture, with estimates of those directly engaged around 937. [8] When they have water rights, mining firms extract water from surface dams and groundwater in the highlands above the diversion points for the surface water used by indigenous villagers as well as above the source of seepage for salty wetlands and related ecosystems. As with indigenous agriculture, the regional wetlands are very limited, encompassing an area of around 6,904 hectares, around 0.2% of total land area (Centro de Ecología Aplicada 2011, p. 54). Mining extraction, however, has potential to diminish groundwater outflows to the villages and the wetlands. These are the uses of concern to environmental lobbyists.

*A. Emergence of Environmental Lobbying and Environmental Regulation*

Indeed, water access regulations on mining firms have emerged gradually to protect artisanal agriculture and regional wetlands. These regulations have been relatively new. Under the military government of Augusto Pinochet (1973-1990), neoliberal economic reforms were implemented that moved Chile towards a free market economy. In this process, groups advocating for environmental concerns along with advocates for indigenous rights had little influence and were relegated to the opposition movement (Silva 1996; Tecklin et al 2011). Proposition (i) suggests that the absence of an environmental lobby will lead politicians to provide less than the optimal level of environmental goods. When the country's water code was reformed in 1981 to provide strong, tradable water rights that were separable from the land, few environmental controls were included (Bauer 1998; Mentor 2001) and rights to environmental amenities associated with *in situ* water use were not defined statutorily (Grafton et al. 2011). In the Antofagasta region, the period after the adoption of the water code was characterized by rapid development of water resources by mining firms that received water rights and that purchased others from the agricultural sector. Water infrastructure projects focused on providing water for mining rather than local or wetland uses (Peña 2011). With greater mining water extraction from groundwater basins, streamflow in the Loa River, the main water source in the region, decreased by two-thirds from 1961 to 1990.[9]

Greater environmental emphasis in policy occurred after the fall of Pinochet and the emergence of democracy. This outcome is consistent with proposition (ii). After the military dictatorship ended in 1990, environmental interests organized to lobby for the provision of environmental goods by the government. Following lobbying from scientific, ecological, and indigenous pressure groups, a major environmental law (Chilean Law 19.300) was passed only four years later, in 1994. The law required that large projects, such as those in mining, undergo an Environmental Impact Assessment (EIA). Initially, however, the law provided only limited regulatory oversight of water transfers and these did not importantly affect the mining industry and its access to water. Changes to both the law itself and the rules associated with the regulatory approval process occurred after 1994, and these included greater restrictions on water trades to mining companies. Under post-1994 regulations, a regulatory review of a water transfer is triggered when a water right exchange results in a change in the location of water extraction or use. In a review, the new titleholder must provide the regulator with a description of the quantity and quality of water following the location change. The proposed new use can be prohibited if it causes damage to other rights holders or the environment.[10] If water is currently in-use by a mining firm and the location does not change, the trade is not reviewable by the agency, but even so, the mining project's use of water may still be subject to an environmental impact assessment.

Since 1997, mining firms have been required to provide an EIA assessment on their use of any water in a new project. Gradually, but especially after 2003 in the Antofagasta region, mining companies have purchased water rights only to have the EIA reject the use of water in the project.[11] Between 1999 and 2003, 70% of proposed water rights trades were approved (Figure 3). Beginning in 2003, however, regulatory approvals abruptly stopped with only one application for a change in surface water approved since that year. Figure 2 summarizes data on DGA reviews from 1995 to 2010 from the *Catastro Público de Aguas* of the DGA. The dashed line shows the cumulative number of transfer requests by year of request, while the solid line plots the total number of denials by year of decision. There is a clear jump in denials in 2003 when several water right requests were blocked over a short period of time. From February 2003 to July 2004, five surface transfers representing 46% of the total amount of water transfer requests submitted up to that date were prohibited. Figure 3 shows the overall rejection rate of water transfers over time. In 2003, the rate of rejection jumped from 30% to 50%. The increased difficulty of getting freshwater use approved has led some mining firms to build desalination and pumping facilities to utilize ocean water at very high cost.

*B. The Nature of Lobby Advocacy*

*Industry*

Mining firm lobbying outcomes regarding water regulations in Antofagasta, especially after 2003, appear to be inconsistent with the capture hypothesis, whereby a concentrated industry succeeds in limiting the regulatory costs of environmental good provision. In the region, around 32% of copper is produced by the state-owned firm, Codelco, and over 90% of the remaining production is from five multinational mining firms (Cochilco 2014). Although we do not have data on lobby expenditures by the mining industry, it seems likely that after enactment of the water code in 1981 mining firms expected water transfers to be approved and therefore had little reason to lobby intensely to protect access via water markets. Two related factors also help explain the apparent absence of strong lobby efforts by the industry and the appearance of greater regulatory restrictions after 2003: foreign ownership and royalty payments. Proposition (iii) suggests that where firms are price takers, the existence of a royalty tax can decrease the level of environmental damage allowed by the government because industry lobbying incentives to protect profits through costly lobbying are reduced. Additionally, foreign ownership creates cultural and political restrictions on lobbying effectiveness by those firms.

On the first point, Chilean government mining income arrives through two channels: direct government ownership of mining interests and tax and royalty payments from private copper production. For large copper mining firms, those with sales over 50,000 metric tons, the Chilean government placed a tax burden ranging from 5-14% on operational income starting in 2005 (PricewaterhouseCoopers 2012, p. 21).[12] All profits from the Chilean national copper company, Codelco, go to the central government.

On the second point, starting in 1990 non-government, primarily foreign multinational, mining expanded rapidly. Figure 4 shows copper production in the Antofagasta region, broken down by ownership. By the mid-1990s private multinational ownership of mining production exceeded that held by Codelco. These firms, however, are reluctant to exert pressure on the government because doing so potentially puts at risk the profitability of mining concessions by drawing the ire of powerful nationalist constituencies, including local environmental and indigenous groups. For instance, the 2005 Specific Mining Tax "enjoyed strong inherent popularity and legitimacy based on nationalist sentiments (Fairfield 2014)." Foreign mining firms also have invested less in lobbying than might be expected because water in mining is a relatively small input cost and because a portion of the cost of regulation is offset with reduced royalty payments.

*Environmental Advocates*

On the other hand, there are factors that would strengthen the lobby efforts of environmental groups. If the efficiency of environmental lobby groups increases, proposition (iv) predicts that the provision of environmental damage via government regulation will fall. As above we do not have data on environmental group lobby expenditures but we do observe the emergence of factors that would likely increase the productivity of such lobbying. One factor is rising urban per capita income. By 2003, Chile had the most successful economy in Latin America. In 1995, for example, per capita income in current dollars in Chile was 68% of that of neighboring Argentina, but by 2003 Chilean per capita income was 146% of Argentina's (Authors calculations from Worlds Bank Data). The relationship between increased income and demand for environmental protection is inherent in the environmental Kuznets's curve (McConnell 1997).

Another key factor is the ability to link an otherwise relatively abstract environmental objective, such as protecting stream flows and wetlands, with a more concrete and perhaps more salient objective among urban voters, such as protecting indigenous populations. Indeed, Rojas (1994) argues that environmental groups in Chile have been most successful at influencing policy when "the environmental movement converges with native peoples in their effort to defend their culture." By 2000 concern about native cultures was growing in Chile so that there emerged a combination of environmental and indigenous lobbying efforts emphasizing: "local livelihood, culture and environment" (Urkidi 2010). The effort to combine indigenous and environmental issues was formalized in 2010 when the Chilean Environmental Law 19.300 was modified to include the increasing concerns about indigenous communities. The amended law requires environmental actions to include "the proper conservation, development and strengthening of identity, language, and social institutions and cultural traditions of the peoples, communities and indigenous people." Similarly in 2012 a new regulation was approved for environmental assessments requiring consideration of the effects of the projects on indigenous communities.

Peña (2004) points out that in areas dominated by Aymara and Atacama cultures, the native people of the Antofagasta region, groups have formed to advocate for legislation that consolidates the ownership of water rights in native communities, rather than individuals. The legislation promoted by these interests would impose restrictions on or eliminate entirely the ability of individuals to transfer or sell water. Corresponding to these lobbying efforts, a new group of water buyers have emerged to purchase water rights and hence, prevent their sale to mining enterprises. These buyers include representatives of native villages, irrigation organizations, and government agencies charged with serving indigenous communities. Other lobby groups have been active in advocating for the preservation of indigenous cultures through direct constraints on water transfers to the mining sector from agriculture. In the city of Calama, interior from the coastal city of Antofagasta, 2,260 meters (7,415 feet) above sea level, and the closest urban area to several of northern Chile's largest mines, interest groups like the *Coordinadora por la Defensa del Agua* have formed to oppose new mining water use and to lobby the DGA to halt further water trades from agriculture. Further, the group *Human Rights Everywhere* criticized the move by the Antofagasta water utility that owns gravity-fed inland water rights, to trade 550 L/s of freshwater rights (92 acre feet) to an inland mine in exchange for desalinated water provided by the mining company near the city.

These narratives are suggestive of a shift in lobby influence between the mining industry and environmental and indigenous group lobbyists after 2003 with greater regulation of water transfers to mines. Although we cannot directly test the propositions of the model with available data, we can obtain a more precise understanding of the growth of regulation and of its aggregate costs with water trading data that are available.

## V. Data and Empirical Approach

### *A. Data on Water Transfers*

Water rights in Chile, once granted, are fully protected as private property under the Chilean Constitution (Mentor 2001). Under the 1981 Water Code, water rights are completely separated from land ownership and may be freely bought, sold, mortgaged, and transferred, like other forms of real property. Rights are subject to real estate title registration and transfers are registered with real estate certifiers so that characteristics of the parties involved in a sale as well as the price become public record.[13] In the Antofagasta region farmers, indigenous communities, water utilities, and mining companies have traded water rights in water markets, but successful trades between sectors are no longer common. To analyze the effect of regulation on water markets, a dataset of surface water transfers was obtained from records in the Loa River basin, near the city of Calama in the Antofagasta region and the center of mining activity in Chile, for the time period 1995 to 2009. This geographic location was selected because it is the primary surface water source in the area and the availability of data: transaction records are compiled by the Superintendencia de Servicios Sanitarios for use in setting water rates to urban customers. Records were obtained only for the "Official Database" for the entire time period, which had already eliminated trades beyond the Loa basin, those with missing water price data, exchanges involving land and buildings where water is bundled in the transaction, groundwater transfers, and trades among family members.

The data set consists of 442 observations for which information is available on the buyer, seller, quantity transacted, and the price at which the transfer occurred. Purchasers are classified into three categories: government buyers that are communities and the federal government that buys and transfers water to indigenous communities; private parties, who generally are farmers; and industry buyers that are primarily mining firms. Table 2 provides summary statistics on these transfers. Because of the 2003 change in the rate of regulatory denial, we provide the median price, number of transfers, and mean of total quantity transferred for the preceding years (1995-2003) and following years (2004-09), as well as for the entire dataset (1995-2009).

Sales are broken into category by buyer type. For the full time period, the median price for exchanges is higher for governmental and industrial buyers than for private buyers for agriculture. The number of sales is evenly distributed among the three groups in the pre-denial period, while the quantity of water industrial users purchase tends to be larger than private and government buyers. Following 2003, industrial and private parties participated in fewer transactions relative to government buyers, and the average quantity transferred per sale decreases for all buyer types. The overall trend of increasing median prices, in real terms, is observed across all three groups from pre- to post-denial periods. It is notable, however, that mining firms always pay more for water than do private, agricultural purchasers. Regulation that restricts water transfers holds water in agriculture, explaining the continuing high water consumption in low-valued agriculture indicated in Table 1.

When mining firms cannot secure local water, they turn to desalinization and transport of water from sea level to remote mines (Cristi et al 2014). We have assembled data on these costs in order to compare them with the observed market price for local water. All projects in northern Chile beyond the planning stages, those that are currently operational or have passed their environmental impact assessment, are included in the dataset. Table 3 shows the desalinization plants associated with particular mines and the cost of desalinated water for each plant.[14] Mine locations are fixed and pumping cost is estimated based on elevation and distance from the ocean.

Figure 5 shows a map of the region and includes the locations of the mines to which each project pumps. The figure also shows the main local water source for the region, the Loa River, which forms a "U" near the border with the Tarapacá region.[15] The small triangles represent locations of indigenous agriculture and principle wetland sites, although only a portion of these are active wetlands. The mines of Esperanza, Sierra Gorda, and Algorta Norte lie near a gravity-fed pipeline bringing Loa River water to the coastal city of Antofagasta. The second and third columns of Table 3 provide data on the pumping distance and elevation as required with desalination, which are the variable determinants of water price shown in column four. The desalination process itself has costs of around $1.00/m$^3$ of water with pumping costs around 6¢ per 100km horizontally and 5¢ per 100m vertically (Zhou and Tol 2005).[16] Because seawater is 1.025 times heavier than desalinated water, direct seawater projects, where seawater is shipped to the mines, impose a slightly higher pumping cost, but no desalination cost.[17] Capital costs of constructing pipeline facilities and pumping stations are not included.[18] The present value of a perpetual payment for desalinated water is calculated so that desalination price is in units equivalent to a permanent freshwater rights transfer.[19] The equivalent permanent price for a right to one m$^3$/day of water, Column 4, varies from $4,265 for direct seawater at the Michilla mine, near the coast, to $19,542 at the Escondida mine further inland. Project status is provided in Column 5. Columns 6 and 7 provide details on the capacity of each plant in terms of direct seawater or desalinated water. All of the cost estimates are orders of magnitude higher than the cost of local water shown in Table 2.

*B. Empirical Strategy: Water Market Restrictions and their Costs.*

A difference-in-difference estimation approach is used to find the effect of increased regulatory denials on water right prices after the 2003 increase in regulatory rejections. The past rejection rate would have been a useful indicator of expected future rejections for buyers. The jump was unanticipated by the market and provides a test of the extent to which observed market price distortions are caused by environmental regulations.

This test precedes our cost estimate to demonstrate that DGA policy, as opposed to other factors, prevent water reallocation from agriculture to mining and justifies treating price differences as policy costs.

Water rights that have high probability of passing a review sell at higher prices, while those that have a lower probability sell at lower prices. Thus, the water market is segmented by the regulatory framework into buyers who purchase rights likely to pass regulatory approval and those that do not. Industrial and government buyers are likely to selectively attempt to purchase high-probability-of-approval water rights. Industrial buyers do so to use the water, whereas government buyers do so to retire the water rights into agricultural use (World Bank 2011).[20] The low-value rights are purchased primarily by private individual agricultural producers who do not need regulatory approval as the water remains in agriculture. Thus, we use buyer type as a proxy for market segmentation to examine the price response to a sudden decrease in regulatory approvals. Because the decrease in approvals is expected to increase the supply of water rights in the market with a low probability of approval after 2003, the expected outcome post-2003 is a decrease in the price of water paid by private buyers, relative to industrial and government buyers. Our identifying assumption requires parallel trends, that is, absent the regulatory shock the relationship between the price paid by private buyers and industrial or government buyers would have remained the same after 2003.

Accordingly, define an indicator variable such that $I_R(t)=1$ when time $t>2003$ and $I_R(t)=0$ otherwise, where 2003 is the change in regulatory approval. The price of a water right j per unit (cubic meters per day) of type q is $Y_j$. We regress the log of $Y_j$ on indicators $I_q$, where $q=\{I,G\}$. For each observation j, $I_I$ takes on 1 if it includes an industrial buyer and

0 otherwise; $I_G$ is 1 when there is a government buyer and 0 otherwise. The third potential indicator for private agricultural buyers is excluded from the regression—the expression for an agricultural buyer occurs where $I_I$=$I_G$=0. Generally, our empirical specification takes the form:

$$\log(Y_j) = \sigma + \beta \cdot I_R + \sum_{q \in \{I,G\}} \gamma_q \cdot I_q + \sum_{q \in \{I,G\}} \delta_q \cdot I_R \times I_q + \tau_t + u_j \quad (13)$$

In the above specification, $\tau_t$ is the unobserved time-period characteristics that are constant across types of water sales. $\tau_t$ can be controlled for by including a quadratic time trend or an individual control variable for each year in the sample. When using a quadratic time trend, copper price is also included as a regressor. When individual year controls are used, copper price and the term $I_R$ do not contain any variation not included in the controls and are therefore dropped.

The coefficient $\sigma$ is the average log sale price to agricultural buyers pre-2003. All other coefficients are interpreted as a change from this coefficient, or the change relative to pre-2003 agricultural buyers. The estimates of $\gamma_q$ give the pre-change price level for industrial and government buyers, relative to an agriculture purchased right. The coefficient estimate of $\beta$ gives the overall effect of the change in rejection rate on the price for agricultural buyers. We test the relative change in the pricing of water rights of different buyer types pre- and post-2003 by examining the coefficients $\delta_q$. We expect positive coefficients for both industrial and government buyers, indicating relative to agricultural buyers, decreases in the regulatory approval rate decreased the relative price that private agricultural buyers paid for a water right. Regulation constrained water to remain in the agricultural sector.

*C. Welfare Calculation of Water Market Restrictions*

To estimate the private welfare cost of regulatory restrictions on water trades, partial equilibrium is assumed—the effect of farming on the regional economy is extremely small and the effect of high water prices on copper mining investment likely is also small when world copper prices are high, which they were throughout the study period. Figure 6 graphically illustrates the approach we use via a three-axis plot in a framework developed by Griffin (2006, 40). The total surface water availability is represented on the x-axis. Demand for water is represented in terms of percentage of total surface water rights available. The mining demand curve for consumptive use of water is $CD_M$. The demand curve for consumptive use of agricultural water, $CD_A$, has been flipped to serve as a supply curve. Thus, movement from left to right along the x-axis represents an increase in the percentage of total water used by the mining sector.[21]

We estimate the private costs of regulatory restrictions on water trades. The estimate provides the denominator for a benefit-cost ratio and indicates the value that the public benefits of regulation, as the numerator, would have to be for the ratio to equal to one. The welfare gain from water transfers to mining from agriculture is shown as the shaded grey triangle in Figure 6. $\hat{Z}$ is the allocation of water post-policy implementation, and $\hat{P}_M$ and $\hat{P}_A$ are consumptive surface water prices in mining and agriculture, respectively. Because agricultural water is only partially used consumptively, the average price observed in the agricultural market for a water right, $\hat{P}'_A$, is related to $\hat{P}_A$:

$$\hat{P}_A = \frac{\hat{P}_A'}{1 - \mathrm{R}_0} \tag{14}$$

where $R_0$ is the proportion of the water from the agricultural right that returns to the system. We estimate $CD_A$ using $\hat{P}_A$ as a point estimate to calibrate a CES demand curve using previous estimates of agricultural price elasticity and an assumption of a constant elasticity of substitution (CES) demand function (Lichtenberg and Zilberman 1986).

Because each right purchased by a mining firm has an unobserved risk of denial, we do not attempt to calculate the mining water demand curve, $CD_M$, using water market prices. Instead, current and planned desalination projects in the region, which represent the backstop source of water, are used as a measure of mining firm willingness to pay for local water rights. The key environmental concern with water right exchanges to mining relies on the fact that mining firms extract the water far upstream, so it can be diverted to mines. The effect of these mining water extractions is a reduction of downstream surface water availability to agricultural users, but not necessarily at a 1:1 rate. We parameterize the factor that determines the actual decrease as β. If β=0, there is no effect of mining extractions on surface agricultural water users, while if β=1, a one-unit withdrawal of water leads to a one-unit decrease to agricultural users. Therefore, the importation of desalinated water of quantity $q_d$ offsets a quantity of surface water use, $q_{s=}$ $\beta q_d$. We use the current estimate of water use by sector in the region to fix $\hat{Z}$, then determine the intersection $Z^0$ using the slopes of the demand curves. The area between $CD_M$ and $CD_A$ between $\hat{Z}$ and $Z^0$ then is the estimated policy cost.

## VI. Results

*A. Response to Regulatory Change*

We begin with analysis of the buyer response to the 2003 regulatory change. The abrupt nature of the change allows us to test the effect regulations have on the price of water rights. The analytic framework assumes that governmental rules restrict the transfer of water and this action results in differential prices paid by different buyer types. Before calculating the magnitude of these costs, analysis of the regulatory change demonstrates the extent to which government rules actually affect market prices.

Figure 5 shows trends in average price weighted by quantity of sale over time for private party (agriculture) and combined industrial-government buyers, with prices in 2014 dollars. In the five years 1999-2003, prices for both groups track closely. This occurs during a period with relatively high approval rates, and the change in this relationship coincides with the increase in disapprovals; for the five years from 2004 to 2008 private party agricultural prices are lower. The post-2003 downward trend in price for private agricultural sales relative to industrial and government sales is expected because the reduction of approvals locks additional water rights into the low-value sector, decreasing price. We now turn to the approach laid out in the previous section to test these trends statistically.

Table 4 presents the results of eight empirical models all based on the general model from equation 13. Models (1)-(4) are for the full time period for which we have data, while models (5)-(8) are for a six year window, 2001-2006. The six-year window is used to isolate the change after 2003 from regulatory or other changes, such as the requirement for an additional EIA review for water right changes that was enacted in 1997.[22] Returning to

Figure 5, the pre-treatment period 2001-03 captures the parallel trend of increasing prices in the two categories of buyers.

In the models we regress the log of per-unit water right price on buyer type, and then interact buyer type with post-2003 change in approval rate. The increase in disapprovals is predicted to decrease the relative price that agricultural users pay for water. This will be apparent in positively signed coefficients on the interaction terms between government and industrial buyers and the post-2003 indicator. In models (1), (3), (5), and (7) Industrial and Government are treated as two separate categories of buyers, while the even numbered models combine these groups into a single variable. For the period prior to the change in approval rate, coefficients on Industrial, Government, and Government/Industrial represent the pre-trend in prices relative to agricultural buyers. A coefficient $\gamma_q$ represents a $100 \times (e^{\gamma_q} - 1)$ percent increase in price paid by buyer type $q$ over a private buyer. For instance, in model (3) the coefficient on Industrial is positive and significant, $\gamma_I = 0.513$, meaning the industrial buyers paid more for water than private buyers, on average about 67% more over the period 1995-2003. Similarly, industrial buyers paid even more, as indicated by the coefficient $\gamma_I$. Importantly, however, for the relevant three-year window *before* the change in approval rate, there is not a statistically meaningful difference between the prices paid by any of the three types of buyers.

All models include controls for external trends in the economy that might influence the prices of all water rights. In models (1), (2), (4), and (5), the time controls are quadratic, and we have included copper price as an additional control, because it is likely an important driver of water demand. In the other models, we control for individual year effects. These controls remove all year-to-year changes in overall water right prices. The variable Post-

2003 is a dummy that switches on after 2003. Its coefficient represents the change in the average price of water paid by agricultural buyers, post-2003. The point estimates are slightly negative in all specifications for which they are included, indicating a downward price trend after 2003.

It is necessary to examine the interaction coefficients to get an idea of whether a relative price change occurred following the anticipated reaction to regulatory change. Based on the data, government buyers appear to be a better counterfactual to private buyers as they track agricultural prices more closely than do industrial or combined industrial/government prices for the period up to 2003. As anticipated, all the measures of the prices paid by government post-2003 relative to agricultural buyers have positive coefficients. These results become statistically significant when smaller windows around 2003 are used.

The industrial sales do not have the anticipated sign, but this is due in part to the absence of industrial purchases of water from 2004-06. This prevents us from estimating the post-2003 effect for industrial buyers for the shorter window, and suggests the industrial response to the regulations might have been to hold off on water right purchases. However, by pooling industrial and government buyers, all post-2003 years have price observations. For the full sample period, the results show limited statistical significance. However, for the six-year window around 2003, the results become significant at the 99% confidence level. In specification (8), the coefficient on the interacted term ($\delta_G = 1.585$) estimates a 388% increase, after 2003, in the difference between the price paid by industrial and government buyers, relative to agricultural buyers.

As a robustness check, a placebo test is run where each year, 1998-2006, is treated as if it were the year in which the approval rate decreased. The same regression as model (8) is then run for the six year window around the placebo treatment. We expect that coefficient $\delta_G$ should not be less than zero and statistically significant under the placebo treatments. Figure 8 shows the results of these tests by plotting the point estimates of $\delta_G$ with a 95% confidence interval for each year. The only treatment year that shows a significant negative result is 2003, which is consistent with our prior that only 2003 saw a price decrease by non-government buyers. Although there was not a documented change in regulatory policy in 2003, Figures 2 and 3 show the empirical evidence of a significant policy shift on DGA approvals in the Antofagasta region. The DGA implemented formal policies corresponding with these changes in 2005, when it required all change applications go through a rigorous EIA review. The placebo test provides evidence it was the 2003 disapprovals that changed market prices. Although not shown, similar placebo tests were run for eight year windows, with 2003 being the only treatment year with a statistically significant, at the 95% level, negative coefficient.

*B. Welfare Cost*

Per unit desalination costs are far in excess of the market price of water. While the cost of desalination ranges from \$4,265 to \$19,542 per $m^3$/day, water in the market traded at average prices ranging from \$343 to \$655 per $m^3$/day for the post-2004 period. Yet according to the data in Table 3, over 673,000 $m^3$/day of desalinated and direct seawater is in use or in an advanced planning stage in the region. Some of these projects would not be undertaken if government restrictions did not limit water transfers to mining firms. To calculate the policy costs, water market price data and estimates drawn from a variety of sources are used to parameterize the model shown in Figure 6 and described in the prior section. Estimates of four parameters are needed: $\hat{P}_A$, the price of water in the low-value agricultural market; $R_0$, the effect of upstream water extractions on downstream surface water availability; the price elasticity of demand for water in the low-value agricultural market; and estimates of the total amount of available water.

To estimate $\hat{P}_A$ the weighted mean of sale prices to private individuals for the period 2004-09 is used in Equation 14. DGA officials indicate that it is typically assumed that $R_0$=0.80 for agricultural water use. This low rate of consumptive use occurs because agricultural water rights are not continuously used, not because the water is reused after agricultural application. We use an estimate for the price elasticity of demand for water use of -0.79 (Schoengold et al. 2006). Total recorded agricultural surface water rights on the Loa River and tributaries of 2,149 L/s (185,674 $m^3$/day) are used as the estimate of current agricultural water use (DGA 2005; Salazar et al 2003). We use calculations for the Ojos de San Pedro sub-basin within the Loa River watershed to estimate $\beta$=0.45 (Edwards and Kirk-Lawlor 2013), the impact on downstream flows from water diversions to agriculture.[23]

According to these estimates, marginal prices are equalized when 168,308 $m^3$/day of water is transferred from agriculture to mining, offsetting 374,018 $m^3$/day of pumped desalinated and direct seawater. These numbers suggest a present value cost of $6.15 billion arising from current water trading restrictions. As in Kotchen and Burger (2007), the environmental and other public good benefits achieved through time must be at least this high for the regulations to have a benefit-to-cost ratio of 1.

We provide a sensitivity analysis in Figure 9 by changing the agricultural elasticity of water demand $\eta$, agricultural consumptive water use $R_0$, quantity of agricultural water available for transfer $Q_0$, and the damage coefficient of groundwater extraction $\beta$ on downstream uses from their point estimates. The dashed vertical lines show a 95% confidence interval based on the distribution of the market price transaction data.[24] The horizontal lines represent the high and low cost estimates of changes in each parameter value. For agricultural water demand elasticity a range from -0.25 (Nieswiadomy, 1988) to -0.9 is used. The $R_0$ parameter is varied between 0.7 and 0.9 and the $\beta$ parameter between 0.25 and 1. The cost estimate is very sensitive to $\beta$, a parameter regarding released water after initial use that is related to properties of aquifer rock and other factors which are not well known throughout the region, suggesting acquiring this information would be valuable. For $Q_{0,}$ a range was chosen based on a low estimate of available water using gauged flow in the Loa River of 129,600 $m^3$/day[25] and the estimate of water use in agriculture from Table 1 of 285,811 $m^3$/day.

*C. Discussion*

Our lobbying framework suggests such a high-cost policy is adopted because the policy benefits are local and are highly valued by an influential group of environmental and indigenous advocates. These benefits include the protection of specific environmental resources and retention of water in indigenous agriculture. The costs of the policy are distributed to foreign mining firms and to general Chilean taxpayers through lost mining royalties. The mining royalty rate on Chilean copper ranges up to a 5% tax on revenues minus production and financing costs (Cademartori et al 2011). The policy cost of $6.15 billion is distributed primarily to mining firms at $5.84 billion. The incidence of policy costs is largely on multinational mining firms that have limited lobbying power. In contrast, Codelco, the government-owned producer is less restricted in access to water and avoids many of these costs.

The portion of the policy cost that falls on the Chilean people through lost royalty revenue, around $307.5 million, may not be well understood by the Chilean public. The per capita policy cost is $17.50 per Chilean. Given the low per-capita cost, and the difficulty and cost of understanding the issue of lost royalty revenue and organizing to oppose it, the lack of response from the general public is predictable. Available data suggest that few Chileans are aware of the costs of water trade restrictions or would value the policy benefits equal to those costs. Although in recent surveys 55% of Chileans report concern about environmental issues, only a quarter feel knowledgeable about causes of them and only 16% feel knowledgeable about solutions (CEP 2010). In the case at hand, knowledge may be particularly lacking because of the remoteness of the sites from the general urban Chilean population.

The overall environmental benefits of the policy may not be positive. A back-of-the-envelope calculation shows that the projected increase in electricity use for desalination as a result of the policy leads to the release of an additional 4.62 million metric tons of $CO_2$ per year.[26] According to IPCC estimates this amounts to an additional cost of $198.5M per year in worldwide social costs.[27] Because these costs are broadly spread, there is limited lobbying pressure to reduce carbon emissions. If Chile undertakes actions to reduce carbon emissions, either unilaterally or through international agreements, the specific policy design will determine the incidence of these additional costs and whether these costs will fall on influential interest groups or the general population (Libecap 2014).

Because a key function of a market is to provide information, the ability of water markets to achieve their potential benefits requires designing property rights and trading systems that make regulatory rules explicit in asset definition. If not, rules designed to limit environmental damage may result in price differentials that fail to convey signals of scarcity to policy makers. The sharp increase in regulatory denials in Chile dramatically decreased the price of water rights, even in the face of rising demand and severely constrained supply in the Atacama Desert. A thorough understanding of what restrictions govern water trades is necessary to understand water right prices and their response to shifting demand. Investment in alternative water sources, for instance costly infrastructure investment in desalination or new dam construction, may be undertaken even when water prices are relatively low if these sources of water are restricted for trades, resulting in water remaining in low-value sectors at low prices.

## VII. Conclusion

In the Antofagasta region of northern Chile, a high-cost environmental good is provided that benefits a narrow constituency while imposing costs on large, dominantly foreign, mining firms and the Chilean public. To understand this outcome, it is necessary to examine the relative power of lobby groups, how they compete, and how completely they reflect broad citizen interests. When the involvement of the general public and industry is low, a higher level of environmental provision may be chosen by politicians that exceeds what is general welfare enhancing.

In lobbying and in responding to interest group pressures, advocates, politicians, and agency officials do not bear the full costs of the policies adopted. In this sense the problem of social cost (Coase 1960) emerges because the private costs facing decision makers and those influencing their decisions are less than social costs. Excessive provision, in this case, environmental benefits, follows. The model demonstrates the necessity of examining these types of cases empirically to assess the extent to which the preferences of lobby groups deviate from those of the general public. Divergence between private and social cost and benefits creates incentives for lobbyists to favor allocations that increase their benefits while potentially reducing total welfare. Welfare-improving regulatory policies require competitive lobby pressures to generate outcomes that generally reflect broad public interests.

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### *Endnotes*

[1] Assumptions and data sources are described later in the paper.
[2] A similar case is the coalition of "Bootleggers and Baptists" who oppose liquor sales on Sundays for different reasons but form a strong lobbying coalition (Yandle 1993).
[3] We assume for the sake of modeling simplicity that all politicians accept lobbying influence.
[4] This outcome implies that lobby groups are not beneficial, relative to a world without them. Of course, groups may provide useful information as to what can happen to the environment or to industry with changes in Z. Indeed, once it is assumed that there are lobby groups, the formation of a lobby group can provide valuable information to politicians.
[5] An interior solution is assumed—at least some production occurs.
[6] Whether this output is below the optimal level, $Z' < Z^*$, is determined by whether the firm, as a monopolist, would choose $Z<Z^*$. Thus, the overall welfare effect of the industry-environmental group coalition is ambiguous.
[7] New permanent and consumptive water rights to underground water cannot be granted if DGA declares a hydrologic sector as a "restricted area" (Water Code, numbers 65-67). This is the case of most underground water in the Antofagasta region. Moreover, DGA can approve a "Resolution of water exhaustion" (Water Code, number 282) for a natural source of surface water, and in this case DGA cannot grant new consumptive and permanent water rights. This has been the case for the Loa River and tributaries since the year 2000.
[8] Data from the 2002 census shows 22,808 people "belonged to an ethnic group" in the Antofagasta region, out of 493.984 (4.62%) (Instituto Nacional de Estadísticas 2008). Indigenous agriculture participation data is from the Instituto Censo Agropecuario y Forestal (2007a)
[9] Measurement data from 1961 is from Universidad de Chile and Corporación de Fomento de la Producción; data from 1990 is from DGA monthly means.
[10] Interview with Ricardo Katz of the DGA conducted by Cristi and Edwards in 2013.
[11] As an example, the mining company Quadra Minino bought underground water in 2008 in the Pampa Llalqui, 30 kilometers from Calama, for around US$40 million, but has not been able to use it for environmental reasons.
[12] Revenues minus production costs including depreciation (Cadematori 2011).
[13] According to Chilean law, these Property Registers (Conservadores de Bienes Raíces) provide official certification of legal tenure.
[14] Note the municipal facilities La Chimba and Desaladora Sur are located in the city of Antofagasta.
[15] The river's surface water disappears somewhere after Calama although the river bed continues to the Pacific.
[16] $\$0.63/m^3$ is the total variable cost of desalinated water, an average including capital costs is around $\$1.00/m^3$, with $\$0.44/m^3$ from the cost of electricity.
[17] Our estimate for direct seawater projects is an underestimate because it does not include the cost of upgrading the on-site equipment and ore processing changes necessary to use seawater, which have proved problematic and expensive.
[18] We exclude these capital costs to simplify the analysis. It may be in the company's interest to build the infrastructure to ensure a reliable water supply, and the usable lifetime of these investments is less clear than for desalination plants for purposes of amortizing capital costs over time.
[19] Brewer et al. (2007) discount the water flow itself in comparing one- and multi-year leases with permanent transactions. Using a similar approach we convert the desalinated and seawater costs per year into the equivalent permanent water right price. Example: To convert a cost of $\$X/m^3$ to the equivalent permanent right price, we divide by 1000 to get the price per liter, then multiply by 365x24x60x60 to get the number of seconds per year. This is the equivalent to providing 1 L/s for an entire year at $\$X/m^3$. We then divide this single-year price by the interest rate, like a perpetuity, to find the equivalent permanent right price.
[20] Purchases are made through the Fund for Indigenous Lands and Waters.
[21] This inter-sector allocation only examines the change in profit due to changes in water input. Mining firms face a constant price for copper and lobby according to the relationship shown in Figure 1B.
[22] The six-year window is somewhat arbitrary, balancing a desire to isolate the change with the need for enough observations to identify the effect. Four- and eight-year windows yield similar results.
[23] In this region, estimated water withdrawals of 1,551L/s reduced surface water flow downstream by 700L/s
[24] Calculated by finding the unbiased variance of the weighted mean of the log of per unit transfer price.
[25] Personal correspondence with Naomi Kirk-Lawlor, May 19, 2014

[26] Total project energy projections imply around 3.08B kWh of electricity per year based on $0.10/kWh electricity price. The Northern grid is 33% coal, 57% natural gas, remainder fuel oil and diesel (http://www.cdec-sing.cl/html_docs/anuario2010/pdf/SING2010EN.pdf). $CO_2$ releases are 2.1 lbs/kWh for coal, 1.12 lbs/kWh for natural gas, 1.7 lbs/kWh for fuel oil no. 6 (http://www.eia.gov/tools/faqs/faq.cfm?id=74&t=11). This leads to an estimate of 4.62M metric tons $CO_2$

[27] The social cost of carbon $43.00 (http://www.ipcc.ch/publications_and_data/ar4/wg2/en/ch18s18-4-2.html) → $198.7M/yr

***Tables***

**Table 1. Freshwater Consumption by Sector in the Antofagasta Region**

| **Economic Sector** | **Consumption ($m^3$/day)** | **Percentage** |
|---|---|---|
| Agriculture & Livestock | 285,811 | 25% |
| Urban | 94,608 | 8% |
| Industry | 164,678 | 15% |
| Mining | 419,472 | 37% |
| Energy | 128,995 | 11% |
| Other | 29,117 | 3% |
| Total | 1,122,682 | |

Sources: DGA, 2007, DGA, 2008.

**Table 2. Surface Water Sale Summary Statistics**

| Buyer Type | Pre (1995-2003) | Post (2004-2009) | All (1995-2009) |
|---|---|---|---|
| | *Median Price (2014\$/$m^3$/day)* | | |
| Government | $444 | $655 | $518 |
| Industrial | $305 | $966 | $462 |
| Private Party (Ag) | $144 | $343 | $196 |
| | *Number of Transactions* | | |
| Government | 83 | 80 | 163 |
| Industrial | 87 | 44 | 131 |
| Private Party (Ag) | 99 | 49 | 148 |
| | *Quantity Transferred per Sale ($m^3$/day)* | | |
| Government | 112.7 | 58.7 | 86.2 |
| Industrial | 467.3 | 260.1 | 397.7 |
| Private Party (Ag) | 90.9 | 65.7 | 82.5 |

**Table 3. Antofagasta Region Current and Planned Mining Desalination Projects and Projected Water Costs**

| Mine Name / Project | Pumping distance (km) | Elevation (m) | Water cost estimate (per $m^3$/day)[28] | Status[29] | Seawater Capacity ($m^3$/day) | Desal Capacity ($m^3$/day) |
|---|---|---|---|---|---|---|
| Michilla / Michilla | 15 | 835 | $4,265 | Operational | 6,500 | 2,300 |
| Municipal / La Chimba | - | - | $5,155 | Operational | - | 52,013 |
| Municipal / Desaladora Sur | - | - | $6,993 | Prequalification | - | 86,400 |
| Municipal / Taltal | - | - | $7,300 | Operational | - | 432 |
| Algorta Norte / Algorta | 65 | 1,300 | $7,300 | EIA Approved | 12,960 | - |
| Sierra Gorda / Sierra Gorda | 141 | 1,700 | $7,300 | EIA Approved | 128,736 | - |
| Esperanza / Michilla II | 145 | 2,200 | $8,881 | Operational | 62,208 | - |
| Escondida / El Coloso | 170 | 3,150 | $19,542 | Operational | - | 45,360 |
| Escondida / El Coloso II | 170 | 3,150 | $19,542 | Preferred Bidder | - | 276,480 |
| Total | | | | | 210,404 | 462,985 |

[28] 5% discount rate
[29] Source: GWI, 2012

**Table 4. Estimates of Statistical Models**

| VARIABLES | (1)<br>1995-2009 | (2)<br>1995-2009 | (3)<br>1995-2009 | (4)<br>1995-2009 | (5)<br>2001-2006 | (6)<br>2001-2006 | (7)<br>2001-2006 | (8)<br>2001-2006 |
|---|---|---|---|---|---|---|---|---|
| Log(Copper Price) | 0.191 | 0.141 | | | -1.503 | -1.486 | | |
| | (0.299) | (0.288) | | | (2.543) | (2.614) | | |
| Government | 0.676*** | | 0.513** | | -0.420 | | -0.422 | |
| | (0.209) | | (0.260) | | (0.281) | | (0.342) | |
| Industrial | 1.267*** | | 1.308*** | | -0.655 | | -0.655 | |
| | (0.246) | | (0.254) | | (0.556) | | (0.557) | |
| Govt./Indust. | | 0.992*** | | 1.019*** | | -0.483 | | -0.498 |
| | | (0.203) | | (0.227) | | (0.306) | | (0.346) |
| Post-2003 | -0.533 | -0.303 | | | -1.513* | -1.575* | | |
| | (0.463) | (0.431) | | | (0.866) | (0.877) | | |
| Government x Post-2003 | 0.264 | | 0.419 | | 1.507*** | | 1.509*** | |
| | (0.295) | | (0.330) | | (0.496) | | (0.534) | |
| Industrial x Post-2003 | -0.277 | | -0.275 | | | | | |
| | (0.303) | | (0.311) | | | | | |
| Govt./Indust. x Post-2003 | | -0.0275 | | -0.0577 | | 1.570*** | | 1.585*** |
| | | (0.278) | | (0.296) | | (0.510) | | (0.535) |
| Observations | 442 | 442 | 442 | 442 | 143 | 143 | 143 | 143 |
| R-squared | 0.324 | 0.311 | 0.343 | 0.325 | 0.187 | 0.185 | 0.187 | 0.185 |
| Year Control | Quadratic | Quadratic | Dummies | Dummies | Quadratic | Quadratic | Dummies | Dummies |

Robust standard errors in parentheses
Statistical significance: *** p<0.01, ** p<0.05, * p<0.1

## *Figures*

**Figure 1. Allocation with declining marginal benefits (left panel); allocation with constant marginal benefits (right panel)**

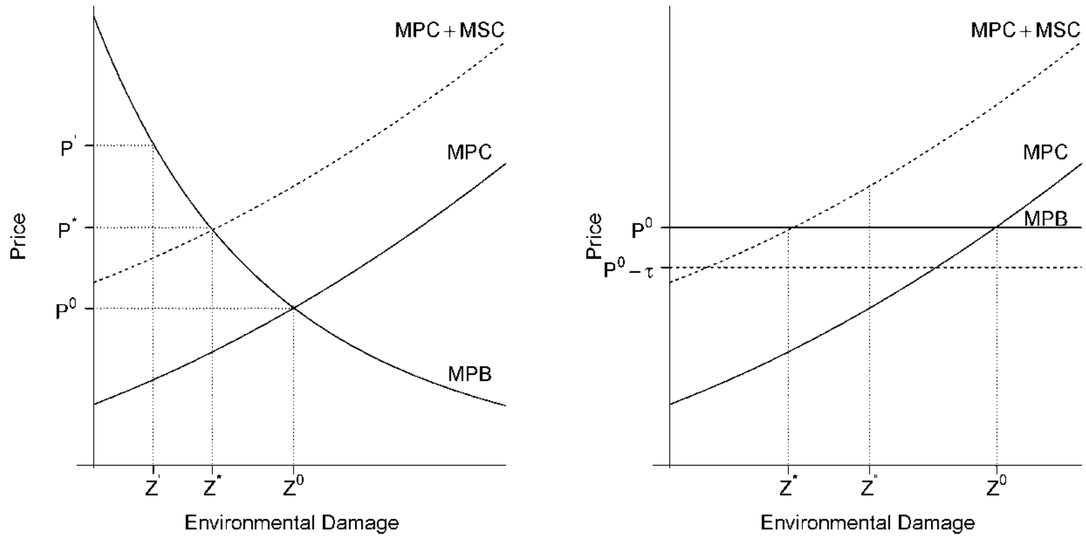



**Figure 2. Cumulative Water Right Change Requests and Denials (1990-2010)**

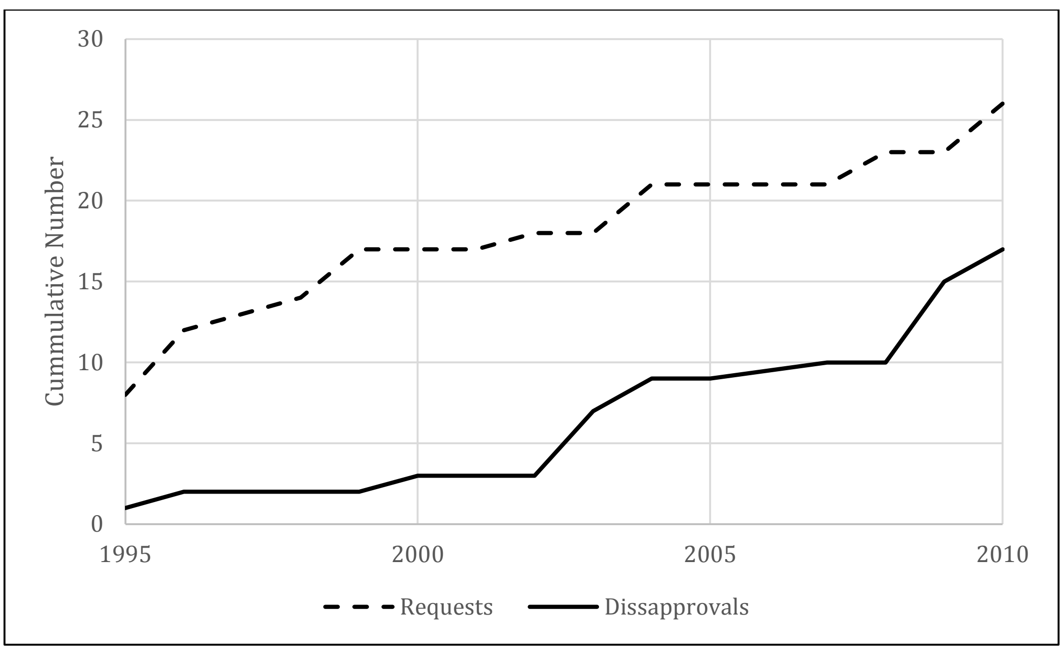


**Figure 3. Percentage of Total Requests Denied (1995-2010)**

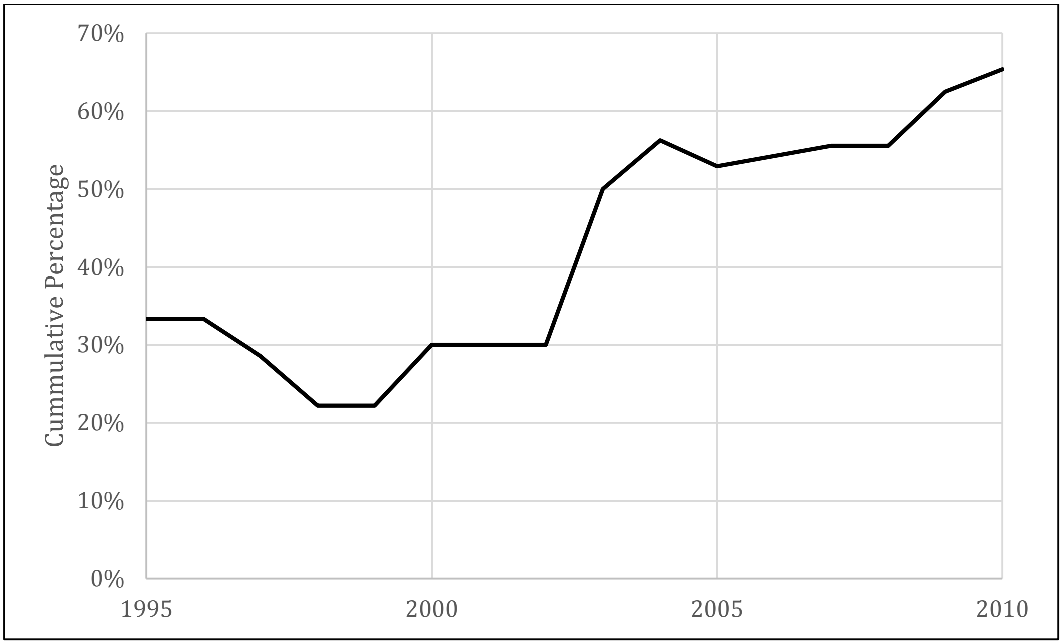



**Figure 4: Copper Production by Ownership in Antofagasta Region**

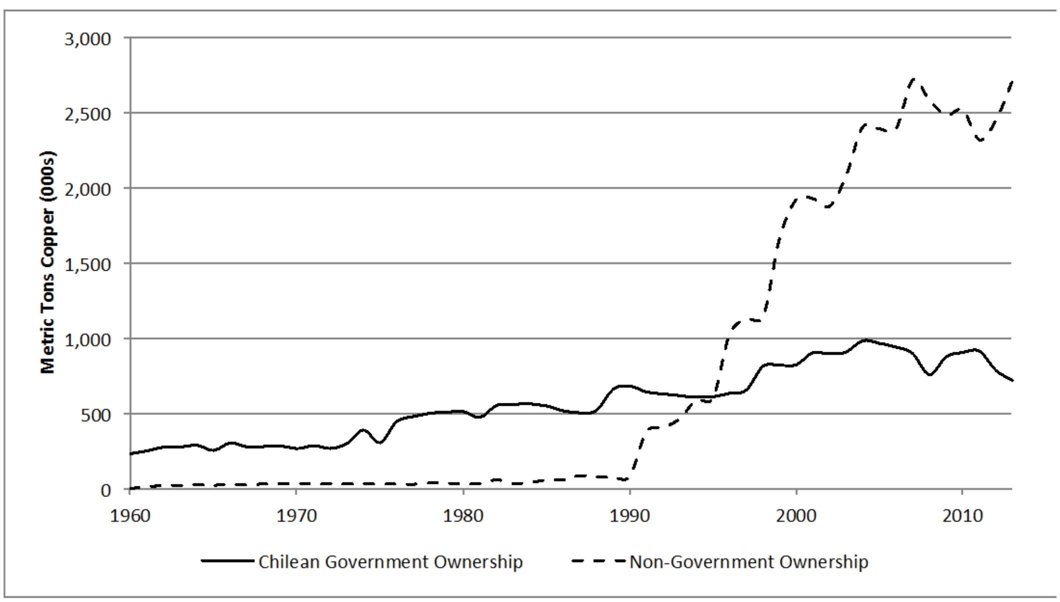


**Figure 5. Map of Groundwater Sources, Mine Locations, and Ecosystems in Northern Chile's Antofagasta Region**

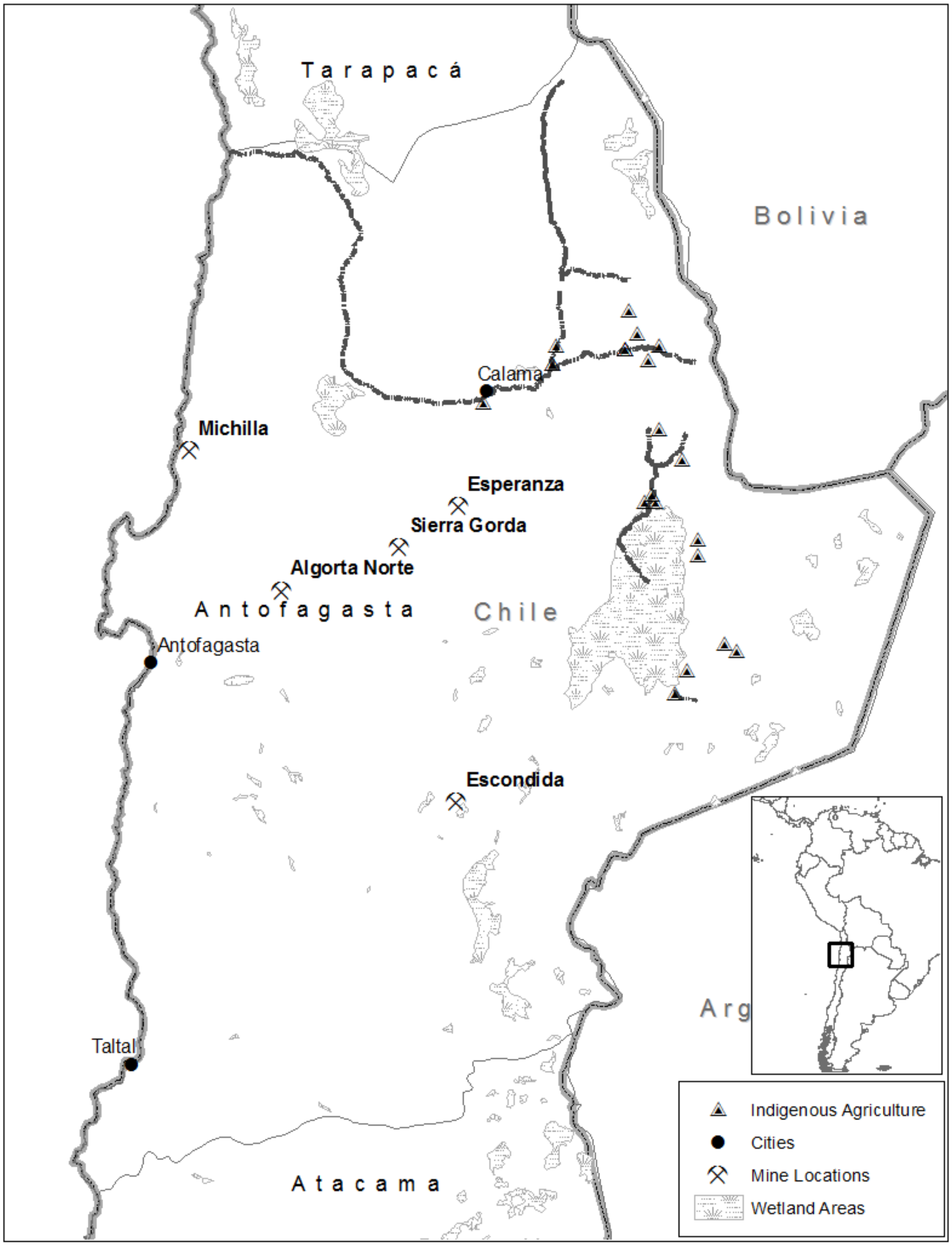


**Figure 6. Surface Water Demand and Welfare**

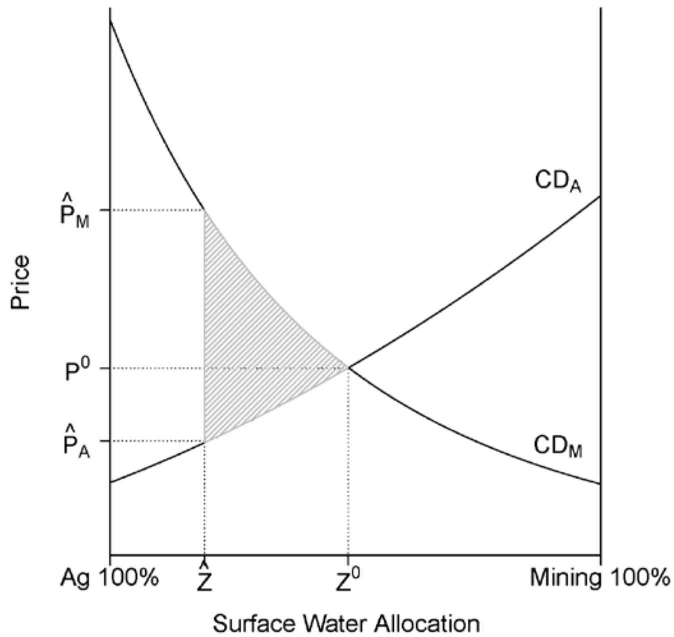



**Figure 7. Weighted Average Sale Price by Buyer Type**[30]

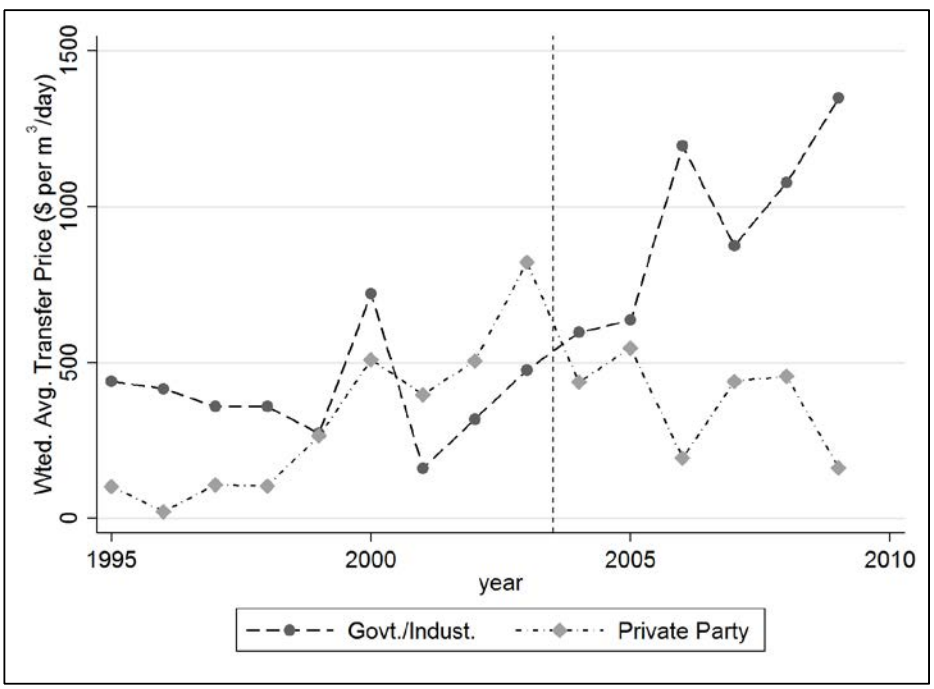



**Figure 8. Placebo Test of Six-Year Windows for Coefficient on Non-Government with 95% Confidence Intervals**

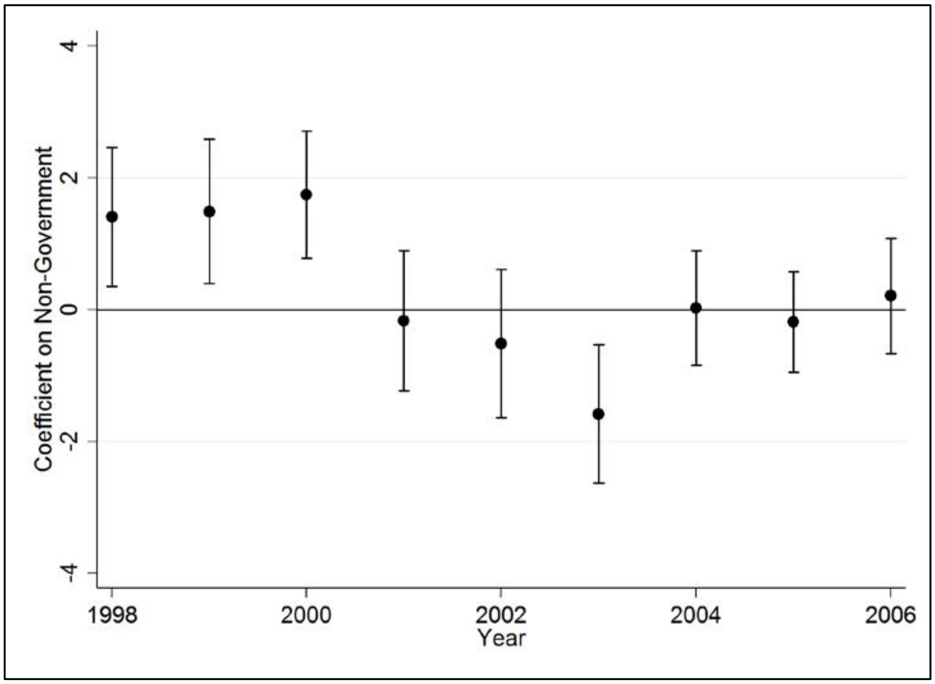
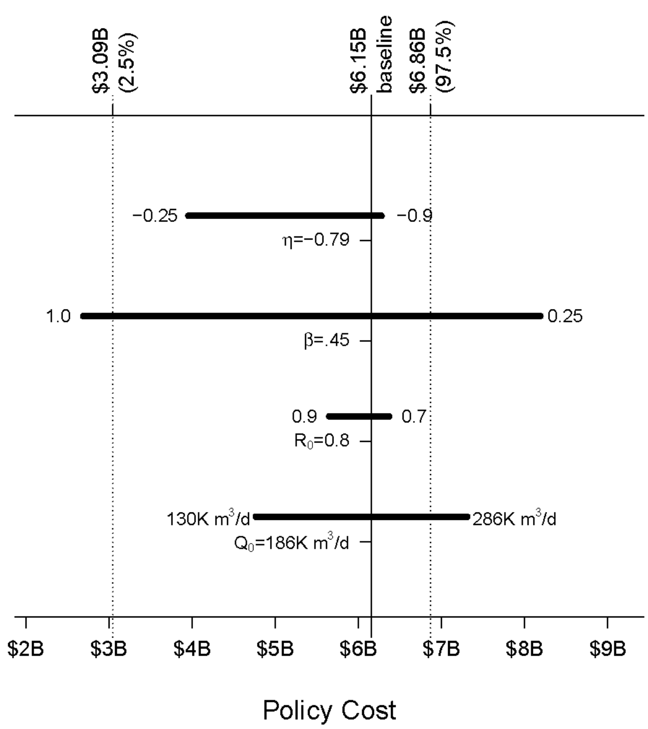



**Figure 9. Cost Sensitivity Analysis**

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30 From the dataset the figure drops a single outlier sale for over \$4000 per $m^3$/ day to an industrial buyer from 1998. This is done to better display the data; the sale is not dropped from the statistical analysis.