Groundwater Conservation via Desalination: Regulator Behavior and Welfare Implications

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I. Introduction

Why does environmental regulation exist? Environmental quality, as a public good, is underprovided in many markets. Creating regulation that increases environmental good provision provides an opportunity for government to enhance society’s welfare. This is the traditional explanation of regulation via welfare economics—that it emerges in response to market failure and is efficiency enhancing (Pigou, 1952). Because environmental goods are underprovided in the market, the need for regulation and the inefficiency of command-and-control versus market-based regulation has been the emphasis of research in the regulatory arena. What has been absent from many discussions is the considerable economics literature on government failure. The seemingly high cost of regulation has been documented in a few cases including the prohibition on drilling for oil in the Alaska National Wildlife Refuge (Kotchen and Burger, 2007), the Endangered Species Act (Brown and Shogren, 1998), and pesticide regulation at the Environmental Protection Agency (Cropper et al, 1992). When market failure is identified, the presupposed remedy is often government intervention.

In environmental decision making settings, like those mentioned above, a high degree of technical expertise is required to assess the problem and balance the costs and benefits of potential remedies. Regulation can delegate technical rule making to agencies, which develop expertise in highly specialized areas. When an agency develops expertise, the information and decisions generated are not easily accessible by the public or policymakers, creating the potential for asymmetric information. A consistent pattern of scientific agency decisions is that the environmental good is provided at a high cost,
seemingly beyond what economists would consider the socially optimal quantity. And this provision may occur at a high cost to a narrow economic interest. This outcome is counterintuitive because the literature on agency behavior and capture predicts, and examples from regulated industries like utilities corroborates, that narrow economic interests influence agencies, allowing the interest to capture rent at the expense of social well-being (Laffont and Tirole, 1991).

The contribution of this paper is to explore the apparent disconnect between the theory of capture and observed outcomes in environmental regulations. We explore the case of water trading regulation in northern Chile’s Antofagasta Region. Chile has strong property rights to water, but its property rights are not complete in protecting environmental services (Hearne 1998; Grafton et al., 2011). Because of this, market failure potentially exists in the form of excessive trading of water away from low-intensity uses that generate unpriced public benefits. Regulatory intervention in the water market in the Antofagasta Region occurred with the implementation of environmental laws and rules after environmental degradation due to intensification of water use by mining firms. The effect of the policy has been to severely limit transfers of water to mining companies, locking in the current pattern of water use. As a result of this intervention in the market, the use of additional freshwater in mining has been greatly restricted and mining firms have responded by switching to desalinated water. The conservation of the groundwater resource in this manner is costly in terms of reduction in mining firm profits, and thus government tax revenue. This outcome also has a high, albeit different, environmental cost due to the intensity with which desalination uses fossil fuels.
This paper first provides a general model of regulatory behavior consistent with a regulator that provides a high provision of environmental quality and why such an outcome is supported by policymakers in the presence of a concentrated economic interest. We then turn to the illustrative case of Chile’s Antofagasta Region illustrate the model and document the private cost of conserving the freshwater resource. We do not attempt to determine the efficiency of this decision, which requires technical expertise on the nature of the environmental good provided that we do not possess. Instead, we provide an estimate of the magnitude of the value of the environmental benefits necessary for the current policy to break even under benefit-cost analysis.

II. Sketch of Regulatory Model

The view of a regulator as wholly concerned with maximizing total social welfare, e.g. the “noble bureaucrat,” is too simplistic (Schroeder 2009). Regulator (as opposed to policymaker) decisions are: (1) affected by interest groups (Ando 1999); (2) constrained by the principle-agent relationship between regulators and policymakers (Prendergast 2007); and (3) made to avoid criticism (Leaver, 2009). A clear example of regulator failure is the case of regulatory capture where a regulatory agency is directly or indirectly incentivized to provide rules favorable to the regulated party. The capture literature has largely focused on regulated utilities and political influence (Laffont and Tirole, 1991; Dal Bó, 2006). Concentrated economic interests, like firms, will exert pressure for favorable policies (Stigler 1971; Peltzman 1976; Posner 1974). Competition among interest groups may even lead to more efficient government provision of goods (Becker 1958; 1976), but this outcome will change if information is distorted (Johnson and Libecap 2001).
In the case of an environmental regulator tasked with intervening in an incomplete market, we observe provision of environmental goods even when there exists the potential for capture. A concentrated economic interest is often harmed by environmental decision and the benefits may be broadly dispersed. We have colloquially started referring to such outcomes as “anti-capture,” and believe information distortion is the an explanatory factor. In a capture model, an economic interest gains favors from a regulator because the regulator’s actions are not directly observable by policymakers. Clearly, environmental regulators have the same control over information. Our model attempts to explain why these agencies have an incentive to provide an environmental service without a concentrated interest incentivizing their decision and why policymakers are content to allow this behavior.

In our model of intervention by a water regulator, water is presently used in low-value but environmentally beneficial purpose by interest 1. Interest 2 is a high-value user interested in purchasing the water and placing it in a less environmentally beneficial purpose. Markets are potentially inefficient in presence of these environmental benefits.

Terminology:

- \( x \) – Amount of water in use by interest 2 after transaction
- \( x_H \) – Initial amount of water
- \( x_{H-x} \) – Amount of water transferred to interest 2
- \( b_E(x) \) – Environmental benefit of water use
- \( b_1(x) \) – Private benefit of interest 1
- \( b_2(x_H-x) \) – Private benefit of interest 2
- \( c_2(x_H-x) \) – Private cost of moving water from interest 1 to interest 2
- \( p \) – Transaction price of transferred water

Interest 1 maximizes profit:

\[
\Pi_1 = b_1(x) + p \cdot (x_H - x)
\]

FOC:
\[ b'_1 = p \]

Likewise, interest 2 maximizes profit:

\[ \Pi_2 = b_2(x_H - x) - c_2(x_H - x) - p \cdot (x_H - x) \]

FOC:

\[ b'_2 - c'_2 = p \]

Market outcome is \( x \) such that:

\[ p = b'_2 - c'_2 = b'_1 \]

The socially optimal outcome is found by maximizing the social benefit function \( N \):

\[ N = b_E(x) + b_1(x) + b_2(x_H - x) - c_2(x_H - x) \]

FOC:

\[ b'_E + b'_1 = b'_2 - c'_2 \]

The market outcome is efficient only when, \( b'_E = 0 \), there is no change in environmental benefits from decreasing the amount of water. Otherwise, there is market failure.

In the presence of environmental benefits, the government can attempt to intervene to prevent market failure by creating an agency that finds information about the value of the resource and sets a quota. Assume the environmental benefits are linear in the amount of water transferred \( b_E(x) = b_E \cdot x \) and can take two values. With probability \( p \), \( b_E = \overline{b_E} \) and with probability \( 1-p \), \( b_E = \underline{b_E} \). Without information, the social benefit function is:

\[ N = p \cdot \overline{b_E} \cdot x + (1 - p) \cdot \underline{b_E} \cdot x + b_1(x) + b_2(x_H - x) - c_2(x_H - x) \]

FOC:

\[ p \cdot \overline{b_E} + (1 - p) \cdot \underline{b_E} + b'_1 = b'_2 - c'_2 \]
Agency observes signal $\sigma$, with probability $\zeta$; $\sigma=b_E$ when the agency observes the correct signal, $\sigma=\phi$ when the agency does not observe a signal. There are four states: (1) $\sigma=b_E$ with probability $p \cdot \zeta$; (2) $\sigma=b_E$ with probability $(1 - p) \cdot \zeta$; (3) $\sigma=\phi$ when the true state is $b_E$, with probability $(1 - p) \cdot (1 - \zeta)$; and (4) $\sigma=\phi$ when the true state is $b_E$, with probability $(1 - p) \cdot (1 - \zeta)$. *Demonstrate how by paying the agency to provide information can improve from the outcome without information.*

If interest 2 is able to spend money to capture the agency, the agency may be induced to report $\sigma=b_E$ at times when this signal is not received. *Demonstrate how this leads to inefficient outcome.*

This can be countered by creating a “scientific” agency, which is not subject to capture by interest 2. This is done be creating an agency that places a high value on the resource. This agency correctly reports $\sigma$ when $\sigma=\{b_E, \bar{b}_E\}$. However, this agency also follows the precautionary principle, reporting $\sigma=b_E$ when they receive signal $\sigma=\phi$. *Derive the conditions for the social benefit of this type of agency to be higher than that of the agency that can be captured.*

This simplified model illustrates why a rational government creates an agency which may overprovide environmental quality, in order to avoid capture, and why it is difficult for external parties to question the agency’s findings, only infrequently, with probability $(1 - p) \cdot (1 - \zeta)$, does the agency report higher environmental benefits than actually exist.

**III. Chile Case: Background**
A. Motivation

For the remainder of this paper, we document a case of water regulation in Chile, a country the Heritage Foundation’s *Index of Economic Freedom* rated second in the world in security of property rights. Many economists have argued that strong property rights lead to more efficient water allocation under a broad range of circumstances (Howe et. al. 1986, Easter et. al. 1999), and Chile is touted as the world’s leading case for strong water property rights and treating water as a marketable good (Bauer, 1998; 2004). However, Chile’s growing income has led to increased demand for environmental amenities (Hearne, 1998), rights to which are not well defined under the private water right system (Grafton et al., 2011).

Chile’s Antofagasta Region is located in the arid northern part of the country, often labeled the driest area in the world. Here, increasing water demand from the copper mining industry has coincided with increased calls for the environmental goods and services water provides. Studying water allocation in this region offers a chance to study regulator intervention for the express purpose of environmental regulation, in an otherwise well-defined water rights market. The extreme scarcity of water in region also makes it an early data point in what could be increasing conflict over water allocation as demand for economic and environmental supplies increases.

B. The Water Resource

The Antofagasta Region region slopes up from the Pacific Ocean, which defines Chile’s border in the west, to the Andes, which define its border with Bolivia and Argentina to the east. Precipitation falls in the higher areas (the eastern portion of the region and western Bolivia) and infiltrates into the groundwater system. At lower
elevations west of the Andes, the water table intersects topography and natural springs emerge, feeding rivers and wetlands. Figure 1 is a map of the region providing information on the location of groundwater sources, which along with surface water sources are primarily located at higher altitudes in the east. The mines, cities, and farms are located at lower elevations, allowing gravity-aided water conveyance at relatively low costs.

In addition to supporting ecosystems, water also sustains a population of around 580,000 with groundwater used primarily for mining and surface water used for local (non-export) and indigenous agriculture. Table 1 provides a current estimate of water use in the region by sector. While copper produces the majority of the region’s income, indigenous agriculture is an important cultural amenity for many Chileans not engaged in it, along with providing livelihood and sustenance for those who are.\(^3\) Farming undertaken by indigenous communities represents 96.7% of the acreage in cultivation in the Antofagasta Region and there are a reported 374,704 hectares classified as agricultural production (Chilean Agricultural Census 2007). The number of acres may overstate the importance of the region’s agriculture on a national accounts basis, where it only represents 0.05% of the Antofagasta Region’s Gross Regional Product (Central Bank of Chile 2008).

There are two distinct types of water property rights in Chile, surface and ground. Mining and industrial users primarily own groundwater rights while surface water users are primarily agriculturalists, although mining companies own some surface water as

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\(^3\) Sufficient water for drinking, sanitation, bathing, and food preparation is usually considered a basic human right, but water to grow food is in a different category, as food can be grown in wetter regions and imported (Gleick, 1996). Thus, it is useful to think of the preservation of regional agricultural in an economic framework as a decision that provides benefits as well as imposes costs.
well. In the Antofagasta Region, around 1/3 of the allocated rights are to surface water (Cristi et al 2013), although only a portion of the groundwater rights are actually in-use. Although the rights are allocated and traded separately, the physical system is not separate. Groundwater extraction is located high on the eastern edge of the region, often near or above points of natural discharge from the groundwater system. The flow of water to the riparian and wetland systems decreases as the water table is lowered through groundwater pumping. Thus, an increase in groundwater withdrawal that lowers the water table can lead to a decrease in surface water availability with both private and public costs due to reduced indigenous agriculture and environmental damages (Edwards and Kirk-Lawlor 2013).

C. Water Rights and Management

Water rights in Chile, once granted, are fully protected as private property under the Chilean Constitution (Constitución Política de la República de Chile, article 19(24)). Chilean water rights under the 1981 Water Code are completely separated from land ownership and can be purchased, sold, mortgaged, and transferred, like other real property (Mentor, 2001). Rights are defined for total diversion, not consumptive use. Thus, a right to 1 L/s allows any user to divert 1 L/s, regardless of whether the user will consumptively use all the water, or use it year-round. For instance, moving water to an off-site mine and using it year-round will likely increase the intensity of use relative to irrigating crops locally.

Chile is often held as the world’s leading case for strong water property rights and treating water as a marketable good, but the results of this in terms of actual marketability

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4 While this is a simplification of the real situation in the Antofagasta Region, it is mostly true: 27% of surface water versus virtually all groundwater rights are held by mining companies.
of rights and trading volume is unclear (Bauer, 1998; 2004). In assessing the security of Chile’s water rights, it is important to distinguish between the transfer of title to a water right and the actual transfer of its use. Title transfers are protected by the constitution and can be conducted freely without regulatory review. However, regulatory restrictions to the use of water right titles have emerged as a response to issues with the way the country’s water law deals with environmental impacts.

Efforts to conserve the Antofagasta Region’s water resources through environmental laws has led to much uncertainty and impacted water use approvals in the region (Cristi et al., 2013). Two types of review exist prior to use of a right: a regulatory review and an Environmental Impact Assessment (EIA) review. The regulatory review is triggered when a trade changes the location of water extraction. The EIA review (Chilean Law 19.300), which was passed in 1994 but has only recently become fully operational, is more far-reaching and applies to any water use in a new mining project. Although the water regulator undertaking the EIA review is the same agency that performs the regulatory review, different criteria and a different department evaluates the EIA.5 In Figure 2, top panel, we show the outcomes of regulatory reviews (not including EIA reviews) in which mining companies petitioned to use previously purchased groundwater right titles in the Antofagasta Region and in the bottom panel we show the cumulative rejection rate, comparing surface mining applications with those of other users. The

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5 For these projects, the EIA must document a source of water and the water regulator is asked to provide an opinion if (i) the right is not currently in use or (ii) the report concedes potential environmental harm of its use of the water. By definition, a change in location is treated as potential harm and requires a regulator response, and this type of transfer is reviewed whether EIA review takes place or not. For example, projects in the agricultural sector do not need an EIA, but will need to get authorization from the regulator if the project changes the extraction point. While focusing on the environment, the outcome of an EIA review is based on agency negotiations with the project developer and can include indigenous issue as well as environmental concerns.
decreasing frequency of approvals is indicative of the increasing difficulty mining firms face in using their water rights.\textsuperscript{6}

\textit{D. Response to New Demand}

The emergence of mining, especially copper mining, in the Antofagasta Region has put increasing pressure on the region’s limited water resource. Copper production increased 240\% from 1990 to 2010 and mining provides 97\% of the region’s exports. Chile is by far the largest copper producer in the world, and the region produces 54\% of the country’s copper. Copper is a key source of income for the Chilean government with per year copper revenues averaging 10\% of total government income from 1994-2006 (Fuentes 2007).

The socially optimal allocation of water across mining and agricultural sectors is explored by Gaudet et al. (2006) and termed the “Alberta Dilemma.” They reach the conclusion that for a period of time when the value of water in mining is high, a social planner would allocate all water to the mining sector. An efficient water market will do the same—as prices rise with increasing water scarcity, water will be traded out of low value sectors. The desalination alternative changes the social planning problem slightly, but qualitatively desalination will be undertaken when its cost is comparable to acquiring water in the market. This means most water would move from agriculture to mining prior to the adoption of desalination, because the marginal product of all but the highest marginal value uses of water is less than the cost of importing desalinated water.

Because the cost of pumping water to inland areas can be quite high, with energy costs associated with pumping up to four times as high as the actual desalination cost, \textsuperscript{6} The regulatory and environmental review processes, however, do not encompass all water trades. If the location does not change, the use by the new owner is not reviewable and if the water is certifiably in-use, it is likely there will not be cause to prevent approval.
economic theory predicts expansion of desalinated water in the Antofagasta Region to begin near the coast. An example is the coastal city of Antofagasta, which owns inland water rights, contracting water service to a subsidiary of a mining firm—Aguas Antofagasta. They built a desalination plant on the coast to supply the city and transferred the freshwater right to an inland mine. Although this example fits the pattern of an efficient water market, the move was heavily criticized in Chile under the premise that the mining firms have the means to pay to pump the desalinated water and should therefore not purchase the city’s freshwater rights.

The estimate of even the cheapest mining desalinated water project is substantially higher than the current price at which some freshwater rights trade.\(^7\) While the weighted mean price of a groundwater right purchased by mining firms averaged $474,291 per L/s, one of the largest copper mines in the world, Escondida, constructed a 525 L/s desalination plant at a variable pumping and desalination cost estimated to be near $1.7M per L/s. Where these types of price differences exist, regulatory restrictions to protect certain types of uses or particular groups of users may be responsible (Chong and Sunding 2006) and the price of groundwater right titles likely reflects the uncertainty of their use once purchased (Cristi et al 2013). Escondida invested in the desalination project after it was prohibited from exercising its holdings of water right titles, reported to be around 1,095 L/s (Lemus and Duclos, 2008).

In the regulated Antofagasta water market, trades between agricultural users are conducted at prices lower than those between mining firms. This price difference reflects the belief of agricultural users that their rights are not convertible into rights that can be

\(^7\) Direct seawater is also an option for mining firms, but the pumping costs remain the important economic variable.
sold to mining users. The weighted-average transfer price between agricultural users between 2005 and 2009 was $111,354 per L/s. This price reflects the current opportunity cost of water in the agricultural sector. The order of magnitude difference between this price and the cost of desalinated water motivates the remainder of this paper, which focuses on the theoretical considerations necessary to estimate the potential welfare effect.

IV. Model of the Water System

In this section we introduce a model of the physical water system and show how the regulator controls environmental provision.\(^8\)

A. Water System

The simple physical system we model here is represented in Figure 3. The depth to groundwater at elevation \( H \) from surface elevation \( S \) is \( S - H \) (this model is based on work by Edwards and Kirk-Lawlor, 2013). The combined outflow of water from the aquifer, this water feeding the riparian and wetland systems, is \( F(H) \). This flow is a function of aquifer elevation, \( H(K, w_m) \), which in turn is a function of recharge, \( K \), and groundwater withdrawals for mining, \( w_m \). An increase in groundwater withdrawals leads to a decrease in surface water availability:

\[
\frac{\partial F(\cdot)}{\partial w_m} < 0
\] (1)

\(^8\) This model makes two important assumptions. First, we assume that the entire system was near an equilibrium state prior to human water use. Second, because the system has a high flow to stock ratio, and because Chilean law prohibits groundwater mining, we assume the main issue is the allocation of the renewable portion of the water supply. Edwards and Kirk-Lawlor (2013) find the value of mined water from a sub basin in the region where the water table was depleted was less than 10% of value of increased renewable water available due to decreased evapotranspiration.
F(H) is approximated as a linear function of H. Likewise, holding K fixed and assuming a uniform cross-sectional aquifer area, H(K, w_m) is a linear function of w_m. Therefore:

$$F(w_m) \approx \alpha - \beta w_m$$

This leads us to an approximation of the effect of pumping on outflow, -\beta:

$$\frac{\partial F(\cdot)}{\partial w_m} = -\beta; 0 < \beta < 1$$

The hydrologic properties of the aquifer determine the change in outflow with increased pumping. If \(\beta=0\), there is no effect of pumping on surface water users, while if \(\beta=1\), a one-unit withdrawal of water leads to a one-unit decrease in water availability for surface users, as would occur in a flowing river where the extractor doesn’t return any flow. It is clear that when \(\beta>0\), the exercise of a groundwater right decreases the flow to surface water users, with two possible costs. First, it decreases the water available to agricultural users, reducing their ability to farm and creating a private cost. Second, it reduces public goods: riparian ecosystems, wetland ecosystems, and cultural values of indigenous farming.

We assume for simplicity that the agricultural sector uses surface water and the mining sector uses groundwater. The agriculture sector uses a portion of its water right consumptively, that is they return a portion of their allocation to the natural system, while the mining sector uses all its water consumptively.\(^9\) If rights are provided to the total flow of water (for the remainder of the paper we refer to this type of right as a diversion right), failing to account for the consumptive/non-consumptive distinction, a market equalizes marginal net private benefits of the two sectors: \(\text{MNB}_A = \text{MNB}_M\). This is not the socially

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\(^9\) Mining firms will purposefully evaporate wastewater within slag piles to lock away contaminants like heavy metals rather than release it back into the water system.
optimal allocation, however, because the agriculture sector uses only a portion of their water consumptively. Likewise, unless $\beta=1$, the use of groundwater decreases the availability of surface water by less than one unit. This is similar to the behavior of return flows in agriculture. Although the mining sector has no return flows, its use of one unit of groundwater only reduces water to surface users by $\beta$ units. To find the optimal allocation of water between sectors, we can write a maximization problem for allocation between $w_M$, the amount of water in mining, and $w_A$, the amount in agriculture in terms of net benefits of each sector:

$$\max_{w_A, w_M} NB_A(w_A) + NB_M(w_M)$$ \hspace{1cm} (4)

subject to $F(w_M) \leq w_A - R(w_A)$

Where $R(w_A)$ is the return flow as a function of agricultural extraction. The constraint is the surface water budget equation. This equation can be rewritten by substituting (2) for $F$ on the left hand side as:

$$a - \beta w_M \leq w_A - R(w_A)$$ \hspace{1cm} (5)

Now, we can easily turn this into a maximization problem in a single variable.

Assuming the constraint holds at equality, the first-order condition is:

$$\frac{MNB_A}{1 - dR/dw_A} = \frac{MNB_M}{\beta}$$ \hspace{1cm} (6)

To demonstrate this efficiency condition we suppose agriculture has a constant return flow so that $R(w_A) = R_0 w_A$ and plot two-sector water allocation via a three-axis plot, Figure 4, in a framework developed by Griffin (2006, p. 40). The total surface water availability is represented on the x-axis and demand for water is represented in terms of percentage of total surface water rights available. The demand curve for a diversion right to agricultural water, $D_A$, is shown in the normal fashion. The mining demand curve for a
diversion water right, $D_M$, has been flipped. Thus, movement from left to right along the
x-axis represents an increase in the percentage of total water use by the mining sector.

Because the two sectors use water at different intensities, optimal allocation
requires each sector to pay for the proportion of the diversion right that reduces the total
surface water available. For agricultural users, this is the amount of water not returned,
and for mining users this is the proportion that groundwater use reduces surface flows.
This optimal allocation in terms of surface diversion rights is depicted at point $A_0$ in
Figure 4. Where the condition of (6) is rewritten in terms of demand curves and a fixed
proportion of agricultural return flows:

$$\frac{D_A}{1 - R_0} = \frac{D_M}{\beta}$$  \hspace{1cm} (7)

This point occurs where the actual benefit of consumptive use of agricultural
water (the willingness to pay for a diversion right, $D_A$, divided by the consumptive use)
equals the actual benefit of reduction of surface water availability due to groundwater use
(the willingness of the mining sector to pay for a diversion right, $D_M$, divided by the
reduction to surface water availability caused by that use, $\beta$). To reach an optimal
allocation via trading of rights, a rate of exchange on trades between the sectors must be
established to account for the differing intensities of use. At the optimal allocation, the
price $P_A$ is paid for a diversion right to surface water by the agriculture sector and $P_M$ is
paid by the mining sector for a diversion right to groundwater. These prices induce the
sectors to purchase diversion rights at the socially efficient rate.

**B. Social Welfare Implications**

Figure 5 shows the allocation of consumptive surface water between the mining
and agricultural sectors as predicted when the regulator limits water allocation to the
mining sector. CD_A is the demand curve of the agricultural sector for consumptive use of water and likewise CD_M for the mining sector. The current allocation, enforced by restrictions on the use of water rights by the mining sector is located at A_R. This allocation is efficient if the MSB are such that the curve CD_A+MSB intersects CD_M at A_R, as shown in the figure. We are interested in finding the private welfare cost of this allocation, shown as the shaded region in this figure. CD_M is the willingness of the mining sector to pay for the ability to reduce surface water availability by one unit, CD_A is the willingness of the agricultural sector to pay for the consumptive use of one unit of surface water. The difference between these curves from allocation A_0 to A_R is the private cost due to the moratorium.

V. Estimation Procedure and Data

As water moves from the agriculture sector to the mining sector, the remaining agricultural water should become more valuable, and the marginal willingness of the agricultural sector to pay for water will increase. The cost of desalinated and direct seawater projects will be limited in their distance from and elevation above the sea, such that the cost of this water is approximately equal to the cost of obtaining freshwater rights. The willingness to pay and thus the demand for water in mining is limited by the availability of the desalinated and direct seawater backstop.

We do not directly observe prices \( \hat{\rho}_M \) and \( \hat{\rho}_A \) in the water right market, as these are prices in consumptive water use and we observe rights in terms of diversion right price. Instead, we observe \( \hat{\rho}_M' \) and \( \hat{\rho}_A' \), the diversion right prices in the market, then use estimates for \( R_0 \) and \( \beta \) we estimate these values:

\[
\hat{\rho}_M = \frac{\hat{\rho}_M'}{\beta}, \quad \hat{\rho}_A = \frac{\hat{\rho}_A'}{1 - R_0}
\]
We devote this section to our approach for estimating $CD_M$, $CD_A$, $\hat{P}_M$ and $\hat{P}_A$.

A. Desalination Cost Schedule

Desalinated water costs vary depending on the height above and distance from the ocean. Because firms are willing to use desalinated water, we assume they would also pay an equal amount for freshwater rights, and beyond this curve no freshwater would be purchased because it could be more cheaply obtained with desalinated water.\(^\text{10}\) Therefore, projections of adoption of desalinated water can provide an estimate of the willingness of mining firms to pay for freshwater rights. Through this method we can obtain an estimate of $D_M$.

To estimate the desalination curve, we project forward mining expansion plans and estimate water use requirements. Mining companies who plan to use desalinated or direct seawater must receive EIA approval well in advance of actually undertaking these projects. We use the plans provided by mining firms in these EIA reports and other public records to determine expected water use. While EIA reports are not commitments, and plans change, this is the most accurate information regarding the estimates of water quantity-price schedules of mining firms. Because the mine locations are fixed, knowing projected water use also provides a cost estimate based on elevation and distance from the ocean. We look only at water projects beyond the planning stages, which we define as those that are currently operational or have passed their EIA. We then assess the cost of desalinating (where used) and transporting water to these sites using estimates from the

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\(^{10}\) Were mining to become unprofitable at the backstop price, $D_M$ would be estimated using actual mining water demand.
literature. Capital costs of constructing pipeline facilities and pumping stations are not included.\footnote{11}

The desalination process itself has costs of around $1.00/m$^3$ of water with pumping costs of around 6¢ per 100km horizontally and 5¢ per 100m vertically (Zhou and Tol, 2005).\footnote{12} Because seawater is 1.025 times heavier than desalinated water, direct seawater projects require a slightly higher pumping cost, but no variable desalination cost. Our estimate for direct seawater projects is conservative because it does not include the cost of upgrading the on-site equipment and process changes necessary to use seawater.

We would like to compare the variable cost of desalinated and direct seawater to prices in freshwater transactions. Permanent water rights entitle the user to a given amount of water, measured in L/s, in perpetuity. In order to compare these prices with the price of desalinated water and direct seawater, we find the present value of a perpetual payment for desalinated water.\footnote{13} Brewer et al. (2007) discount the water flow itself in comparing one- and multi-year leases with permanent transactions. We reverse this and place the desalinated and seawater costs in terms of permanent prices. The cost estimates are provided in Table 2. This table indicates that almost 8000 L/s of desalinated and direct seawater is currently in use or planned in Chile’s Antofagasta Region, with the equivalent permanent price for a right to one L/s of water varying from $369K for direct

\footnote{11} 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.

\footnote{12} $0.63/m^3$ 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.

\footnote{13} 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.
seawater at the Michilla mine near the coast to over $1.69M at the Escondida mine further inland. Figure 1 shows the locations of Escondida, Michilla, and several other important mines in the region.

Water plays an essential role in all phases of the copper production process after the ore is mined and crushed. Yet water is also a relatively cheap input. At the end of 2011, a metric ton of copper was selling for $7,559. At this price, we estimate that the furthest and highest mine using desalinated water, Escondida, would pay a maximum of 2% of total revenue for desalinated water. When lower copper prices do occur, for instance at the end of 2002 when a metric ton of copper was $1,989 in 2011 dollars, the estimated cost of using desalinated water would be around 7.5% of revenue. Because water, even expensive desalinated water, is a relatively small cost of input, we assume for this estimation that the availability (or lack thereof) of freshwater does not change mining company investment decisions significantly. If this assumption does not hold, the cost estimate will be understated.

B. Agriculture Water Demand

Our estimation procedure for agricultural demand relies on the price of agriculture water in the surface water market. Based on this price, the willingness to pay for the consumptive use of water, the price at point $\hat{P}_A$ in Figure 4 is the price of water on the agricultural market divided by the consumptive use proportion, $1-R_0$. Conversations with many officials at the water regulatory agency in Chile indicate that it is typically assumed

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14 Water use is given in Brantes and Olivares (2008) as an average of the water used for raw material processing. For sulphide ores it is 0.79 m$^3$/ton while for oxide ores it is 0.13 m$^3$/ton. Ore grades range from approximately 0.5% to 2% in the region, meaning a range of 39.5-158 m$^3$ of water per ton of pure copper for sulphides and 6.5-26 m$^3$ of water per ton of pure copper for oxides. We estimate the variable cost of delivery of desalinated water to the Escondida mine at $2.35. Escondida is estimated to have 1.24% sulphide ore grade, meaning 63.7 m$^3$ of water per ton of pure copper.
that $R_0=0.80$ for agricultural water use. This low rate of consumptive use occurs because agricultural water rights are not continuously used. However, river monitoring data and centralized data on actual agricultural water use are limited or nonexistent. We therefore provide sensitivity analysis along with this point estimate.

To complete our estimate of private welfare loss, we follow Lichtenberg and Zilberman (1986) in using previous estimates of agricultural price elasticity and an assumption of a constant elasticity of substitution (CES) demand function. This allows us to estimate the willingness to pay for agricultural water when water availability decreases. The CES demand function has the form:

$$Q = A_0 p^\eta$$

Where $A_0$ is a constant, $Q$ is quantity of agricultural production, $p$ is price of water, and $\eta$ elasticity of water demand. To our knowledge, there are not quantitative estimates of elasticity of water demand specific to farmers in the Antofagasta Region or for farmers practicing subsistence agriculture generally. In place of these estimates, we use an estimate from US irrigated agriculture of -0.79 from California farmers using surface water (Schoengold et al., 2006). For sensitivity analysis we use another estimate of -0.25 for Texas farmers using groundwater (Nieswiadomy, 1988). Compared to the US farmers studied in these papers, we expect Antofagasta farmers to have lower margins and less adaptability. Unfortunately for elasticity estimates, these observations tend to move the estimate in opposite directions. US farmers can more easily change irrigation practices and crop type than farmers practicing traditional methods of farming, which would lead to lower elasticity of demand for water of farmers in Antofagasta Region. But, water may be priced at low levels due to subsidy in the US, which means that increases in
price may reduce rent earned by farmers but not actually change the water use decision, particularly for small changes in price (Moore et al., 1994). This fact would lead us to believe US water demand is less elastic than that in the Antofagasta Region.

We use water trade price data from the Antofagasta Region to find the current price of agricultural water, $\hat{P}_A$. We analyzed 467 transactions recorded by real estate certifiers in the Antofagasta Region.\(^{15}\) We categorized buyers of water rights into three categories: agricultural buyers, municipal and government buyers\(^{16}\), and mining and industrial buyers.\(^{17}\) (Additional discussion of these prices and trading data are provided in Cristi et al., 2013.) After log transforming the data, we find the mean price paid for agricultural surface water, which is weighted by the quantity sold, to be $17,500 per L/s.\(^{18}\) The average agricultural water transaction price is used as a point estimate to find the CES demand curve for agricultural water. We also use a one standard deviation confidence interval to check robustness.\(^{19}\)

The constant $A_0$ can be estimated using the price, the elasticity, and the total water use in agriculture. Because we are interested in estimating the demand for consumptive water, we use the willingness to pay for a consumptive right, which is the observed price divided by the consumptive use portion $1-R_0$. Total current water use in the agriculture and livestock sector in the Antofagasta Region is estimated to be around 3307 L/s (Ayala, 15)

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\(^{15}\) Data is from Catastro Público de Aguas, DGA. One transaction is dated 1988, but all others are between 2000 and 2009. Of 467 transactions there are 314 transactions with price data, 247 of which, including 23 groundwater transactions, were arms length deals we did not eliminate.

\(^{16}\) These transactions are primarily groups or communities of agriculturalists acquiring water rights. There is a program where the Chilean government purchases water rights for indigenous communities in perpetuity, and these transactions are included as well.

\(^{17}\) Pairwise t-tests on the log of per-unit prices of the three categories are significant at 99%. While the distributions of this transformation of price for the population appear normally distributed, we also find the results significant using a two-sample Wilcoxon rank sum test at a 99% confidence interval.

\(^{18}\) Data is log transformed to fit a normal distribution, which is useful for confidence interval generation.

\(^{19}\) Calculated by finding the unbiased variance of the weighted mean calculation.
2007). The intensity of use in agriculture determines how much of this water is used consumptively. For example, if agriculture returns 50% of its flows to other users, we calibrate the CES demand function for consumptive water using a price of $35,000 and a quantity of 1,653.5 L/s

V. Estimate of the Welfare Cost and Discussion

To determine the amount of desalinated water offset by transfers from agriculture to mining, we must estimate $\beta$, the proportion groundwater pumping reduces surface flows. As $\beta$ decreases, groundwater extraction has less impact on surface water availability. Thus the private welfare loss is decreasing in $\beta$ because a lower value means more water is available for use in mining. We use calculations for the Ojos de San Pedro sub-basin to estimate $\beta=0.45$. In this region, estimated groundwater pumping of 1,551 L/s reduced surface water flow by 700 L/s (Edwards and Kirk-Lawlor 2013). In this region, $\beta<1$ because lowering the water table reduces the amount of evapotranspiration. Other areas with more or less water near the surface may have different $\beta$s and for this reason we provide a sensitivity analysis.

Using the baseline estimates of $\eta=-0.79$, $R_0=0.8$, and $\beta=0.45$ gives the estimate of the private welfare loss of trade restrictions at $2.055$ billion. Table 3 provides a sensitivity analysis for the three key variables, using the confidence interval developed from the water trade data. These estimates indicate how estimates change sharply with differing assumptions about the key parameters. Because these estimates are based on the price of permanent right transfers, these are total present value estimates of the private welfare costs of the trade restrictions.
For perspective, the range of estimates of private welfare loss can be compared to yearly Chilean government spending for the entire country on the Ministry of the Environment ($61.5M) and Indigenous Development ($163.4M). Studies of willingness to pay (WTP) for environmental public goods in other developing countries have focused on public goods which directly affect those surveyed, for instance a study in the Philippines for WTP for water quality improvements found average mean and median values around $1 per month (Choe et al., 1996). A survey of residents of an Indian city found WTP to protect a nearby national park used for recreation and water quality protection to have a mean of about $0.38 (RS 21) per month (Hadker et al., 1997). These estimates are comparable to more local public goods, which in the Chilean case could perhaps be valuable for the entire Antofagasta Region with a population of around 580,000. For this population, the estimate of the cost of protecting the cultural and environmental amenities is around $15 per month (assuming a 5% discount rate).

Another perspective is to look at the cost of the policy is the equivalent of each Chilean paying $119.

The implication of the paper thus far is that there are large welfare costs from preventing additional utilization of freshwater resources by mining firms. These costs would be quite costly to Chilean society at-large, as an average of 30% of mining firm earnings are paid in taxes. Additionally, almost the entire cost of desalination is carbon-based electricity charges from desalinating and pumping water. The planned direct and seawater projects lead to an additional 2.64 million metric tons of CO₂ released into the atmosphere.²⁰ According to IPCC estimates this amounts to an additional cost of

²⁰ Total project energy projections imply around 8.87B kWh of electricity based on $0.10/kWh electricity price. The Northern grid is 33% coal, 57% natural gas, remainder fuel oil and diesel (http://www.cdec-
$113.4M per year in social costs.\textsuperscript{21} We do not include the cost of carbon in our estimate of private costs, and note that only a portion of the currently planned or operational desalination projects would be switched off if even all freshwater were transferred to mining.

\textbf{VI. Conclusion}

Regulator behavior in the provision of environmental services warrants further examination. Our model suggests that regulatory anti-capture, where an agency provides too much environmental good or service without being incentivized by a narrow economic interest occurs due to an asymmetric information problem that is likely to occur in environmental agency. We demonstrate a potential case from Chile where regulations seem to impose costs on society in excess of their benefits.

\textsuperscript{21} The social cost of carbon $43.00 (http://www.ipcc.ch/publications_and_data/ar4/wg2/en/ch18s18-4-2.html) \(\Rightarrow\) $116.6M/yr
VII. References


Tables

Table 1: Water Use in Antofagasta Region

<table>
<thead>
<tr>
<th>Sector</th>
<th>Water Use (L/s)</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture &amp; Livestock</td>
<td>3,308</td>
<td>25.5%</td>
</tr>
<tr>
<td>Mining/Industrial</td>
<td>6,761</td>
<td>52%</td>
</tr>
<tr>
<td>Energy</td>
<td>1,493</td>
<td>11.5%</td>
</tr>
<tr>
<td>Other (incl. urban)</td>
<td>1,432</td>
<td>11%</td>
</tr>
<tr>
<td>Total Consumption</td>
<td>12,994</td>
<td></td>
</tr>
</tbody>
</table>

Table 2: Antofagasta Region Current and Planned Mining Desalination Projects and Projected Water Costs

<table>
<thead>
<tr>
<th>Mine Name / Project</th>
<th>Pumping distance (km)</th>
<th>Elevation (m)</th>
<th>Status²⁴</th>
<th>Seawater Capacity (L/s)</th>
<th>Desal Capacity (L/s)</th>
<th>Water cost estimate (per L/s)²⁵</th>
</tr>
</thead>
<tbody>
<tr>
<td>Michilla / Michilla</td>
<td>15</td>
<td>835</td>
<td>Operational</td>
<td>75</td>
<td>27</td>
<td>368,509</td>
</tr>
<tr>
<td>Municipal / La Chibia</td>
<td>-</td>
<td>-</td>
<td>Operational</td>
<td>-</td>
<td>602</td>
<td>630,720</td>
</tr>
<tr>
<td>Municipal / Desaladora Sur</td>
<td>-</td>
<td>-</td>
<td>Prequalification</td>
<td>-</td>
<td>1,000</td>
<td>630,720</td>
</tr>
<tr>
<td>Municipal / Taltal</td>
<td>-</td>
<td>-</td>
<td>Operational</td>
<td>-</td>
<td>5</td>
<td>630,720</td>
</tr>
<tr>
<td>Algorta Norte / Algorta</td>
<td>65</td>
<td>1,300</td>
<td>EIA Approved</td>
<td>150</td>
<td>-</td>
<td>445,430</td>
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<tr>
<td>Sierra Gorda / Sierra Gorda</td>
<td>141</td>
<td>1,700</td>
<td>EIA Approved</td>
<td>1,490</td>
<td>-</td>
<td>604,208</td>
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<tr>
<td>Esperanza / Michilla II</td>
<td>145</td>
<td>2,200</td>
<td>Operational</td>
<td>720</td>
<td>-</td>
<td>767,283</td>
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<tr>
<td>Escondida / El Coloso</td>
<td>170</td>
<td>3,150</td>
<td>Operational</td>
<td>-</td>
<td>525</td>
<td>1,688,437</td>
</tr>
<tr>
<td>Escondida / El Coloso II</td>
<td>170</td>
<td>3,150</td>
<td>Preferred Bidder</td>
<td>-</td>
<td>3,200</td>
<td>1,688,437</td>
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<tr>
<td>Total</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2,435</td>
<td>5,358</td>
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</tbody>
</table>

²² *1 L/s is equivalent to 256 acre-feet per year; Sources: Dirección General de Aguas (2007; 2008)
²³ The water supply in the Antofagasta Region is extremely limited. The Colorado River Compact, which divides the flow of the Colorado between seven arid US states, by comparison, allocates 586 m³/s.
²⁴ Source: GWI, 2012
²⁵ 5% discount rate
Table 3: Sensitivity Analysis for Private Welfare Cost Estimate ($ millions)

<table>
<thead>
<tr>
<th>Sensitivity Type</th>
<th>Parameter A Values</th>
<th>Parameter B Values</th>
<th>CI (1sd)</th>
<th>Point Estimate</th>
<th>CI (1sd)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline and Ag Price Sensitivity:</td>
<td>η=-0.79</td>
<td>R₀=0.8</td>
<td>β=0.45</td>
<td>1,210</td>
<td>2,055</td>
</tr>
<tr>
<td>Elasticity Sensitivity:</td>
<td>η=-1.33</td>
<td>η=-0.79</td>
<td>η=-0.25</td>
<td>2,288</td>
<td>2,055</td>
</tr>
<tr>
<td>Return Flow Sensitivity:</td>
<td>R₀=0.7</td>
<td>R₀=0.8</td>
<td>R₀=0.9</td>
<td>3,238</td>
<td>2,055</td>
</tr>
<tr>
<td>Groundwater Intensity Sensitivity:</td>
<td>β=0.25</td>
<td>β=0.45</td>
<td>β=0.65</td>
<td>3,848</td>
<td>2,055</td>
</tr>
</tbody>
</table>
Figures

Figure 1: Map of Groundwater Sources, Mine Locations, and Ecosystems in Northern Chile’s Antofagasta Region
Figure 2: Surface Water Right Changes and Approvals in Antofagasta; top – mining groundwater applications and approvals; bottom - surface right application rejection rate over time.

Change Application Rejection Rate

- Non-Mining Surface Water
- Mining Surface Water
Figure 3: Basic Ground/Surface Water System
Figure 4: Two-Sector Diversion Right Demand

Figure 5: Welfare and Regulator Decisions