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ABSTRACT

Urban water conservation is typically achieved through prescriptive regulations, including the rationing of water for particular uses and requirements for the installation of particular technologies. A significant shift has occurred in pollution control regulations toward market-based policies in recent decades. We offer an analysis of the relative merits of market-based and prescriptive approaches to water conservation, where prices have rarely been used to allocate scarce supplies. The analysis emphasizes the emerging theoretical and empirical evidence that using prices to manage water demand is more cost-effective than implementing non-price conservation programs, similar to results for pollution control in earlier decades. Price-based approaches also have advantages in terms of monitoring and enforcement. In terms of predictability and equity, neither policy instrument has an inherent advantage over the other. As in any policy context, political considerations are important.

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1. Introduction

Cities, towns, and villages around the world struggle to manage water resources in the face of population increases, consumer demand for water-intensive services, and increasing costs (including environmental costs) of developing new supplies. In this paper, we provide an economic perspective on reducing urban water demand through pricing and non-price conservation policies. We compare price and non-price approaches to water conservation along five dimensions: the ability of policies to achieve water conservation goals, cost-effectiveness, distributional equity, monitoring and enforcement, and political feasibility.

The worst drought on record continues to unfold in the American southeast, affecting major cities such as Atlanta, Georgia, and Raleigh, North Carolina. In the arid Western U.S., the Colorado River system faces the worst drought on record, lasting (thus far) from 1999 to 2008 and leaving Lake Mead (the source of more than 90% of Las Vegas's water) about half empty.

Municipal water consumption comprises only about 12% of total freshwater withdrawals in the United States; and agricultural irrigation, the single largest water use, comprises just over one-third of all withdrawals (Hutson *et al.* 2004). While analysis suggests that re-allocating water from agriculture to cities would be efficient in many regions, in the current legal and political setting, large-scale transfers of water rights from agriculture to cities are relatively uncommon (Brewer *et al.* 2007, Brown 2006, Howe 1997). Thus, cities often must reduce water consumption during acute shortages due to drought, or in the long run due to constraints on their ability to increase supply.

26 The efficient water price is the long-run marginal cost (LRMC) of its supply. LRMC
27 reflects the full economic cost of water supply – the cost of transmission, treatment and
28 distribution; some portion of the capital cost of reservoirs and treatment systems, both those in
29 existence and those future facilities necessitated by current patterns of use; and the opportunity
30 cost in both use and non-use value of water for other potential purposes. Many analysts have
31 noted that water prices, for many urban as well as agricultural uses, lie well below LRMC (Sibly
32 2006, Timmins 2003, Hanemann 1997), with significant welfare consequences (Renzetti 1992,
33 Russell and Shin 1996). In the short run, without price increases acting as a signal, water
34 consumption proceeds during periods of scarcity at a faster-than efficient pace. Water
35 conservation takes place only under “moral suasion or direct regulation” (Howe 1997). In
36 contrast, if water prices rose as reservoir levels fell during periods of limited rainfall, consumers
37 would respond by using less water, reducing or eliminating uses according to households’
38 particular preferences. In the long run, inefficient prices alter land-use patterns, industrial
39 location decisions, and other important factors. The sum of all these individual decisions affects
40 the sustainability of local and regional water resources.

41 Implementation of efficient water prices would be challenging, to say the least. Some of
42 the opportunity costs of urban water supply are exceedingly difficult to quantify. What is the
43 value of a gallon of water left instream to support endangered species habitat, for example?
44 While economists have developed a variety of useful methods for estimating such values, the
45 expectation that every water supplier will develop full individual measures of the LRMC of
46 water supply is unrealistic. If LRMC represents an ultimate water pricing goal, there are smaller,
47 less ambitious steps toward efficiency that can be accomplished more readily.

48 Even with inefficient prices, injecting better price signals into the processes of water use
49 and allocation can result in important improvements. For example, given a particular public
50 goal, such as the conservation of a particular quantity of water or percentage of current
51 consumption, various policies can be employed, some more costly than others. Choosing the
52 least costly method of achieving some water-provision goal is characterized in economic terms
53 as cost-effective water management. Even if the water conservation goal is, itself, inefficient,
54 society can benefit from the minimization of costs to achieve it.

55 We focus on this issue of policy instrument choice for water conservation, summarizing
56 research from the economics literature, including both our own work on this issue and that of
57 other economists. Given the strong theoretical cost advantages of market-based approaches to
58 water conservation over conventional alternatives, and the emerging empirical evidence for the
59 potential cost savings from moving to market-based approaches to conservation, the time is ripe
60 for a discussion of the relative strengths and weaknesses of these policy instruments.

61

62 **2. Cost-effectiveness of water conservation policies**

63 Decades of theoretical and empirical economic analysis suggest that market-based
64 environmental policies are more cost-effective than conventional policies, often characterized as
65 command-and-control (CAC) or prescriptive approaches. Market-based regulations encourage
66 behavior through market signals rather than through explicit directives regarding conservation
67 levels or methods. These policy instruments, if well-designed and implemented, encourage firms
68 and households to undertake conservation efforts that are in their own interests and collectively
69 meet policy goals. CAC approaches, in contrast, allow little flexibility in the means of achieving

70 goals and often require households and/or firms to undertake similar shares of a conservation
71 burden regardless of cost.

72 In the area of pollution control, the cost-effectiveness advantage of market-based
73 approaches over CAC policies has been demonstrated theoretically (Pigou 1920, Crocker 1966,
74 Dales 1968, Montgomery 1972, Baumol and Oates 1988) and empirically (Keohane 2007,
75 Teitenberg 2006). Perhaps the best-known application of these principles to environmental
76 regulation is the U.S. SO₂ trading program, established under Title IV of the Clean Air Act
77 Amendments of 1990, which has produced cost savings on the order of \$1 billion annually
78 (Stavins 2003). Dozens of other market-based policies have been applied to air and water
79 pollution control, fisheries management, and other environmental problems in industrialized and
80 developing countries (Kolstad and Freeman 2007, Stavins 2003, Sterner 2003, Panayotou 1998).

81 Economists' attention has only recently turned to examining the potential economic gains
82 from adopting market-based approaches to water conservation, rather than CAC approaches.
83 Whereas the gains from market-based approaches to pollution control depend critically on
84 heterogeneity in marginal abatement costs across firms (Newell and Stavins 2003), the cost
85 savings from market-based approaches to water conservation derive largely from heterogeneity
86 in households' marginal benefits from water consumption (Mansur and Olmstead 2007). This is
87 because current CAC approaches to water conservation are essentially rationing policies. This
88 makes the application similar to other cases in which rationing has been replaced with price-
89 based allocation, such as traffic congestion on roadways (Parry and Bento 2002) and at airports
90 (Pels and Verhof 2004). Recent studies demonstrate how raising prices, rather than
91 implementing non-price policies, can substantially reduce the economic cost of achieving water
92 consumption reductions (Collinge 1994; Krause *et al.* 2003; Brennan *et al.* 2007).

93 In order to illustrate the basic economics, we can examine one typical CAC approach to
94 water conservation – a citywide required demand reduction achieved by uniformly restricting
95 outdoor uses. Figure 1 portrays two households with the same indoor demand curves, but
96 different preferences with respect to outdoor demand. The difference in slopes of the three
97 demand curves is associated with differences in elasticity – the percentage change in demand
98 prompted by a one percent price increase. (For all but one specific class of demand function,
99 price elasticity varies along the demand curve, thus while we can speak broadly about
100 comparisons across demand curves, there may be points on a relatively steep demand curve at
101 which price elasticity exceeds that on some parts of a flat demand curve.) Here we assume that
102 indoor demand (frame C in Figure 1), the steepest curve, is inelastic, because indoor uses are less
103 easily reduced in response to price changes, reflecting the basic needs met by indoor water use.
104 For outdoor demand, there is a relatively elastic household (Panel A), and a somewhat less
105 elastic household (Panel B). The more elastic household is more likely to reduce outdoor demand
106 in response to a price increase – perhaps because it has weaker preferences for outdoor
107 consumption (e.g., in the short run, it would rather allow the lawn to turn brown than pay a
108 higher water bill to keep it green).

109 Unregulated, with price set at \bar{P} , both households consume Q_C units of water indoors, the
110 less elastic household consumes Q_B^{unreg} outdoors, and the more elastic household consumes Q_A^{unreg}
111 outdoors. Outdoor reduction mandated under a CAC approach (which leaves indoor use
112 unchanged, and reduces outdoor uses to Q_B^{reg} and Q_A^{reg}) creates a “shadow price” for outdoor
113 consumption (λ) that is higher under the current marginal price (\bar{P}) for household B than for
114 household A, because household B is willing to pay more for an additional unit of water than
115 household A. If instead the water supplier charges price P^* , that achieves the same aggregate

116 level of water conservation as the prescriptive approach, consumers would realize all potential
117 gains from substitution within and across households, erasing the shaded deadweight loss (DWL)
118 triangles. Consumption moves to Q_C^* indoors for both types of households, and to Q_A^* and Q_B^*
119 outdoors. The savings from the market-based approach are driven by two factors: (1) the ability
120 of households facing higher prices rather than quantity restrictions to decide which uses to
121 reduce according to their own preferences; and (2) allowing heterogeneous responses to the
122 regulation across households, resulting in substitution of scarce water from those households
123 who value it less, to those who value it more.

124 How large are the losses from non-price demand management approaches when
125 examined empirically? We know of only three cases in which the welfare losses from
126 prescriptive water conservation policies have been estimated. Timmins (2003) compared a
127 mandatory low-flow appliance regulation with a modest water price increase, using aggregate
128 consumption data from 13 groundwater-dependent California cities. Under all but the least
129 realistic of assumptions, he found prices to be more cost-effective than technology standards in
130 reducing groundwater aquifer lift-height in the long run.

131 Another study of 11 urban areas in the United States and Canada compared residential
132 outdoor watering restrictions with drought pricing (Mansur and Olmstead 2007). For the same
133 level of aggregate demand reduction as implied by a two-day-per-week outdoor watering
134 restriction, the establishment of a market-clearing drought price in these cities would result in
135 welfare gains of approximately \$81 per household per summer drought. This represents about
136 one-quarter of the average household's total annual water bill in the study.

137 Using a different approach, Brennan *et al.* (2007) constructed a household production
138 model to estimate the welfare cost of urban outdoor water restrictions in Perth, Australia, and

139 arrived at similar conclusions. The household welfare costs of a two-day-per-week sprinkling
140 restriction are just under \$100 per household per season, and the costs of a complete outdoor
141 watering ban range from \$347-\$870 per household per season.

142 Based on both economic theory and empirical estimates, the inescapable conclusion is
143 that using price increases to reduce demand, allowing households, industrial facilities, and other
144 consumers to adjust their end-uses of water is more cost-effective than implementing non-price
145 demand management programs.

146

147 **3. Predictability in Achieving Water Conservation Goals**

148 **3.1 *Effects of Price on Water Demand***

149 If policymakers are to use prices to manage demand, the key variable of interest is the
150 price elasticity of water demand. Because an increase in the price of water leads consumers to
151 demand less of it, all else equal, price elasticity is a negative number. (Elasticity figures may
152 also be reported in absolute value, and the negative sign is then implicit. We use the more
153 conventional negative sign in this paper.) An important benchmark in elasticity is -1.0 ; this
154 figure divides demand curves into the categories of elastic and inelastic.

155 There is a critical distinction between “inelastic demand” and demand which is
156 “unresponsive to price”. If demand is truly unresponsive to price, price elasticity is equal to
157 zero, and the demand curve is a vertical line – the same quantity of water will be demanded at
158 any price. This may be true in theory for a subsistence quantity of drinking water, but it has not
159 been observed for water demand more broadly in fifty years of empirical economic analysis.

160 That said, water demand in the residential sector is sensitive to price, but demand is
161 inelastic at current prices. In a meta-analysis of 124 estimates generated between 1963 and

162 1993, accounting for the precision of estimates, Espey *et al.* (1997) obtained an average price
163 elasticity of -0.51 , a short-run median estimate of -0.38 , and a long-run median estimate of $-$
164 0.64 . Likewise, Dalhuisen *et al.* (2003) obtained a mean price elasticity of -0.41 in a meta-
165 analysis of almost 300 price elasticity studies, 1963-1998. And a recent, comprehensive study of
166 water demand in eleven urban areas in the United States and Canada found that the price
167 elasticity of water demand was approximately -0.33 (Olmstead *et al.* 2007). The price elasticity
168 of residential demand varies substantially across place and time, but on average, in the United
169 States, a 10% increase in the marginal price of water in the urban residential sector can be
170 expected to diminish demand by about three to four percent in the short run. This is similar to
171 empirical estimates of the price elasticity of residential energy demand (Bohi and Zimmerman
172 1984, Bernstein and Griffin 2005).

173 There are some important caveats worth mentioning. First, elasticities vary along a
174 demand curve, and any estimate represents an elasticity at a specific price, in particular, actual
175 (current) prices. Were prices to approach the efficient levels discussed earlier, water demand
176 would likely be much more sensitive to price increases.

177 Second, consumers are relatively more sensitive to water prices in the long run than they
178 are in the short run, because over longer time periods, capital investments are not fixed. For
179 example, households might change appliance stocks, retrofit water-using fixtures, or alter
180 landscaping from lawns to drought-tolerant plants; firms can be expected to change water-
181 consuming technologies, increase recycling, or relocate to areas in which water is more plentiful.
182 In the long run, a 10% price increase can be expected to decrease demand by about six percent.

183 Third, price elasticities vary with many other factors. In the residential sector, high-
184 income households tend to be much less sensitive to water price increases than low-income

185 households. Also, price elasticity may increase by 30 percent or more when price information is
186 posted on water bills (Gaudin 2006). And price elasticity may be higher under increasing-block
187 prices (in which the marginal volumetric water price increases with consumption) than under
188 uniform volumetric prices (Olmstead *et al.* 2007). Price elasticities must be interpreted in the
189 context in which they have been derived.

190 **3.2 *Effects of Non-price Conservation Programs on Water Demand***

191 Historically, water suppliers have relied on non-price conservation programs to induce
192 demand reductions during shortages. We consider the effects of such non-price programs in
193 three categories: (1) required or voluntary adoption of water-conserving technologies; (2)
194 mandatory water use restrictions; and (3) mixed non-price conservation programs.

195 **3.2.1 *Required or Voluntary Adoption of Water-Conserving Technologies***

196 Many urban water utilities have experimented with required or voluntary adoption of
197 low-flow technologies. (Since the 1992 Energy Policy Act, U.S. law has required the installation
198 of low-flow toilets and showerheads in all new residential construction, but some cities have also
199 mandated or encouraged retrofitting.) When water savings from these programs have been
200 estimated, they have often been smaller than expected, due to behavioral changes that partially
201 offset the benefit of greater technical efficiency. For example, households with low-flow
202 showerheads may take longer showers than they would without these fixtures (Mayer *et al.*
203 1998). The necessity of the “double flush” was a notorious difficulty with early models of low-
204 flow toilets. In a recent demonstration of similar compensating behavior, randomly-selected
205 households had their top-loading clotheswashers replaced with more water efficient, front-
206 loading washers. In this field trial, the average front-loading household increased clothes-

207 washing by 5.6 percent, perhaps due to the cost savings associated with the appliances' increased
208 efficiency (Davis 2006).

209 Several engineering studies have observed a small number of households in a single
210 region to estimate the water savings associated with low-flow fixtures. But most of these studies
211 used intrusive data collection mechanisms, attaching equipment to faucets and other fixtures in
212 homes (Brown and Caldwell 1984). Study participants were aware they were being monitored as
213 they used water, which may have led to confounding behavioral changes.

214 One comprehensive study that was not characterized by this monitoring problem
215 indicates that households fully constructed or retrofitted with low-flow toilets used about 20
216 percent less water than households with no low-flow toilets. The equivalent savings reported for
217 low-flow showerheads was 9 percent (Mayer *et al.* 1998). Careful studies of low-flow
218 showerhead retrofit programs in the East Bay Municipal Utility District, California, and Tampa,
219 Florida estimate water savings of 1.7 and 3.6 gallons per capita per day (gpcpd), respectively
220 (Aher *et al.* 1991; Anderson *et al.* 1993). In contrast, showerhead replacement had no
221 statistically significant effect in Boulder, Colorado (Aquacraft 1996). Savings reported for low-
222 flow toilet installation and rebate programs range from 6.1 gpcpd in Tampa, Florida to 10.6
223 gpcpd in Seattle, Washington (U.S. General Accounting Office 2000). Renwick and Green
224 (2000) estimate no significant effect of ultra low-flush toilet rebates in Santa Barbara, California.

225 It is not surprising that studies of the water savings induced by such policies vary widely,
226 from zero to significant water savings – the scope and nature of policies vary widely, as well.
227 More important than the raw water savings induced by these programs, however, is the cost per
228 gallon saved, in comparison with alternative policies. The costs of toilet retrofit policies
229 implemented in U.S. cities range from less than \$100,000 to replace 1,226 toilets in Phoenix,

230 Arizona to \$290 million for 1.3 million toilets in New York City (U.S. General Accounting
231 Office 2000). These can be expensive programs, but in most cases no analysis is done to
232 estimate the magnitude of price increases that would have induced demand reductions equivalent
233 to those observed with technology standards. Only with such information can price and non-
234 price demand management programs be compared as policy options on the basis of cost.

235 3.2.2 *Mandatory Water-Use Restrictions*

236 Non-price management tools also include utility implementation of mandatory water use
237 restrictions, much like the traditional command-and-control approach to pollution regulation.
238 These include restrictions on the total quantity of water that can be used, as well as restrictions
239 on particular water uses, usually outdoors, such as lawn-watering and car-washing. Empirical
240 evidence regarding the effects of these programs is mixed. Summer 1996 water consumption
241 restrictions in Corpus Christi, Texas, including prohibitions on landscape irrigation and car-
242 washing, did not prompt statistically significant water savings in the residential sector (Schultz *et*
243 *al.* 1997). However, a longer-term program in Pasadena, California resulted in aggregate water
244 savings (Kiefer *et al.* 1993), as did a program of mandatory water use restrictions in Santa
245 Barbara, California (Renwick and Green 2000).

246 3.2.3 *Mixed Non-Price Conservation Programs*

247 Water utilities often implement a variety of non-price conservation programs
248 simultaneously, making it difficult to determine the effects of individual policies. One analysis
249 of the effect of conservation programs on aggregate water district consumption in California
250 found small but significant reductions in total water use attributable to landscape education
251 programs and watering restrictions, but no effect due to non-landscape conservation education
252 programs, low-flow fixture distribution, or the presentation of drought and conservation

253 information on customer bills (Corral 1997). Another study of southern California cities found
254 that the number of conservation programs in place in a city had a small negative impact on total
255 residential water demand (Michelsen *et al.* 1998). An aggregate demand study in California
256 found that public information campaigns, retrofit subsidies, water rationing, and water use
257 restrictions had negative and statistically significant impacts on average monthly residential
258 water use, and the more stringent policies had stronger effects than voluntary policies and
259 education programs (Renwick and Green 2000).

260 *3.2.4. Summing up the predictability comparison*

261 Predictability of the effects of a water conservation policy may be of considerable
262 importance to water suppliers, although in most cases the objective of water conservation
263 policies is water savings, without any specific target in mind. In this case, an estimate of the
264 reduction expected from policy implementation is necessary, but precision is less important.

265 If certainty is required, economic theory would suggest that the quantity restrictions
266 typical of traditional, prescriptive approaches to water demand management would be preferred
267 to price increases, particularly if water suppliers could be sure of near-total compliance, or at
268 least be able to adjust their water savings target upward to account for a reliable estimate of the
269 noncompliance rate (Weitzman 1973). But suppliers generally cannot rely on substantial
270 compliance with quantity-based restrictions. In a comprehensive study of drought management
271 policies among 85 urban water utilities during a prolonged drought in Southern California,
272 analysts reported that 40 agencies adopted mandatory quantity restrictions, but also found that
273 more than half of the customers violated the restrictions (Dixon *et al.* 1996). Such non-binding
274 quantity constraints are common, but how are utilities to predict the water savings achievable
275 through quantity restrictions when less than half of consumers typically comply? In the same

276 study, about three-quarters of participating urban water agencies implemented type-of-use
277 restrictions (most of them mandatory). Few penalties were reported, and enforcement was weak,
278 again raising questions regarding compliance. With such low rates of compliance with
279 traditional quantity-based regulations, neither price nor non-price demand management programs
280 have an advantage in terms of predicting water demand reductions.

281

282 **4. Equity and Distributional Considerations**

283 The main distributional concern with a market-based approach to urban water
284 management arises from the central feature of a market – allocation of a scarce good by
285 willingness to pay (WTP). Under some conditions, WTP may be considered an unjust allocation
286 criterion. Think, for example, about the negative reaction to selling food and water to the highest
287 bidder in the aftermath of a natural disaster. This sense that there are some goods and services
288 that should not be distributed by markets in particular contexts is behind the practice of rationing
289 during wartime. A portion of water in residential consumption is used for basic needs, such as
290 drinking and bathing. “Lifeline” rates and other accommodations ensuring that water bills are
291 not unduly burdensome for low-income households are common. Thus, policymakers
292 considering market-based approaches to water management must be concerned about equity in
293 policy design.

294 What does the empirical evidence tell us about the equity implications of water pricing as
295 a conservation tool? Agthe and Billings (1987) found that low-income households exhibited a
296 larger demand response to price increases in Tucson, Arizona, but the study did not compare the
297 distributional effects of price and non-price approaches. Renwick and Archibald (1998) found
298 that low-income households in two Southern California communities were more price-responsive

299 than high-income households, reflecting water expenditures' larger share of the household
300 budget. Thus, if water demand management occurs solely through price increases, low-income
301 households will contribute a greater fraction of the cities' aggregate water savings than high-
302 income households. This is not surprising to economists – price elasticity tends to decline with
303 the fraction of household income spent on a particular good.

304 Importantly, the distributional implications of non-price policies vary by type. For
305 example, requiring particular landscape irrigation technologies results in demand reduction
306 mainly among higher-income households (Renwick and Archibald 1998).

307 Mansur and Olmstead (2007) examined the distributional impacts of various demand
308 management policies, and found that raising prices to reduce consumption would cause a greater
309 consumption reduction for low-income than for high-income households. (If we return to Figure
310 1 and assume that households of type A are low-income and type B are high-income, we can see
311 why this happens.)

312 The fact that price-based approaches are regressive in *water consumption* does not mean
313 they are necessarily regressive in *cost*. Likewise, the fact that non-price programs are progressive
314 in water consumption does not mean they are progressive in cost. The impact of non-price
315 programs on distributional equity depends largely on how a non-price program is financed. And
316 progressive price-based approaches to water demand management can be designed by returning
317 utility profits (from higher prices) in the form of rebates. In the case of residential water users,
318 this could occur through the utility billing process.

319 Drought pricing, like LRMC pricing, would cause utilities to earn substantial short-run
320 profits (Mansur and Olmstead 2007). These profits would have to be returned to consumers in
321 some form, as regulated utilities usually are required to earn zero or very low profits. Profits

322 could be re-allocated based upon income, in order to achieve equity goals. Any rebate scheme
323 that is not tied to current consumption can retain the strong economic-incentive benefits of
324 drought pricing, without imposing excessive burdens on low-income households, relative to
325 traditional approaches.

326

327 **5. Monitoring and Enforcement**

328 Price-based approaches to water demand management hold a substantial advantage over
329 non-price approaches in regard to administrative costs for monitoring and enforcement. Non-
330 price demand management policies require that water suppliers monitor and enforce restrictions
331 on particular fixtures, appliances, and other technologies that customers use indoors and out, the
332 particular days of the week or times of day that customers use water for specific purposes, and in
333 some cases, the quantity used for each purpose.

334 The great difficulty in monitoring and enforcing these types of command-and-control
335 approaches is one reason for the prevalence of outdoor watering restrictions – outdoor uses are
336 usually visible, and it is relatively easy to cruise residential streets searching for violators.
337 Overall, monitoring and enforcement problems explain the low rates of compliance with many
338 non-price demand management programs. Where low-flow fixtures are encouraged or required,
339 they are often replaced with their higher-flow alternatives if consumers are dissatisfied with
340 performance.

341 In contrast, non-compliance in the case of pricing requires that households consume
342 water “off meter,” since water consumption is metered and billed volumetrically in most U.S.
343 cities. Of course, higher prices generate incentives not only for conservation, but also for
344 avoidance. However, at prevailing prices and even with substantial price increases, the

345 monitoring and enforcement requirements of price changes are likely to be far less significant
346 than those of a comparable non-price approach.

347

348 **6. Political Considerations**

349 Water demand management through non-price techniques is the overwhelmingly
350 dominant paradigm in cities around the world. Raising prices, particularly for what people
351 perceive to be a “public service” (though water is supplied by both public and private entities),
352 can be politically difficult. After a two-year drought in the late 1970s, the city of Tucson,
353 Arizona was the first U.S. city to adopt marginal-cost water prices, which involved a substantial
354 price increase. One year later, the entire Tucson city council was voted out of office due to the
355 water rate increase (Hall 2000). Just as few elected officials relish the prospect of raising taxes,
356 few want to increase water rates.

357 Ironically, non-price programs are more expensive to society than water price increases,
358 once the real costs of programs and associated welfare losses are considered. A parallel can be
359 drawn in this case to market-based approaches to environmental pollution control, including
360 taxes and tradable permit systems. Cost-effectiveness has only recently been accepted as an
361 important criterion for the selection of policies to control pollution (Keohane *et al.* 1998).
362 Despite empirical evidence regarding their higher costs, political constituencies that prefer non-
363 price approaches have succeeded in preventing management through prices. Some of this
364 resistance to using prices may be due to misinformation, since most policymakers and water
365 customers are not aware of the cost-effectiveness advantage of the price-based approach. For
366 example, a common misconception in this regard is that price elasticity is “too low to make a
367 difference”.

368 Non-price demand management techniques can create political liabilities in the form of
369 water utility budget deficits. Non-price conservation programs are costly. In addition, if these
370 policies actually reduce demand, water utility revenues decline. During prolonged droughts,
371 these combined effects can result in the necessity for substantial price increases following
372 “successful” non-price conservation programs, simply to prevent water utilities from
373 unsustainable financial losses. This occurred in 1991 in southern California. During a prolonged
374 drought, Los Angeles water consumers responded to the Department of Water and Power’s
375 request for voluntary water use reductions. Total use and total revenues fell by more than 20
376 percent. As a result, the Department requested a rate increase to cover its growing losses (Hall
377 2000). In contrast, given urban price elasticities common in the United States, price increases
378 will increase water suppliers’ total revenues. The extra per-unit revenues from a price increase
379 outweighs lost revenue from the decreased demand.

380 The costs of inefficient water pricing and the relative cost advantages of price over non-
381 price water demand management programs are clear. But like other subsidies, low water prices
382 (on a day-to-day basis, as well as during periods of drought) are popular and politically difficult
383 to change. Some communities may be willing to continue to bear excessive costs from
384 inefficient water pricing, in exchange for the political popularity of low prices. In other cases,
385 rate-setting officials may be constrained by law, unable to increase water prices by a percentage
386 that exceeds some statutory maximum. In these cases, the tradeoffs involved should be
387 measured and made explicit to water customers.

388

389 **7. Conclusions**

390 Using prices to manage water demand is more cost-effective than implementing non-
391 price conservation programs. The gains from using prices as an incentive for conservation come
392 from allowing households to respond to increased water prices in the manner of their choice,
393 rather than by installing a particular technology or reducing particular uses, as prescribed by non-
394 price approaches. Price-based approaches also have important advantages in terms of monitoring
395 and enforcement.

396 In terms of predictability, neither policy instrument has an inherent advantage over the
397 other. Likewise, neither policy instrument has a natural advantage in terms of equity. Under
398 price-based approaches, low-income households are likely to contribute a greater share of a
399 city's aggregate water consumption reduction than they do under certain types of non-price
400 demand management policies. But progressive price-based approaches to water demand
401 management can be developed by returning some utility profits due to higher prices in the form
402 of consumer rebates. Such rebates will not significantly dampen the effects of price increases on
403 water demand, as long as rebates are not tied to current water consumption.

404 Raising water prices (like the elimination of any subsidy) is politically difficult, but there
405 may be political capital to be earned by elected officials who can demonstrate the cost-
406 effectiveness advantages of the price-based approach. At a minimum, communities choosing
407 politically popular low water prices over cost-effectiveness should quantify this tradeoff and
408 make it explicit. Where water rate-setting officials are constrained by law from raising water
409 prices, during droughts or in general, a discussion of the real costs of these constraints would be
410 useful.

411 We are reminded of the debate, beginning in the late 1980s, over market-based
412 approaches to pollution control. While some opponents of environmental taxes and tradable
413 permit systems still resist these approaches, policymakers have succeeded in implementing them
414 in many cases, achieving impressive pollution reductions at great cost savings over more
415 prescriptive approaches. A similar shift in the area of water conservation, where the principles
416 are essentially the same, is long overdue.

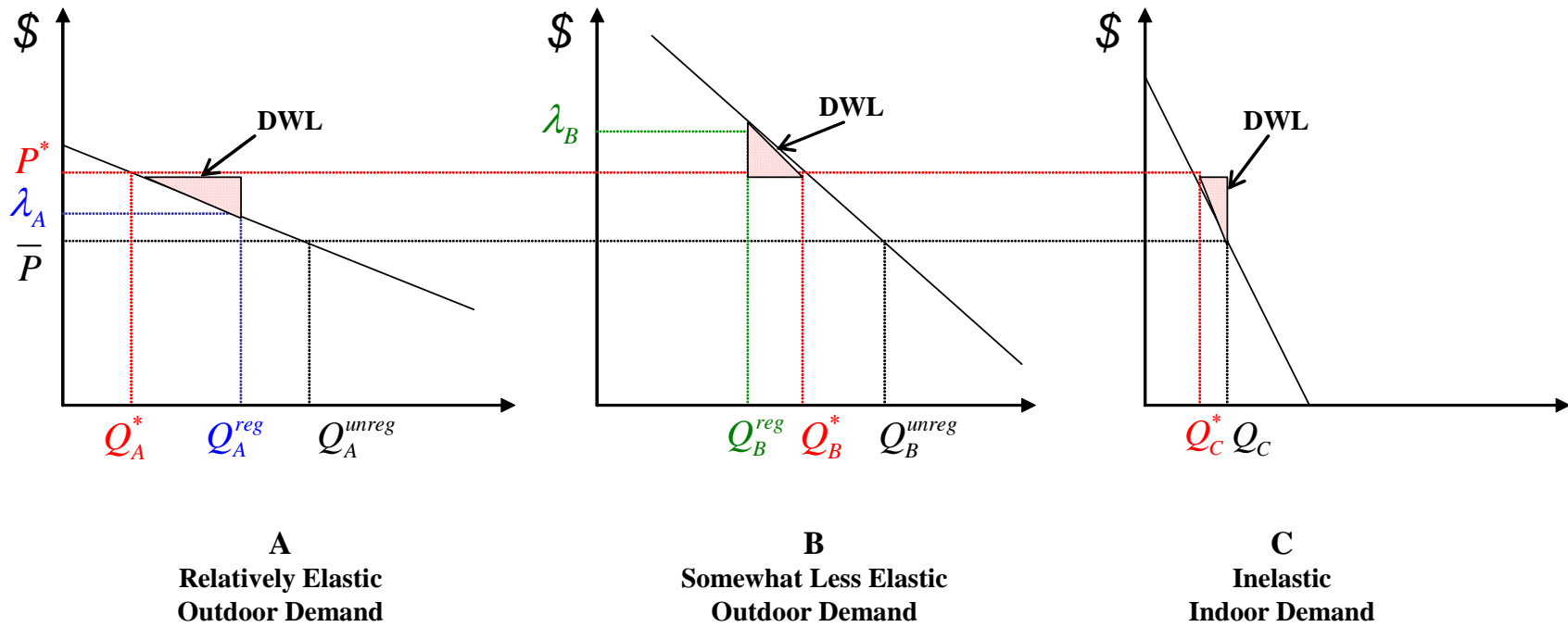
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421 Research.

Figure 1. Welfare Losses from Outdoor Consumption Restrictions with Heterogeneous Outdoor Demand



(Where P^* is the market-clearing price for $Q_A^{reg} + Q_B^{reg} + Q_C = Q_A^* + Q_B^* + Q_C^*$).

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