Climate change adaptation and water resource management: An analytical review of the current empirical economics literature

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Abstract

This paper considers the extent and usefulness of the existing empirical literature on water supply, demand, and adaptation to climate change for application to integrated assessment modeling efforts. The focus is on water resource impacts outside of the agricultural sector, and excluding changes in the distribution of extreme hydrological events – topics of other papers in the workshop. We review the existing literature on the likely economic impacts of climate change, acting through water supply and demand impacts in specific river basins, and the ability of adaptation to mitigate those impacts. Since adaptive responses will largely be implemented by local, regional, and national water management institutions, we also review what is known about the responses of water users to water prices, non-price water conservation policies, water trading, investment in and operations of storage and conveyance infrastructure, and transboundary water allocation mechanisms – the set of policy levers typically available to water managers at various geographic scales. Remaining gaps in the empirical economic literature on these topics are identified. The paper also describes the potential contributions of linking existing and new empirical research on water resource adaptation with IAMs. The importance of further empirical economic and political-economic research on the role of water management institutions in adaptation, or maladaptation, to climate change emerges as an important theme.

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1. INTRODUCTION

Climate change may affect both the long-term availability and the short-term variability of water resources in many regions. Potential regional impacts of climate change could include increased frequency and magnitude of droughts and floods, and long-term changes in mean renewable water supplies through changes in precipitation, temperature, humidity, wind intensity, duration of accumulated snowpack, nature and extent of vegetation, soil moisture, and runoff (Solomon et al. 2007). Behavioral changes associated with climate change, such as changes in demand for heating and cooling, will also impact water use. While annual global per capita runoff will probably increase in a warming climate, increases (mostly in East and Southeast Asia) are expected to occur mostly in high-flow seasons, increasing the need for water capture and storage as well as the risk of flooding (Bates et al. 2008). Changes in seasonal runoff regimes and inter-annual runoff variability may have greater economic impact than changes in long-term average runoff. Steven Chu, U.S. Secretary of Energy, has suggested that diminished freshwater supplies in some regions might be an even more serious global problem than rising sea levels as the climate changes (Gertner 2007).

Fisher-Vanden et al. (2011) list three main categories of adaptation that should be considered in modeling the economic impacts of climate change: (1) passive general market reactions (such as increases in heating and/or cooling); (2) specific reactive adaptation investments (such as disease treatment); and (3) specific proactive adaptation investments (such as seawall construction). The authors note that Integrated Assessment Models (IAMs) are already reasonably well-equipped to incorporate the first category of adaptation measures. Water, however, is not typically allocated through markets, prices are generally poor signals of resource scarcity and value in use, and many water quality and scarcity problems result from externalities, open access, and other market failures, some of them transboundary in nature (Olmstead 2010a, 2010b). Thus, even this first category of adaptation measures cannot easily be incorporated into IAMs without a thorough understanding of the institutions (most of them non-market) that determine water allocation, pricing, infrastructure investments, and other aspects of water management.

The importance of institutions to the magnitude, nature, and even the direction of adaptation to climate change implications for water resources – whether these changes are truly adaptive, or maladaptive – cannot be overstated. The ideal environment for successful, cost-effective adaptation is characterized by water management policies and institutions that are resilient and robust to uncertainty. Adaptive institutional responses could involve legal changes to water rights regimes, water pricing and price structure changes, implementation or expansion of water banking, leasing and marketing, negotiated ad-hoc water transfers, and changes in investment in and operation of water infrastructure including dams, reservoirs, conveyance infrastructure, and levees (Loomis et al. 2003). In developing countries in particular, changes in common property institutions that manage scarce water may also be important (Ostrom 1990). Maladaptive responses to climate change in the water sector could include local, regional, or national "grabs" for water from shared surface- and groundwater resources, as well as water pollution export to downstream jurisdictions. The thin available literature on these issues indicates that these market failures are currently of concern, though they can be mitigated by institutions. However, climate-related changes in hydrological regimes may exacerbate existing inefficiencies, challenging the ability of institutions to overcome market failures in water management. The key role of institutions, and the need for additional empirical work on how they evolve under conditions of water scarcity and increased hydrological variability, emerges as a major theme in almost every section of this paper.

The potential effects of climate change on water supply and quality will affect every sector of the economy, through impacts on health, agriculture, industry, transport, energy supply, non-market ecosystem services, fisheries, forestry, and recreation. Some of these sectors (agriculture, energy, and health) are addressed in other papers written for the workshop. In addition, some water resource impacts will occur through changes in the frequency and severity of extreme events in the water supply distribution (droughts and floods), which are, themselves, within the scope of other papers written for the workshop. Thus, the scope of this paper is, for the most part, limited to water-related adaptation outside of these sectors and extreme events covered in other papers (though I touch on them in some cases). The reader should keep in mind these limits on the paper's scope when interpreting the paper, since it is not a comprehensive review of what is known about water-related adaptation; the exclusion of agriculture, alone, is important, since irrigation accounts for almost 70 percent of global water withdrawals, and 90 percent of global consumptive use (Shiklomanov and Rodda 2003). Thus, some of the most important water-related adaptation issues may actually be raised by other papers in the workshop.

A final wrinkle in incorporating water resource management adaptation into IAMs is the fact that water tends to be managed at the local level or regional level. This introduces two complications. First, the downscaling of global and even national climate predictions to the local and regional level is unreliable, though downscaling methods are evolving. Thus, water resource adaptation measures will necessarily involve the creation and evolution of responsive, flexible management institutions (Haddad and Merritt 2001). Second, the information required to develop water management data inputs to IAMs seeking to model adaptation lies with these local water management institutions – very little such data is collected at the national level – posing challenges for thorough and consistent data collection. This is of great practical importance for implementing IAMs that incorporate water resource adaptation, but does not address how this problem can be overcome.

The paper proceeds as follows. Section 2 comprises most of the remaining text, reviewing the literature on several aspects of this issue. Section 3 summarizes the remaining gaps in the economic literature with respect to adaptation to the anticipated water resource impacts of climate change. Section 4 considers the potential contributions of connecting existing and new empirical research on water resource adaptation with IAMs, and brief conclusions are offered in Section 5.

2. CURRENT STATE OF THE LITERATURE

This section begins by examining what is currently known from existing empirical work focusing on the "big picture" – the likely economic impacts of climate change, acting through water resource impacts, and the ability of adaptation to mitigate those impacts, at a high level of spatial aggregation. Then, since adaptive responses will largely be implemented by local, regional, and national institutions, which (in the absence of markets) set prices, establish non-price water conservation policies, determine the nature and extent of allowable water trading, determine the level of investment in storage and conveyance infrastructure, as well as their operations, and negotiate transboundary water allocations, the section reviews what is known about the responses of water users to each of these policy levers, in turn.

2.1. U.S. national estimates of climate-related water resource impacts and adaptation

During the 1990s, several studies generated estimates of the economic impacts of climate-related changes in water resource availability on the scale of individual river basins, or for countries as a whole, often extrapolating and/or aggregating to arrive at estimates on a larger scale. For example, Fankhauser (1995) multiplied an average estimated climate-related runoff reduction for the United States (about 7 percent) by the average cost of water (about $0.42 per cubic meter), and suggested that the shift in runoff would impose a cost of about $13.7 billion on the U.S. economy. Modeling efforts such as this one have a naïve, "limits to growth" feel, in that they ignore the likely impacts of scarcity on prices, demand, and supply, and thus, welfare.

However, more recently, these early modeling efforts have been answered with economic models that must also considered naïve in their assumptions. An extensive economic modeling effort for four major basins in the United States (Hurd et al. 2004, Hurd and Harrod 2001, Hurd et al. 1999) assumes, for example, that water allocation is dynamically optimal, flowing to end-users in each period such that marginal water values are equal across users, maximizing the net benefit from water resources in each of the four basins over time. These studies have several advantages over the earlier models; they estimate welfare impacts by summing consumer and producer surplus, and they link infrastructure investment decisions with those regarding water allocation and consumption, as well as the geophysical features of individual basins. Nonetheless, their basic assumptions about how adaptation will take place contrast starkly with what we observe in real-world water management.

Consider current marginal water values and pricing in the arid U.S. West. Farmers in Arizona's Pima County pay $27 per acre-foot, and water customers in the nearby City of Tucson pay $479 to $3267 per acre-foot (Brewer et al. 2008). In Texas' Rio Grande Valley, the value of water in agriculture has been estimated at $300 to $2,300 per acre-foot, and in urban uses at $6,500 to $21,000 per acre-foot (Griffin and Boadu 1992). While these are just two examples, and these water prices and values are for different commodities (raw water vs. treated, piped water), the sharp differences in marginal water values across sectors are also products of inefficient pricing, historic water rights allocations, and subsidized irrigation projects (Wahl 1989). Current water prices do not equate marginal water values across users, in the United States or elsewhere, and it is unlikely that future water prices will do so.

Hurd et al. (1999, 2004) assume that water is fully tradable within watersheds in competitive markets with full information, no externalities, and no transaction costs. Yet, markets such as these would require well-defined property rights, and other legal and administrative institutions, to support trading. The large differences in marginal water values across users noted above give us some indication that water markets in the U.S. west, to the extent that they exist, are not terribly well-functioning, on average; this is confirmed by recent research (Libecap 2011), though the situation is improving in many states (Brewer et al. 2008). Another important barrier to the free flow of water to its highest-valued uses within a watershed are externalities. Third- water users diverting from a shared resource are robustly defended in existing rights regimes, depending on the seniority of their rights (Libecap 2011). In addition, the value for instream uses may be high in many places, and these values are increasingly posing barriers (efficient though they may be) to trade even in functioning water markets. The importance of instream flows to support ecosystem services, and their incorporation into water rights regimes, is likely to grow in importance as countries grow wealthier.

While the small available literature attempting to quantify the water-related impacts of and adaptation to climate change takes some important first steps, it either leaves out adaptation entirely, or assumes that water resources are traded in competitive watershed-level markets which will reach new, dynamically efficient equilibria in response to the new precipitation and temperature regimes induced by climate change. One option would be to integrate both of these classes of results into existing IAMs. It may be that the “no adaptation” models provide an upper bound on adaptation costs (though the possibility of maladaptation due to exacerbation of existing inefficiencies would suggest otherwise), and the “efficient adaptation” models provide a lower bound.

A better approach, however, would be to develop new models that describe how the institutions that manage water supply and demand will evolve over time, under different climate change scenarios, and then estimate or simulate the ways in which these changes will affect the behavior of end-users of water. Models like this would require, as inputs, information about how end users respond to changes in various water policies, such as price increases, or increased opportunities to lease or sell water to other users.

2.2. Estimates of responses to policies for managing water demand

2.2.1. Prices and water demand

In nearly all markets for goods and services, scarce resources are allocated through prices, which transmit information about relative scarcity and value in use. However, in the case of water, prices are administratively determined, through mechanisms that are often political and rarely take economic value into account. Water prices, therefore, do not respond automatically to short-term and long-term changes in supply.

Prices set by public officials are one potential lever for managing water demand when resources are scarce or highly variable. Good estimates of the price elasticity of water demand are critical to any such effort – water managers must understand how demand will respond to changes in price. Thus, much of the economics literature on water demand has focused on the econometric estimation of demand parameters, including price elasticity. Demand estimates can also be used to measure the value of water in both its diverted and instream uses. A substantial literature on the price elasticity of water demand has existed since the 1960s (see e.g., Howe and Lineweaver 1967), although this literature has been somewhat thin over the last decade.

Residential water demand. Residential water demand is inelastic at current prices. In a meta-analysis of 124 estimates generated between 1963 and 1993, Espey et al. (1997) obtained an average price elasticity of –0.51, a short-run median estimate of –0.38, and a long-run median estimate of –0.64. Likewise, in a meta-analysis of almost 300 price elasticity studies conducted between 1963 and 1998, Dalhuisen et al. (2003) obtained a mean price elasticity of -0.41. Perhaps surprisingly, a recent review of studies done in developing countries suggests that residential price elasticity is in the range of -0.3 to -0.6, similar to the range estimated for industrialized countries (Nauges and Whittington 2010). Studies have found that the residential price elasticity may increase when price information is posted on water bills (Gaudin 2006), and that it may be higher under increasing-block prices (IBPs) than under uniform volumetric prices (Olmstead et al. 2007).

In many developing countries, and a few industrialized countries, there are still significant portions of households receiving piped water whose use is unmetered. In these cases, metering would need to be introduced in order to use volumetric prices to manage demand. The introduction of metering, alone, in these settings may reduce water demand, even if initial prices are low. Significant water savings have been reported for U.S. communities switching from unmetered to metered consumption (OECD 1999). A U.S. federal government study suggests an average 20 percent reduction in total water use due to metering (Maddaus 1984).

Industrial water demand. Unlike residential demand, water demand for industry must be modeled as part of the general production process for the particular set of outputs generated with water and non-water inputs. This requires isolating the value of the marginal product of water. Industrial price elasticity estimates for water tend to be higher than residential estimates and vary by industry. The literature contains only a handful of industrial elasticity estimates. Griffin (2006) reports the results of five studies (published between 1969 and 1992), which have elasticity estimates ranging from -0.15 for some two-digit SIC codes (Renzetti 1992) to -0.98 for the chemical manufacturing industry (Ziegler and Bell 1984). A study of 51 French industrial facilities estimates an average demand elasticity of -0.29 for piped water, with a range of -0.10 to -0.79, depending on industry type (Reynaud 2003).

Water supply for power generation is by far the largest component of water use in the industrial sector. Thermal power plants withdraw raw water for cooling; such withdrawals comprised 49 percent of total water withdrawals in the United States in 2005 – but most of this is not consumed and returns to surface water sources (Kenny et al. 2009). There has been a good deal of concern expressed in the media and, to some extent, the peer-reviewed literature about the possible impact of climate-related water scarcity on power plants. For example, during particularly hot summers in 2003 and 2006, many European power plants had to reduce production due to water shortages and high water temperatures (Koch and Vögele 2009). For power plants and other industrial water users that withdraw raw water, instead of or in addition to piped water, water consumption would likely be price-responsive, as well, but pricing would require metering, which is often not done. One study suggests that, if a two-part charge (an annual permit fee of zero to $2,500, plus a volumetric rate of zero to ½ cent per cubic meter) were charged to firms withdrawing raw water in Ontario, non-hydroelectric water withdrawals would decline by one to eight percent, with resulting increases in firms' costs, as well as government revenues (Renzetti and Dupont 1999).

Agricultural water demand. Farmers who withdraw water directly from surface sources usually incur an energy cost to convey water for irrigation, but do not typically pay a volumetric charge for the water itself. Many agricultural water demand curves are estimated for groundwater, using energy costs for pumping to construct a water price variable. Prices can also be obtained if farms purchase water from irrigation districts or other water management institutions. While the economics literature contains many estimates of agricultural water demand elasticity, the available data are rarely of sufficient quality to estimate demand functions. Other techniques commonly applied for the agricultural sector include mathematical programming (Scheierling et al. 2006), field experiments, and hedonic methods (Colby 1989 and Young 2005). A recent meta-analysis of 24 U.S. agricultural water demand studies performed between 1963 and 2004 suggests a mean price elasticity of –0.48 (Scheierling et al. 2006), although estimates vary widely and, unlike in the industrial and residential sectors, often approach zero. Estimates were found to be higher for regions where water is scarce and prices are higher.

Even more important than the level of prices and price elasticity, if prices are to be used to manage water demand for agriculture, is the metering of water use by farmers. If a volumetric price is to be charged, water managers must know how much water is consumed by end-users. It is possible that institutions will adapt to reduced water supply, or even increased supply variability, by increasing the coverage of agricultural water metering – pressure to do this may also come from other sectors (e.g., cities) willing to pay more for water on the margin than are farmers. A recent example is California's Water Conservation Act of 2009, which requires that all large agricultural water suppliers (such as irrigation districts) measure water delivered to farms adopt some form of volumetric pricing. The measurement of groundwater extraction may be more difficult to do, but satellite data may increasingly provide reasonable estimates of net farm water use.[1]

Given the rich available literature, the responsiveness of households, industry, and agriculture to changes in the price of water are relatively well understood, though it would be useful to have additional estimates (particularly in developing countries) for industry and agriculture, where the literature is thinner. What is poorly understood, however, is where prices for water come from – the literature contains no rigorous model of the long run "supply curve" for municipal or agricultural water. There is some evidence that underlying heterogeneity in urban water utilities may explain the choice of price structure (Olmstead et al. 2007), and Hewitt (2000) provides empirical evidence that a municipal water utility's propensity to adopt ''market-mimicking'' water prices may have to do with administrative sophistication, system ownership (public or private), and financial health.

Will water price levels respond to shifts in hydrological regimes in the long run? This is a complicated public choice problem, requiring theoretical modeling, and there are few water price datasets available with sufficient geographic scope that could be used to empirically

[1] According to Hanak et al. (2012), advances in the interpretation of satellite imagery are making it possible to estimate crop water use and groundwater depletion in the western United States. See MacEwan et al. (2010) and www.idwr.idaho.gov/GeographicInfo/METRIC/et.htm.

estimate the determinants of the level of water prices to test such a model. One recent study considers water prices in 319 U.S. cities in 40 different states between 1995 and 2005, and demonstrates that marginal water prices, on average, are lowest in the western states – the U.S. region in which water scarcity is now and has, historically, been of greatest concern (Bell and Griffin 2008). While careful analysis would be required to determine whether, *ceteris paribus*, cross-sectional differences in climate have influenced the chosen level of water prices, the fact that aridity and marginal price levels may be negatively correlated in the U.S. is not an encouraging sign. Correspondingly, how will the metering of water use, so that volumetric water prices can be implemented, evolve, especially in developing countries? It may be that there are historical changes in energy metering and pricing that would serve as a guide to what might happen for water. Reviewing this literature is beyond the scope of the current paper, but is an important area for further research.

2.2.2 Responsiveness to non-price water demand management policies

Urban water suppliers have typically relied on nonprice conservation programs, more than prices, to induce demand reductions during shortages. These programs fall into three main categories: (1) required or voluntary adoption of water-conserving technologies, (2) mandatory water use restrictions, (3) social comparison and information policies; and (4) mixed nonprice conservation programs. These policies have primarily targeted residential customers, so little is known about their potential impact on water consumption for other sectors.

Water-Conserving Technology Standards. When the water savings from technology standards have been estimated, they have often been smaller than expected because of behavioral changes that partially offset the benefit of greater technical efficiency. For example, households with low-flow showerheads may take longer showers (Mayer et al., 1998). The ‘‘double flush’’ was a notorious difficulty with early models of low-flow toilets, though that may be less of an issue with contemporary models (Bennear et al. 2012). In a recent field trial, randomly selected households had their top-loading clothes washers replaced with front-loading models. The average front-loading household increased clothes washing by 5.6%, perhaps because of the cost savings associated with increased efficiency (Davis 2008).

Several engineering studies have observed a small number of households in a single region to estimate the water savings associated with low-flow fixtures. One study indicates that households fully constructed or retrofitted with low-flow toilets used about 20 percent less water than households with no low-flow toilets. The equivalent savings reported for low-flow showerheads was 9 percent (Mayer et al. 1998). Savings reported for low-flow toilet installation and rebate programs range from 6.1 gallons per capita per day in Tampa, Florida to 10.6 gallons per capita per day in Seattle, Washington (U.S. General Accounting Office 2000). Renwick and Green (2000) estimate no significant effect of ultra low-flush toilet rebates in Santa Barbara, California. Such programs are difficult to evaluate, since it can be hard to determine whether adoption is really “additional”, due to the subsidy, or would have been accomplished even without a policy intervention (Bennear et al. 2012).

Mandatory Water Use Restrictions. Mandatory water use restrictions may limit the total quantity of water that can be used or restrict particular water uses. Empirical evidence regarding the effects of these programs is mixed. Summer 1996 water consumption restrictions in Corpus Christi, Texas, including prohibitions on landscape irrigation and car washing, did not prompt statistically significant water savings in the residential sector (Schultz et al. 1997). A longer-term program in Pasadena, California resulted in aggregate water savings (Kiefer et al.1993), as did a program of mandatory water use restrictions in Santa Barbara, California (Renwick and Green 2000).

Social Comparison and Information Policies. Economists have, very recently, begun to explore the impact of providing households with information on their water consumption relative to their neighbors, and estimating the impacts of such social comparisons on water use. Ferraro and Price (2012) implement a field experiment involving more than 100,000 households served by an Atlanta-area water utility. Their results indicate that social comparison messages had a greater influence on behavior (reducing water demand) than simple pro-social messages about the need to conserve during a dry summer, or technical information on how water conservation could be accomplished.

Mixed Nonprice Conservation Programs. Water utilities often implement multiple nonprice conservation programs simultaneously. One analysis of the effect of conservation programs on aggregate water district consumption in California found small but significant reductions in total water use attributable to landscape education programs and watering restrictions, but no effect due to indoor conservation education programs, low-flow fixture distribution, or the presentation of conservation information on customer bills (Corral, 1997). The number of conservation programs in place in California cities may have a small negative impact on total residential water demand (Michelsen et al. 1998). Public information campaigns, retrofit subsidies, water rationing, and water use restrictions had negative and statistically significant impacts on average monthly residential water use in California, and the more stringent policies had stronger effects than voluntary policies and education programs (Renwick and Green, 2000).

While a fair amount is known about the effectiveness of non-price water policies in urban settings, as is the case for research on water pricing, there is no careful model in the literature explaining how or why water utilities choose one or more non-price water conservation policies, or whether this choice of policies is sensitive to climate. Given the ubiquity of these water conservation policies, understanding the forms that such policies are likely to take under increased scarcity or hydrological variability is important, particularly since they have a significant cost-effectiveness disadvantage relative to increasing water prices to reduce demand (Mansur and Olmstead 2012).

2.3 Water supply, property rights, and water markets

Water pricing regimes and other water demand policies that reflect water scarcity are important potential adaptation tools. Another is the development of markets that move scarce water to its highest-valued uses, and the potential gains from water trading have attracted attention from economists for many decades (Hartman and Seastone 1970, Vaux and Howitt

1984, Saliba and Bush 1987). Informal water markets are common. For example, in India and Pakistan, farmers who can afford large groundwater wells with diesel and electric pumps sell water to smaller farmers who cannot afford such infrastructure, with payment taking the form of cash, labor, or share farming (Bjornlund and McKay 2002). However, given the potential gains from trade, formal, inter-sectoral water markets have been slow to develop (Easter et al. 1998).

The transaction costs for water marketing are important barriers to trade. These costs include the costs of physical infrastructure necessary for transporting water from sellers to buyers, search costs (i.e., identifying willing buyers and sellers), and the legal costs of creating and enforcing contracts and obtaining regulatory permission. Carey et al. (2002) find empirical evidence that transaction costs significantly diminish trading opportunities. Libecap (2011) emphasizes the role of the basic water management institutions in the American West, which emerged to enable agriculture and settlement of the arid regions west of the 100th meridian, in limiting the expansion of water marketing and reducing the potential to flexibly respond to climate-change-related hydrological uncertainty. Nonetheless, many studies have demonstrated potential and realized net benefits from trading, in areas as diverse as south Texas (Chang and Griffin 1992), southern Italy and Spain (Pujol et al. 2006), north-central Chile (Hearne and Easter 1997), Morocco (Diao and Roe 2003) and southeast Australia (Bjornlund and McKay 2002).

The largest intra- and inter-sectoral water markets have developed in Chile, Australia, and the American West. Chile's 1981 National Water Code established freely tradable water rights separate from land rights. Significant trading has taken place in north-Central Chile, but transactions have been quite rare in other parts of Chile (though more common in arid regions and during droughts), perhaps due to constraints posed by physical geography, infrastructure, legal and administrative complications, and cultural resistance by farmers (Bauer 2004). Australia's Murray-Darling river basin covers 14 percent of the total Australian land area and supports major agricultural production. Until 1980, withdrawal rights for irrigation in the basin were essentially unlimited. Water trading was introduced in South Australia in 1983, in New South Wales in 1989, and in Victoria in 1991. Permanent inter-state transfers are not allowed, and there are significant limitations on inter-regional sales, but intra-regional trading is active. A cap on water use in the basin was enacted in 1997. Trade appears to have promoted both higher value agricultural production and more efficient irrigation technologies (Bjornlund and McKay 2002).

In the American West, as discussed earlier, relative prices (particularly for urban vs. agricultural water users) provide signals of the potential for gains from water trading. A recent study of water marketing in twelve Western states between 1987 and 2005 suggests that prices are higher, on average, for agricultural to urban transfers than for transfers between agricultural producers, and that this difference is growing over time (Brewer et al. 2008). Water right sales are increasingly more common than short- and long-term leases, and states with the most urban growth appear to engage in the most water trading. A study of trades in Arizona, Colorado, and New Mexico water markets suggests that water prices are lower in wetter periods (supply shifting out) and that income growth (demand shifting out) drives up prices, findings that are consistent with standard economic theory (Brookshire et al. 2004). In addition, areas with higher-valued agricultural productivity tend to have a lower quantity of water traded (Brookshire et al. 2004).

Where they have been implemented, the activity in water markets is consistent with economic theory, with water flowing from lower- to higher-valued uses. Loomis et al. (2003, p. 242) suggest that "climate change may finally break our anachronistic restrictions on the freedom of water rights holders to seek the most valued uses for their water." Others are more skeptical about the ability of existing institutions to foster the more robust water markets that would aid in climate adaptation (Libecap 2011). Taking a historical view, decisions were made as the American West was settled about public investments in irrigation, hydropower, and urban water storage and conveyance infrastructure based on limited information about "typical" runoff and flow regimes in regional rivers, and based on the priorities of the era (enabling agriculture). Historic water allocations are, for the most part, locked in, and institutions are relatively inflexible, and not easily adjusted to new circumstances (Libecap 2011).

However, it is notable that all of the water markets discussed here, and essentially anywhere in the literature, have emerged in arid regions, during periods in which the opportunity cost of historic allocation regimes at least appeared to be increasing. Slaughter and Wiener (2007) point out that Colorado, squeezed by an old mining system of water rights and prior appropriation, significant urban and industrial growth, a semi-arid climate, and little groundwater, has led the evolution of property-rights-based water law, and has, arguably, the most robust water market in the American West. Like the evidence offered above on water pricing, this is anecdotal, and to my knowledge, there is no empirical work that carefully examines the role of climate or other factors in water market emergence or growth. But it is striking that, while water prices, on the whole, do not seem to be higher in more arid regions, water marketing is more prevalent in arid regions. In a Coasian sense, the mere existence of the potential gains from trading water creates pressure for trade to occur – so long as the property rights are clearly assigned. There is no such equivalent pressure for public water rate-setting institutions to raise water prices – rate increases are largely a function of water supply cost increases, which are related to the opportunity cost of urban water supply only indirectly, in most cases.

One of the biggest challenges to welfare improvement from water marketing is dealing adequately with externalities and public goods. Return flows present an important externality. For example, irrigation water not lost to evapotranspiration either recharges groundwater aquifers or augments surface water flows; water transferred to coastal cities may be returned to the ocean through offshore wastewater outfall systems (and urban uses, in general, have a higher consumptive component). The spatial component of water withdrawals and return flows is, therefore, an important consideration in water trading, just as the location of emissions is an important consideration in market-based approaches to water quality regulation. When instream flows have value, water market outcomes can be Pareto optimal only when transferable diversion and consumption rights are established, return flow coefficients are established to identify the location of each diverter's return, and institutional mechanisms are established to create a market presence for instream flow values (Griffin and Hsu 1993).

2.4 Water supply infrastructure and operations

The magnitude and direction of climate adaptation through water infrastructure investments, and changes in infrastructure operation, are critical, because the main purpose of much water resource infrastructure is smoothing in the variability of water supply, either storing water in preparation for intra-annual dry seasons or periodic droughts, or maintaining sufficient storage capacity to absorb excess flows during rainy seasons or periodic floods. And there is significant evidence that adaptation to climate-related changes in the frequency and severity of weather extremes related to water resources (drought and flood) will be more difficult than adaptation to changes in mean temperature and precipitation (Reilly 1999, Hansen et al. 2011).

There is significant empirical evidence that the availability of irrigation provides a buffer against the economic risk from agricultural productivity losses associated with periodic drought (Hansen et al. 2011). Studies also show that irrigation adoption is sensitive to environmental conditions (Schlenker et al. 2005, Dinar et al. 1992). Examining this evidence in the agricultural sector is beyond the scope of this paper. However, infrastructure could, similarly, play an important role in adaptation in other water-using sectors.

Many government agencies and other institutions have issued reports suggesting that the costs of adapting non-agricultural water infrastructure to climate change will be significant (European Environment Agency 2007, California Department of Water Resources 2008, U.S. Environmental Protection Agency 2012). Relevant costs for municipal water and wastewater infrastructure may include: construction or enhancement of flood barriers, or green infrastructure, to protect existing facilities (e.g., low-lying water or wastewater treatment plants); creation or enhancement of infrastructure for natural or artificial groundwater recharge and storage; increased reservoir storage capacity (raising dams, removing sediment from reservoirs, lowering water intakes); and relocation of existing gray infrastructure to higher ground (in coastal areas, for example).

The most significant empirical work to date on the likely extent and cost of such measures in industrialized countries develops engineering cost estimates of adaptive infrastructure investments, and then considers how much these costs could be reduced if water prices increase to reflect growing scarcity, reducing demand and thus reducing the magnitude of needed infrastructure investments (Hughes et al. 2010). The paper examines the costs of water supply, water treatment, and sewage treatment for municipal (residential, commercial, and industrial) use. Results suggest that the costs of adapting existing municipal water infrastructure to climate change are less than 2 percent of total baseline infrastructure provision costs in OECD countries. In addition, these adaptation costs would be reduced dramatically if prices are used to "cap" any shift outward in water demand due to climate change (accounting for the loss in consumer surplus from reduced consumption). This work represents the best empirical evidence, to date, of the likely extent and cost of adaptation to climate change impacts on municipal water resource infrastructure in industrialized countries, and it is a very useful starting point for integrating information about adaptation in municipal water infrastructure into IAMs.

To be used for this purpose, however, there are several issues that would need to be addressed. First, the econometric estimates the authors generate to measure the responsiveness of municipal water infrastructure demand to climate parameters may be problematic. The authors regress measures of water and wastewater infrastructure demand on climate variables, controlling for other factors, but some of these intermediate results in the paper suggest that they are not sufficiently controlling for unobservables. For example, they obtain negative coefficients on temperature for urban and rural drinking water service coverage, as well as wastewater service coverage (Hughes et al. 2010, p. 149). It is likely that this is due to the fact that warmer OECD countries (in southern Europe, or Latin America, for example) have less extensive piped water and sanitation service provision than cooler ones, and that the characteristics that determine this are not sufficiently controlled for in the analysis. Further work to establish robust estimates of these relationships, controlling for unobservables, would be critical to the usefulness of a model like that in Hughes et al (2010) in an effort to integrate adaptation modeling into IAMs.

Second, the authors assume that the amount of water available for future municipal and industrial use is held constant at current levels, and that any reduction in water availability in OECD countries due to climate change will be taken from agriculture. The losses from agricultural impacts are not represented in the model (though they could be, in a separate agricultural adaptation model linked to the same IAM), and if they were, this would increase estimated adaptation costs. In addition, while the idea that agriculture will bear the brunt of reduced supplies is a reasonable assumption in these wealthier countries, in many places (such as the western United States and Australia), cities may need to compensate agricultural producers for the required water transfers, given the current allocation of water rights, shifting distributional impacts relative to a “no compensation” model.

Third, the paper assumes that any *increase* in municipal water *demand* due to climate change results in additional water recycling or desalination, at a constant marginal cost. This assumption is more tenuous, as the cost of acquiring additional supplies from agriculture, where feasible, is likely much lower than the marginal cost of recycling and, almost certainly, desalination in most OECD countries. To my knowledge, desalination is not a current water supply choice on a large scale in any OECD countries; recycling for municipal use may be more common, but it is still not a typical choice, given its high costs (with the exception of some industrial uses, such as cooling). This is one reason why the impacts of pricing on the authors' estimated adaptation costs are so significant – price increases displace the need for some very expensive recycling and desalination. Indeed, the authors note that the increased costs of water and wastewater treatment to meet increased demand essentially drive their cost estimates, in all regions (Hughes et al. 2010, p. 151). An important extension of this work would incorporate more realistic assumptions about the costs of increasing water supply, on the margin, in various regions and countries.

A fourth problematic assumption is that countries invest in the efficient level of infrastructure in each period, and replace existing infrastructure at the end of its useful life – this, unfortunately, is not what is generally observed in the real world. For example, U.S. water distribution systems are aging, and portions of many large urban systems have exceeded their anticipated "useful life," leading to increased leakage, water main breaks, and service disruption.[2] Here again, the key to understanding what will actually happen in terms of climate adaptation in the water sector is a robust model of how water management institutions will react.

[2] At one large utility in the Midwest, the number of main breaks increased nearly tenfold, from 250 to 2200 per year, between 1970 and 1989 (NAS 2005). The most recent wastewater survey (2008) estimates that $322 billion is

If Hughes et al. (2010) is a good starting point for understanding municipal water infrastructure adaptation in industrialized countries, what about developing countries? In contrast to the rather small estimate of the costs of water infrastructure adaptation for OECD countries in Hughes et al. (2010), water supply and flood management adaptation costs are among the top three categories of estimated adaptation costs for developing countries, according to the World Bank Economics of Adaptation to Climate Change study (Narain et al. 2011). The portion of that study focusing on municipal and industrial water supply assumed future climate-related increases in water demand would be met through increasing surface water storage in reservoirs, with some constraints (Ward et al. 2010).[3] The researchers then developed storage-yield curves for selected global river basins, for two different climate scenarios, and estimated the construction costs of expanding reservoir storage for each basin. According to their "best estimate" – assuming the future global distribution of dam and reservoir size would be similar to the current distribution – the results imply an increase in global reservoir storage capacity through 2050 of 2800 to 3000 cubic kilometers, at an annual average net cost of about $12 billion (Ward et al. 2010).

These results are a useful starting point, but again, it is not at all clear that water management institutions would achieve the necessary climate-related increases in water supply for municipal and industrial uses solely through reservoir construction and expansion. The demand estimates in Ward et al. (2010) do assume some increase in municipal water use

needed for projects and activities to address water quality or related public health problems in the U.S. over the next 20 years (Copeland and Tiemann 2010).

[3] Annual water withdrawals in a given basin were assumed not to exceed 80 percent of total river runoff, and the cost of increased water supply from reservoir yield was capped at $0.30 per cubic meter, assuming that alternative measures (e.g., recycling, desalination, rainwater harvesting) could provide additional supply at that cost.

efficiency, so conservation is effectively substituting for some foregone expansion in reservoir capacity that would otherwise be necessary in the modeling. However, as the authors note, the reasons not to construct new dams and reservoirs are legion, new dam construction faces significant political opposition in many regions, and it is not clear why jurisdictions would choose this option on such a large scale, or what the cost implications would be of a more balanced portfolio of supply expansion choices.

In addition to water infrastructure installation, expansion and modification, climate adaptation could include changes in the operations of existing infrastructure. For example, reservoir levels could be lowered (within the limits of existing engineered systems) to hedge against increased flood risk, or raised to prepare for anticipated reductions in low flows during a dry season. There is an engineering literature on optimal reservoir operations under conditions of uncertainty, which could be a useful starting point for thinking about how these practices might, in reality, be altered by water managers in response to climate change. With few exceptions (Raje and Mujumdar 2010), the available literature focuses on U.S. river basins (Brekke et al. 2009, Lee et al. 2009, Li et al. 2010). Further research by economists, in cooperation with engineers, would be necessary to incorporate models of adaptation through reservoir operations into IAMs.

2.5 Transboundary water resource management

Empirical analyses of water pollution spillovers in transboundary settings have found that countries, and even states and counties, free-ride in water quality. Pollution levels are higher near international borders (Sigman 2002; Bernauer and Kuhn 2010) as well as near subnational borders within countries (Sigman 2004, Lipscomb and Mobarak 2008). Water pollution emissions by U.S. pulp and paper plants appear to be higher when out-of-state residents receive a greater share of pollution control benefits (Gray and Shadbegian 2004). Water pollution spillovers may also intensify as the number of political jurisdictions managing the same river increases (Lipscomb and Mobarak 2008). There is also substantial anecdotal evidence that political jurisdictions free-ride in water quantity or allocation, in addition to water quality, and there has been some modeling of this phenomenon in the economic literature (Rogers 1969, Gisser and Sanchez 1980, Loehman and Dinar 1994, Becker and Easter 1999). However, there is no *empirical* evidence in the literature of free-riding in transboundary water allocation.

Domestic river basins may require cooperation among sub-national jurisdictions, and free-riding can be a problem in these cases. But the market failures in water allocation regimes in transboundary river basins may be even more severe.[4] Thus, the case of transboundary water resource management may be one where reactions to climate-related scarcity and other changes in hydrological regimes may be more maladaptive than adaptive. This is a significant concern, as the watersheds of the world's 261 international rivers cover more than 45 percent of the Earth's surface (Wolf et al. 1999).

A recent study of global transboundary river basins identifies those "at risk" due to the combination of: (1) expected future increases in hydrological variability due to climate change; and (2) weakness (or absence) of treaties and other institutions to manage water allocation

[4] This empirically testable hypothesis is the focus of new research in progress, funded by the World Bank (Olmstead, Sigman and Zhang 2012).

(DeStefano et al. 2010). The 16 "at risk" basins are in Africa, East Asia, Eastern Europe and Central Asia, and Central and South America, with the majority (10) in Africa. This and other studies on the topic were not written from an economic perspective, and they do not model any evolution in river management institutions due to increased scarcity. However, economic theory would suggest that if resources dwindle or become less predictable over time, and they are essentially open access, the incentive to over-exploit them will increase, rather than decrease. Thus, water management scenarios with "maladaptation" to climate change may be even bleaker than without, as reactions to increased scarcity may exacerbate existing inefficiencies.

From the perspective of international law, treaties are typically rigid institutions, negotiated to last over long timeframes, and alterable only under limited conditions by mutual agreement. Recent decisions by the International Court of Justice over countries' ability to withdraw from treaty obligations due to what seem, from an economic perspective, to be reasonable arguments about changed circumstances (e.g., the collapse of the Soviet Union and new scientific knowledge about the impacts of dams) suggest that it will not be easy for countries to lobby for treaty modification due to climate-induced changes in water availability (McCaffrey 2003). Treaties resilient to climate change must, then, be written that way, *ex ante*. Once the river basin impacts of climate change become apparent, there will be winners and losers, making it difficult to reach mutual agreement on adaptation measures, and international law will stand on the side of countries supporting the status quo. This asymmetry is likely to cause conflicts, and exacerbate inefficiencies – a problem similar to the inertia in domestic water rights regimes that prevents more robust domestic water trading, discussed by Libecap (2011).

McCaffrey (2003) offers guidance on treaty characteristics that would enhance resilience to climate change: (1) requiring periodic agreements on water use and management; (2) establishing a short treaty lifetime, which can be renewed automatically only without objection by any party; (3) making special provisions for particular circumstances (e.g., droughts); (4) including provisions allowing parties to terminate without notice; (5) including "joint contractual plans" that provide details regarding specific projects such as dams and reservoirs, but allowing technicians to respond flexibly as conditions change; and (6) empowering a joint institution with appointees from each party nation to make, or just recommend, adjustments to treaty regimes. From the perspective of game theory, all of these but (3), and possibly (6), would seem to create significant incentive problems in treaty negotiation.

A growing body of research examines conditions for adoption of international water management institutions (Wolf 1998, Bernauer 2002, Song and Whittington 2004, Dinar 2009), but few studies attempt to assess their effectiveness rigorously. An exception is Mobarak and Lipscomb (2009), who study the effects of basin-level watershed management in Brazil. Previous research on air pollution provides reason for skepticism about the extent to which international environmental treaties constrain behavior (Murdoch et al. 1997, Beron et al. 2003). However, there are numerous examples of successful transboundary water management outcomes (Barrett 2003), and cases in which institutions appear to mitigate free-riding (Sigman 2002). Thus, as for many of the water resource management questions considered in this paper, the key future research question is whether and how transboundary river management institutions will evolve in the face of climate change.

3. REMAINING GAPS IN THE EMPIRICAL LITERATURE

The most significant remaining gap in the empirical economics literature on water resource management under climate change is consistent across all of the topics addressed in this paper – regional economic impact and adaptation models, for the most part, lack any formal modeling of the political economy of water management. Given the way in which water resources are managed, largely by non-market institutions, this work is at least as critical in generating useful adaptation inputs for IAMs as are better downscaled climate predictions, or hydrologic and engineering modeling. This is not the first paper in the economics literature to call for more empirical research on water management institutions, including comparative studies of river basins with different institutions, and their relative effectiveness in mitigating variability, scarcity, water quality degradation, and other problems likely to increase as the climate changes (Blomquist et al. 2004, Slaughter and Wiener 2007). But as the likelihood of significant and timely climate change mitigation efforts dwindles, understanding how water management institutions will respond to climate change impacts becomes more important.

The literature discussed in Section 2 does contain some estimates of the economic impacts of the water resource effects of climate change, assuming that no adaptation will take place, as well as estimates of impacts assuming that water markets will respond in a dynamically efficient manner, maximizing the net benefits of water resources over time. Both sets of assumptions are highly problematic. Integrated assessment modelers could, as a first step, integrate both of these classes of results into existing IAMs. However, it is not clear that the "no adaptation" models provide an upper bound on damages, given the possibility of maladaptive institutional responses in the water sector, which would exacerbate existing inefficiencies. The "efficient adaptation" models could provide a reasonable lower bound on damages.

Alternatively (or in addition), new models of national, regional and/or global adaptation in water resource management with realistic economic and political-economic assumptions about the responsiveness of institutions to climate change, could be developed. The literature currently offers little guidance on the extent to which the prospect of climate change will alter:

(a) the level and structure of water prices;
(b) reliance on non-price water conservation mandates, incentives, and other policies;
(c) legal property rights regimes for water
(d) the allowable extent of and constraints on transferring and leasing water among users, within and across basins;
(e) investment in water supply infrastructure;
(f) water supply infrastructure operations; and
(g) water allocation institutions in transboundary river basins.

Once the ways in which these various policies available to water managers may change over time are modeled, IAM researchers could then apply what is known from the rich available literature on the responsiveness of households, industry, and agriculture to these various policy levers to understand sector-level changes in water use, and their economic implications.

4. POTENTIAL CONTRIBUTIONS OF CONNECTING EMPIRICAL RESEARCH AND IAMs

Most integrated assessment studies of climate change focus exclusively on analyzing the economic implications of greenhouse gas emissions mitigation policies, while giving little or no attention to formal modeling of adaptive responses (Fisher-Vanden et al. 2011). With recent advances in the downscaling of predictions from global climate models to the regional level, there are increasingly rich possibilities to incorporate adaptive responses into IAMs. Given the existing literature on adaptation to the water resource impacts of climate change, and the gaps therein, what would be the potential contributions of using some of the results from this literature to formally link integrated assessment and adaptation models for water supply and demand?

The likely magnitude and breadth of climate change impacts through water resources on multiple sectors of the economy suggests that these efforts would reap significant dividends, in terms of new knowledge about the economic impacts of climate change. The most important water resource adaptation processes to model may have to do with subject matter outside the scope of this paper: (1) agricultural adaptation, through changes in irrigation and other mechanisms; and (2) adaptation to changes in the distribution (frequency and severity) of extreme events like drought and flood. These two aspects of water resource adaptation will be critical due to the fact that irrigation currently represents 70 percent of global withdrawals and 90 percent of consumptive use of water (Shiklomanov and Rodda 2003), and because the challenges of adapting to changes in low-probability, high-consequence events may be more formidable than those of adapting to slow shifts in average runoff and other aspects of regional hydrological regimes that are predicted to occur with climate change.

However, the world is increasingly urban; about one-half of the global population resides in cities, and virtually all expected population growth in the next three decades is expected to occur in developing country cities (Cohen 2006). Providing sufficient water supply for human consumption, sanitation and wastewater treatment is a critical challenge that has not yet been met for current populations, and these needs will only increase in future decades (WHO/UNICEF 2010). Indeed, the IPCC notes that changes such as population growth pose greater challenges for water resource management in the long run than climate change, itself – for example, expected future increases in the global population living in "water stressed river basins" are driven more by population growth than by various climate changes scenarios (Bates et al. 2008). These pressures, and the potential they hold to shape the evolution of water resource management institutions, in combination with changes in hydrological regimes due to climate change, suggest that there would be significant potential benefits to incorporating water supply and demand adaptation modeling into IAMs.

4.1 Priority empirical findings to be incorporated into IAMs

With no expertise in the development of IAMs, it is difficult for me to describe precisely which findings in the current literature could be used directly to incorporate adaptation into these modeling efforts. However, in my view, the low-hanging fruit with reasonably high priority include the following three categories of available information.

(a) *Efficient water resource adaptation models for specific river basins*. The results of existing studies such as Hurd et al. (2004) for particular river basins, assuming dynamically efficient adjustment to new hydrological regimes in terms of water allocation, could be incorporated into IAMs as “lower bound” estimates of climate change’s water-resource-related economic impacts in these basins. In order for this step to be taken on a broad scale, the development of similar models for additional global river basins, with a focus on the largest and/or most highly populated or water-constrained basins, would be necessary. However, the process of integrating such results into IAMs could be piloted in the short run using existing studies.

(b) *Global models of adaptation through municipal water supply infrastructure.* The model in Hughes et al. (2010), with some potentially straightforward changes in the econometric approach and underlying assumptions, may be usefully incorporated into IAMs. Linkage would need to be made to agricultural impacts in some fashion, since additional water supply for urban and industrial uses in Hughes et al. (2010) is taken from agriculture.

(c) *Information regarding existing transboundary water resource institutions*. As noted earlier, new, rigorous political-economic modeling of transboundary water management institutions would be needed to incorporate information about how global water supply will be allocated under new conditions predicted by climate models. However, current global transboundary water treaties, and the constraints they imply for water withdrawals, dam and reservoir construction, and other management activities are well represented in existing databases (Oregon State University 2010). As a starting point, researchers could take the current parameters of these treaties into account as water resource impacts and adaptation are linked to IAMs, assuming that the status quo is a reasonable representation of future transboundary resource management regimes.

4.2 Necessary level of detail to address water management adaptation in IAMs

Transboundary water resource management is generally negotiated between countries (where basins are shared across country borders), thus, the allocation of water supply at this level can be modeled at a somewhat coarse resolution, and the data that would be required to do so are available in reasonably concentrated form. Digital data on existing dams and reservoirs can be obtained from the International Commission on Large Dams (2003) and the Global Reservoir and Dams Database (Global Water System Project 2012), and data on regional water management treaties can be found in the Transboundary Freshwater Dispute Database (Oregon State University 2010). However, for all of the other issues addressed in this paper, water management decisions are made, for the most part, at the local and regional level, posing a significant challenge for both modeling and data collection. For example, one critical input for integrating adaptation to the water resource impacts of climate change into IAMs is data on global water prices. Unfortunately, to my knowledge, there is no existing source of reliable, detailed information on local and regional water prices across sectors, even within industrialized countries. The collection of data on water prices would be very time-intensive, though the value added by these data would be significant.

Several research and environmental advocacy institutions engage in data collection across countries and regions for some of the other significant water demand and supply issues raised in this paper, however. For example, the Pacific Institute issues biennial reports on global freshwater resources, which typically include detailed tables on freshwater supply, withdrawals, water and sanitation coverage, and other water management issues (Gleick et al. 2012).[5] The World Resources Institute manages a "water risk" database and atlas, capturing physical data on various aspects of water supply, as well as local regulatory characteristics (World Resources Institute 2012). While databases like these are summary in nature and populated from a variety of sources of varying quality and consistency, they could serve as a useful starting point for high-level modeling efforts. Given the interest of these institutions in the water resource implications of climate change, and their existing data collection networks, they might serve as useful partners for new data collection efforts to support integration of adaptation modeling into IAMs, given a sufficient source of funding.

5. CONCLUSIONS

The over-arching theme that emerges from this analytical review of the existing empirical literature is that further research on the role of water management institutions in adaptation to climate change will be critical to any comprehensive effort to incorporate water-resource-related impacts into IAMs. The "supply curves" for water management policies are poorly understood, where they are understood at all. How are current water prices, conservation policies, infrastructure investment programs, water markets, and transboundary water allocation treaties designed and implemented, and how will these processes shift in the face of the anticipated impacts of climate change on hydrological regimes?

We know a good deal from the existing empirical literature about responses by end-users of water – municipal, industrial, and agricultural – to manipulation of these various water policy

[5] The earliest version of this report is for the years 1998-1999 (Gleick 1999).

and management levers, and this information will provide valuable input to IAMs. As a start, modelers could assume that each of these classes of institutions will be stationary, even under non-stationary climate conditions, and utilize existing estimates of responsiveness to these policy levers to quantify the economic implications of adaptation within IAMs.

But the development of new models of municipal "water policy supply" under various climate conditions, similar to the work already done on changes in agricultural irrigation supply, would likely be much more illuminating. For example, such models could allow for the possibility that water management responses to climate change may be maladaptive, rather than adaptive, if increasing scarcity and hydrological variability exacerbate existing inefficiencies in water management, for which there is significant empirical evidence in the literature. These existing inefficiencies include lower-than-efficient water prices, free-riding in water quality in transboundary river basins, inefficient allocation of water across sectors by historic water rights regimes, and excessive depletion of open access groundwater aquifers.

Nonetheless, some progress is possible given empirical results and available data from the existing literature. Models of dynamically efficient water resource adaptation have been developed for specific river basins, which could be used as "lower bound" estimates of the incorporated water-related impacts in these basins, though additional modeling efforts would be required for the global river basins as yet unstudied for this purpose. Large-scale modeling efforts have also been undertaken regarding the potential impacts of climate change (and adaptation) on the demand for and supply of municipal water storage infrastructure. There are also centralized sources of information on existing constraints on water supply allocation in transboundary river basins, which could be incorporated into IAMs’ water resource modeling. All of these would be useful starting points for a more comprehensive effort.

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