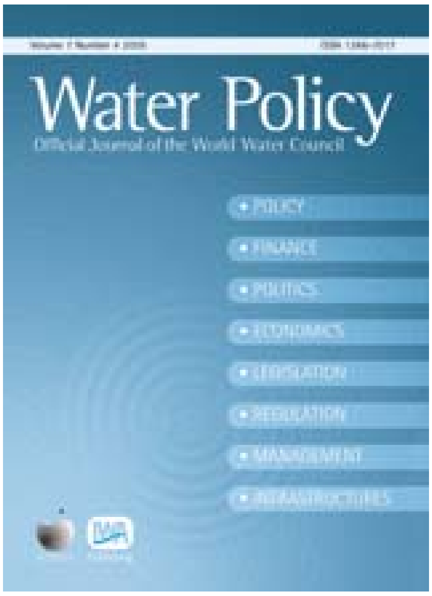
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# Water markets as a response to scarcity

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**Abstract**

Existing water governance systems are proving to be quite ineffective in managing water scarcity, creating severe risk for many aspects of our societies and economies. Water markets are a relatively new and increasingly popular tool in the fight against growing water scarcity. They make a voluntary exchange possible between interested buyers and sellers of water rights. This paper presents direct evidence from seven water markets around the globe to document key economic and ecological challenges and achievements of water markets with respect to water scarcity. We specifically approach water markets as localized cap-and-trade systems, similar to those for carbon emissions. We examine whether water use remains within the set limits on use of water rights (i.e., under the cap), the degree to which water markets help protect the health of ecosystems and species, and whether (as predicted by economic theory) the explicit pricing of water is accompanied by improving efficiency, as less productive water users decide to sell water to more productive water users.

*Keywords:* Cap-and-trade; Closed basins; Edwards Aquifer; Murray–Darling Basin; Northern Colorado Water Conservancy District; Sustainable water use; Tragedy of the commons; Water allocation; Water markets; Water scarcity

## 1. Introduction

Growing water demands for cities, farms and industries are bumping up against the limits of water availability in many regions. In water-scarce regions, urban water managers are struggling to secure

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additional water supplies to support city growth (Richter *et al.*, 2013), agricultural production is suffering as irrigation supplies decline (IWMI, 2007) and freshwater species are becoming increasingly imperiled due to heavy depletion of water sources (IUCN, 2013). Water scarcity has become a major challenge of the 21st century.

Existing water governance systems are proving to be quite ineffective in managing water scarcity in most regions, creating serious risks for many aspects of our societies and economies. Water managers have allowed water sources such as rivers and aquifers to become exploited to such a heavy degree that when dry periods eventually arrive and water becomes less available, managers are unable to sufficiently reduce water demands to avert water crisis. The resulting water shortages are disrupting local and regional economies, causing social conflict and political instability, and damaging ecosystem health (United Nations, 2012; Richter, 2014).

Despite reports of a global water crisis, however, the world is not running out of fresh water (IWMI, 2007). Water is a renewable global resource of which there is more than enough to meet both current and projected human needs. The basic problem is that water availability and needs are mismatched in space and time, and rearranging water's availability with human needs can be very expensive and ecologically damaging. As a result, increasing water demand due to population growth and rising living standards is straining locally-available water resources in many parts of the globe, and climate change will worsen an already bad situation in many water-scarce areas (Droogers *et al.,* 2012; Debaere, 2013). Calls for better water management to alleviate scarcity are being heard around the world (United Nations, 2012).

Water management responses to scarcity have been dominated by supply-side engineering, focused on the construction of reservoirs, canals, pipelines, wells and other infrastructure. In recent years, however, due to the rising cost of water supply projects and growing environmental concern, there has been a marked shift toward economic innovations, including strategies to manage water demand (Griffin, 2006; Zetland, 2011). It is in this context that water markets are being increasingly promoted and applied. A water market, as used here, constitutes a system of formal rules and regulations that govern the buying, selling and leasing of water use rights (also often called water entitlements) that are ideally traded independent of land titles. Many market advocates suggest that markets allocate existing water supplies more productively and coordinate the various demands on water resources more flexibly. This flexibility, however, is often conditioned by the availability of a water infrastructure to store and distribute water among users; interested buyers and sellers may be constrained by the inability to move water from one place to another and to store it until it is needed. The flexibility in water trading can also be inhibited to varying degrees by governmental policies that restrict trade, such as limitations placed on the volume of water that can be moved from one water use sector to another, or the volume that can be traded across geopolitical boundaries, as will be discussed below.

There is a growing literature on water markets (see the review in Chong & Sunding, 2006) with no shortage of papers that argue the pros and cons of water markets analytically, or that calculate potential economic gains that market-like transactions might bring (Vaux & Howitt, 1984; Easter *et al.*, 1998; Hearne, 1998; Sunding, 2000; Chong & Sunding, 2006; Griffin, 2006). Grafton *et al*. (2011) presented a high-level, comprehensive comparison and ranking of water markets in the United States, Australia, Chile, South Africa and China, showing how markets function under very different legal and institutional frameworks, and what these imply for efficiency, equity and sustainability. However, the analysis conducted by Grafton *et al.* (2011) was conducted at a relatively high level of aggregation and in a largely qualitative manner, due to the reality that nationally and internationally comparable data for water markets are simply not accessible or do not exist.

In this paper we examine a select group of markets where data do exist, thereby illustrating empirically the challenges and accomplishments of existing water markets around the world. We use direct evidence from seven water markets to document with available data the key economic and ecological challenges and achievements of water markets. In each case we select one or two markets that illustrate a particular aspect most clearly. We studied four markets within the United States: the Edwards Aquifer of Texas, the Northern Colorado Water Conservancy District, the Central Valley/San Joaquin River of California, and the Columbia River basin in the Pacific Northwest. We also examined three markets outside of the United States: the country of Chile, the Murray–Darling Basin of Australia and the Santiago River basin of Mexico.

Our assessment of water markets focuses on their abilities to effectively manage water scarcity for both economic and ecological benefits. Specifically, we assess water markets as localized cap-and-trade systems, examining whether water use remains within set limits (i.e., under the cap), the degree to which ecosystem health is protected, and whether economic efficiency or other measures of economic productivity appear to improve. We also analyze various impediments or challenges to water trading. In this way, we go beyond the common focus on market transactions and economic outcomes, and consider the impacts of markets on environmental sustainability as well.

## 2. The case for water markets as a management strategy

Water markets emerge in different ways. Some develop gradually, as in southeastern Australia, or are catalyzed by a precipitating event, such as a lawsuit to protect endangered species in Texas or a decision by the Mexican government to decentralize water infrastructure operations in the wake of a financial crisis. Common to most markets is the presence of water scarcity, that is, they emerge as water demands begin to approach the limits of water availability.

Water availability can be influenced both by natural water supply as well as by socially-imposed restrictions on water use. An example of an imposed restriction is when a government or community decides to limit (cap) water use within a specific freshwater source (i.e., a river basin or aquifer) to protect overall water security, or to protect freshwater ecosystems and other values such as river-based cultural practices or recreation (Figure 1). The imposition of a cap is intended to arrest the 'tragedy of the commons', as framed by Hardin (1968). Hardin explained that a tragedy of the commons results when individuals with free access to a scarce but shared resource independently pursue their own self-interest: they deplete the common resource even though they understand their actions are contrary to society's long-term interest because they do not bear the full cost of their actions (Gordon, 1954). Water managers commonly use a regulatory cap in combination with properly defined water rights – which limits the total volume of water that may be extracted or consumptively used – in their efforts to arrest or avert a tragedy of the water commons.

Caps on water use can be set proactively or reactively. Setting a proactive cap, that is, capping water use at a level not yet reached, is highly preferable because it protects existing uses and does not force immediate adjustments in use. In contrast, considerable anxiety and controversy can erupt when cap levels are set at a level lower than existing use. Generally, a reduction of use has been pursued in one of three ways: (1) the government can institute mandatory, regulatory reductions in water rights; (2) water allocations to rights holders can be adjusted proportionally or in a hierarchical fashion (for

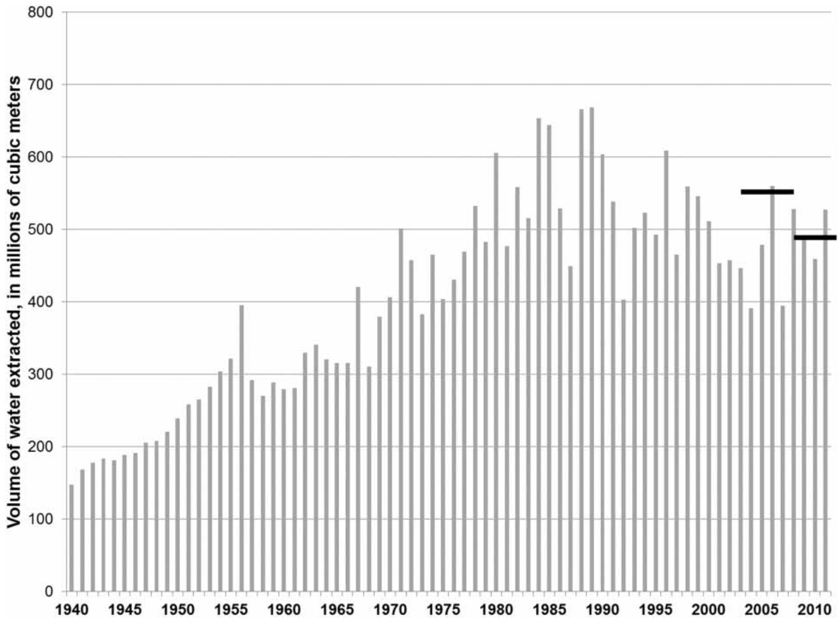


Fig. 1. Water extractions from the Edwards Aquifer in Texas rose sharply until the early 1990s. In 1993, in response to a federal court ruling, the Texas legislature set a cap on pumping at 555 million $m^3$ (450,000 acre-feet) by 2004, to be lowered to 493 million $m^3$ (400,000 acre-feet) by 2008, to protect endangered species living in the aquifer or dependent on aquifer-fed springs (black lines indicate cap levels). In 2007, the cap was replaced by new regulations that sharply reduce pumping during droughts. Total well pumping from the aquifer has generally decreased since 1990, while the population of the region increased by nearly 50% (*Source:* Eckhardt, 2013).

example by seniority of water rights) to match projections of water availability each year; or (3) the overuse can be eliminated through government or private 'buy-backs' of water rights, thereby bringing the volume of rights within cap limits. None of these approaches has been implemented without controversy because each has implications for the livelihoods and sense of security among water users. What is of the utmost importance to the regulation of water use and proper functioning of markets, however, is that water use is constrained within the volume of issued water rights.

As detailed below, with specific market examples, the setting of a cap can create a strong stimulus for water trading and investments in efficiency improvements. Water markets help to facilitate a voluntary exchange of the right to use water between buyers and sellers when water supplies are limited. As such, water markets are an application of a cap-and-trade system, similar to those in place for trading rights to emit atmospheric pollutants such as carbon or sulfur dioxide, or to release nutrients into waterways (Tietenberg, 2007; Tietenberg & Lewis, 2012). Water trading commonly redistributes water among competing users or sectors (e.g., Figure 2). Additionally, the imposition of a cap can create considerable incentive for water conservation. Once water use is capped, any future growth in water demands – such as adding more people to a city or increasing agricultural yields on a farm – can be met only by either accessing alternate water sources (e.g., water importation, desalination, etc.) or by increasing the efficiency of water use. The latter choice is almost always preferable from both a cost-effectiveness and environmental perspective (Richter *et al.,* 2013; Richter, 2014).

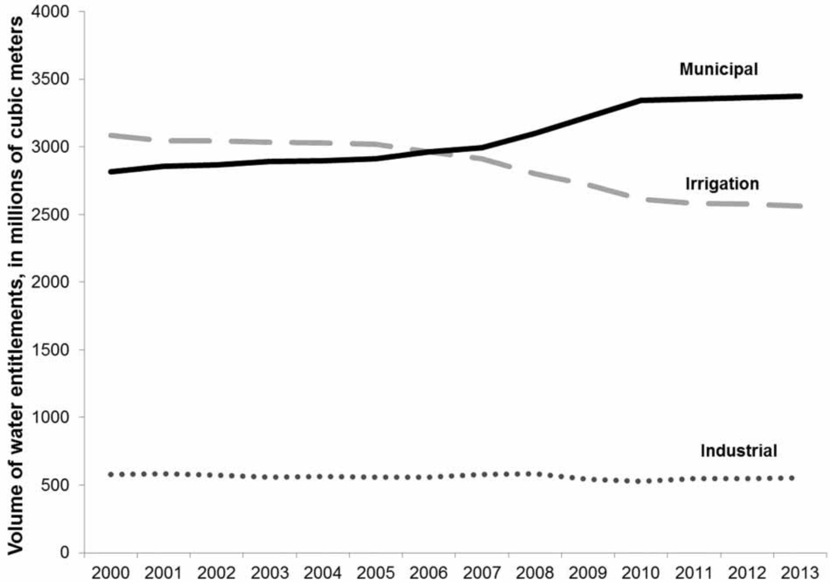


Fig. 2. Since the onset of water trading in the Edwards Aquifer, about 8% of all water rights holdings (>500 million $m^3$) have been transferred from irrigation into municipal use (*Source:* Edwards Aquifer Authority, 2013).

What sets water markets aside from other environmental markets is the complex nature of water that causes water markets to be riddled with imperfections: water is used for a wide variety of purposes, both public and private (e.g., for recreation in a public swimming pool versus for personal drinking); water use is in many instances sequential (via return flow and subsequent use); water is mobile but heavy and its transport governed largely by the path of existing waterways; water is often not observable (i.e., groundwater); and the water supply may be highly variable and uncertain (Young & Haveman, 1985; Hanemann, 2006). These characteristics imply that water use and trading potentially comes with many externalities and/or transaction costs, complicating water market transactions (Young, 1986). The challenge, therefore, is as Gardner (1990) argued, not to 'throw away the baby with the bath water'. In other words, the right conditions and institutions have to be created for water markets to enable their potential gains.

In this paper we assess the performance of individual markets on the basis of three common water management objectives: (1) limiting total water use from a particular source; (2) protecting water-dependent ecosystems or species; and (3) stimulating shifts in water use towards increased economic productivity. After discussing both successes and failures against these objectives, we present data describing the spatial and temporal complexities of markets that help to explain why some markets may not be fully accomplishing these objectives.

### 2.1. *Effectiveness in limiting water use*

Two of the water markets we studied offer insights into the ability of water markets to limit total water use. Water in the Northern Colorado Water Conservancy District (NCWCD) is almost entirely limited by annual deliveries of water through a trans-mountain water importation pipeline (Figure 3). In the

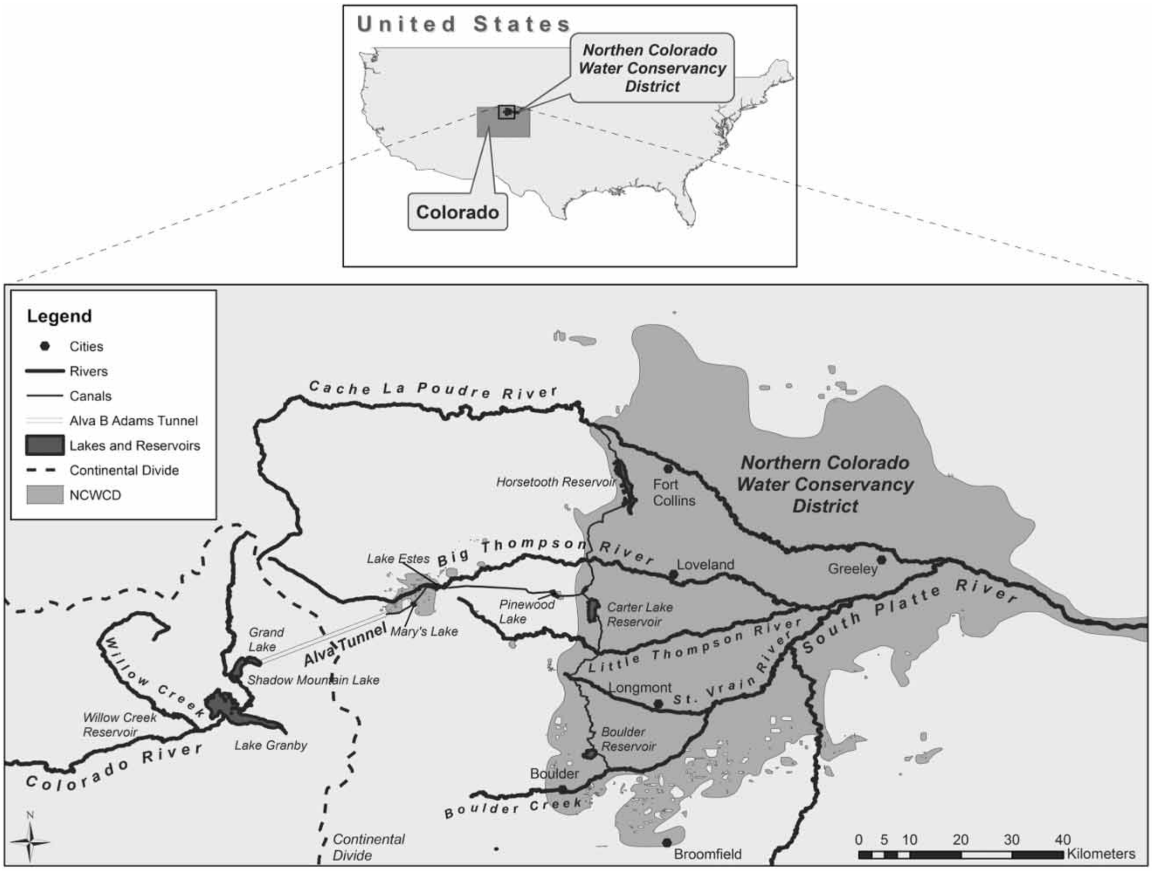


Fig. 3. The Northern Colorado Water Conservancy District (NCWCD).

Edwards Aquifer of Texas, imposed limitations on water use reflect both physical water availability and an interest in protecting natural ecosystems and endangered species. Ultimately, both systems rely upon regulatory control of the volume of water use through imposition of water allocations; in the NCWCD, those allocations vary from year to year, whereas in the Edwards Aquifer they have historically been based on permanent water rights.

The NCWCD is one of the most mature and longest operating water markets of the United States (Tyler, 1992; Griffin, 2006). This market emerged from development of the Colorado-Big Thompson (C-BT) Project, a federally-funded inter-mountain transfer of water from the Colorado River on the wetter western side of the Rocky Mountains to the drier eastern side. The project was originally constructed to provide supplemental water to farmers in the South Platte watershed, where locally-available water supplies had begun to be regularly exhausted as early as the 1890s.

The C-BT Project became fully operational in 1957 and is limited in capacity by the Colorado River water rights acquired for the project, amounting to a maximum of 380 million $m^3$ (310,000 acre-feet) per year, supplemented to a small degree by locally-derived runoff. The volume of water actually delivered to the NCWCD by the C-BT Project each year is conditioned according to the ‘seniority’ of the water rights acquired for the project. In years when water is abundantly available, the C-BT Project water rights are fully satisfied at 380 million $m^3$. However, in drier years, the volume of water delivered to the C-BT Project is limited by the need to first satisfy more senior, higher-priority water rights. The annual allocation of water to the C-BT Project is determined largely on the basis of snowpack data and projected runoff, as well as existing reservoir storage (NCWCD, 2013a). As a result, the average delivery of water has averaged 321 million $m^3$ (260,000 acre-feet).

The volume of water available to each user is then based on the number of 'units' (i.e., shares) held by the user – with each unit equaling one acre foot (1,233 $m^3$), but adjusted by an annual allocation that is based on the projected water delivery from the C-BT Project. Annual allocations have generally varied from 50–100% (Figure 4).

The cap on water use within the NCWCD is therefore an annually-fluctuating limit. Figure 4 illustrates the challenges of setting and managing annual allocations, even in a market system where water availability and use is measured and monitored with considerable effort. Annual water allocations are intended to maximize use of the available water from year to year, but the difficulties of predicting water availability result in differences between water allocations and availability in virtually every year. In some years, the actual water available is greater than what was allocated on the basis of runoff projections; in other years (e.g., 2005 and 2009), the system is over-allocated relative to actual availability. Similarly, total water use does not perfectly match the volume of allocations, or the volume of water available in that year. In some years, it is possible to use more water than is actually available in a given year by using carryover reservoir storage of C-BT water from the previous year. In other years, users do not utilize their full allocations because rainfall makes irrigation unnecessary.

We note with emphasis that in any regulated water allocation system, any differences – surpluses or deficits – between annual water availability and use will impact the local river or aquifer, other water

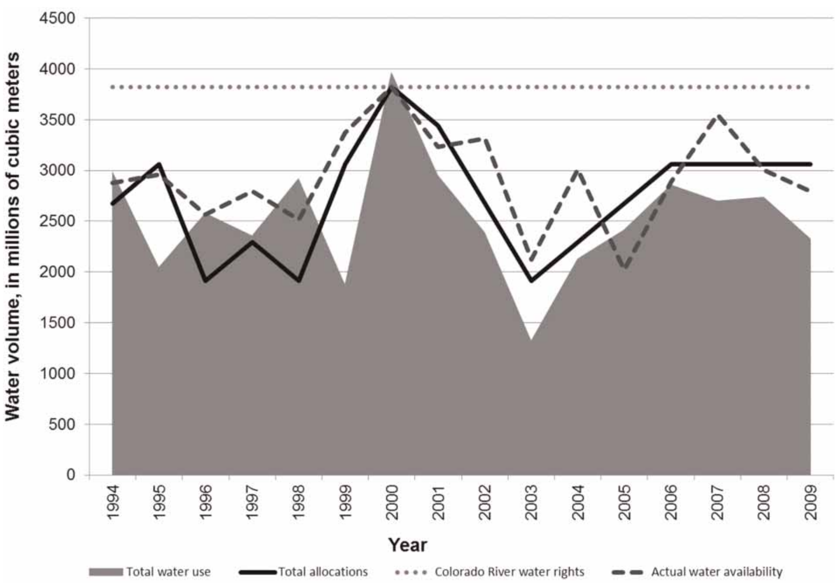


Fig. 4. Variations in water availability, allocation and use in the Northern Colorado Water Conservancy District (*Source:* NCWCD, 2007–2012).

users outside/downstream of the water system, or both.[1] The NCWCD experiences mainly surpluses, with excess water draining into the South Platte watershed, where it supplements natural river flows and becomes available for use by others downstream. In other regulated systems, water use in excess of the allocated volume depletes water sources beyond the intended level and may reduce water available to other users outside of the regulated system. For this reason, careful monitoring of water availability and regulation of use within market systems is of paramount importance.

In the Edwards Aquifer of Texas, limits on aquifer withdrawals have been expressly set for the purpose of protecting plant and animal species dependent on aquifer levels or outflows at springs. The aquifer is a water source for irrigation and also the primary drinking water source for more than two million people in central Texas, including rapidly growing San Antonio. A water market emerged in the late 1990s as a consequence of a federal lawsuit brought by environmental interests against the US Secretary of the Interior for failure to protect endangered species. As a result of the court case, a cap was established on total water extractions from the aquifer, as illustrated in Figure 1.

The Edwards Aquifer cap initially limited withdrawals to 550 million $m^3$ (450,000 acre-feet) per year by 2004, to be reduced to 493 million $m^3$ (400,000 acre-feet) by 2008. These caps drew on extensive scientific analysis of minimum aquifer levels and associated spring flows to support the endangered species.[2] As illustrated in Figure 1, the initial cap of 550 million $m^3$ was met during 2007, with only one minor exceedance (of 1%) in 2006. This was largely accomplished through aggressive implementation of water conservation measures in the San Antonio metropolitan area, investments in improved agricultural irrigation efficiency on regional farms (Richter *et al.*, 2013), and under-utilization of water rights.

In the case of the Edwards Aquifer there has been a substantial difference between the imposed cap levels and the total volume of issued water rights. After many legal challenges, 881 groundwater permits were issued over time, totaling 549,000 acre-feet (677 million $m^3$). This large disparity between the imposed cap levels and the volume of existing water rights posed considerable environmental risk, even though it consisted to a large extent of under-utilized water rights. To reduce the risk of full use of the rights, Texas could have bought down the excessive water rights to the level of the cap, but public funding for such purchases was limited.

Instead, Texas changed strategy. The Texas legislature, in close coordination with the US Fish & Wildlife Service, passed a bill in 2007 that relaxed the cap on total water rights to 705 million $m^3$ (572,000 acre-feet), a level 5% higher than either the existing volume of water rights or the maximum level of withdrawals in 1989 (Figure 1). At the same time, the Legislature shifted strategy from a cap on the volume of permanent water rights towards a cap on water allocations during drought periods (Texas State Legislature, 2007). A 'Critical Period Management' plan was put into place that reduces allowable withdrawals in a staged fashion during drought periods (Table 1; Figure 5). These reductions effectively create an allocation cap that, under certain drought scenarios, is considerably lower than originally set in the 1990s, as reflected by the fact that Texas legislators deemed it necessary to also include a floor on those reductions. From 2007, the allowable withdrawals were not to drop below 419 million $m^3$

[1] Matching water use to availability is also very challenging for water managers wanting to hit detailed environmental flow targets; see Dickens (2007) for South Africa.
[2] Some argue 400,000 acre-feet is still too high. The 1968 *Texas Water Plan* advised the 400,000 limit, acknowledging it might eliminate the flow of the Comal and San Marcos Springs some of the time. Later reports also recommended lower withdrawals (see Votteler, 2011).

Table 1. Critical period withdrawal reductions in Edwards Aquifer. Withdrawal reductions are tied to target levels of springflow at two major springs and aquifer levels at an index well. Reductions are triggered by any one or more of the stated conditions.

| Critical period stage | Comal springs flow ($m^3/s$) | San Marcos springs flow ($m^3/s$) | Index well level (meters, mean sea level) | Withdrawal reduction* (%) |
|---|---|---|---|---|
| I | <225 | <96 | <201 | 20 |
| II | <200 | <80 | <198 | 30 |
| III | <150 | | <195 | 35 |
| IV | <100 | | <192 | 40 |
| V** | <45 | | <190.5 | 44 |

*Reductions apply to the San Antonio portion of aquifer.
**Stage V was added with 2012 EARIP.
Adapted from: Texas State Legislature (2007) and EARIP (2012).

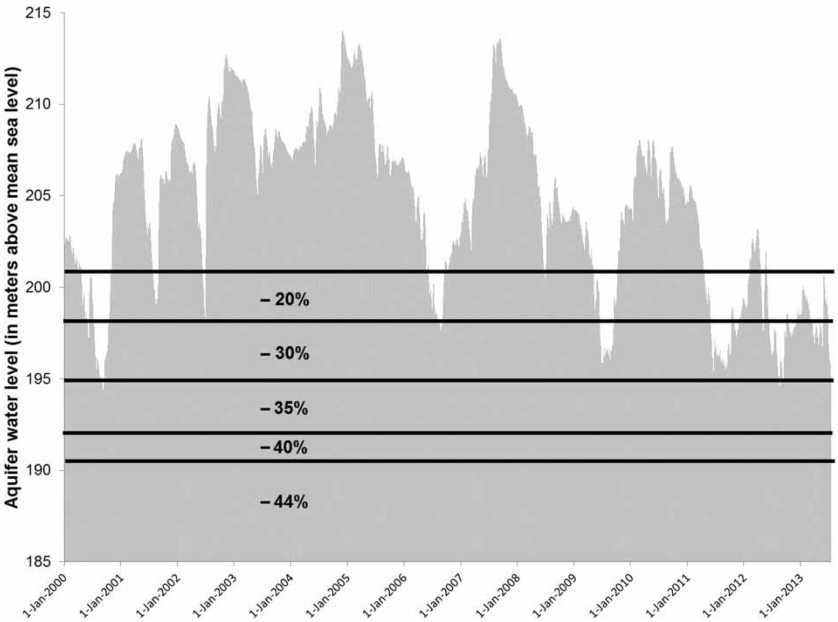


Fig. 5. Edwards Aquifer water level and regulatory thresholds. Shaded area indicates fluctuations in the aquifer level as measured at the J-17 index well since 2000. Regulatory reductions in water withdrawals (keyed to different aquifer levels) became effective at the end of 2012. The lowest recorded aquifer level occurred on 17 August 1956 at 186.7 m msl (*Source:* San Antonio Water System, 2013).

(340,000 acre-feet) under Stage IV conditions, and not below 395 million $m^3$ (320,000 acre-feet) beginning in 2013.

Importantly, the 2007 legislation also called for the creation of an 'Edwards Aquifer Recovery Implementation Programme' (EARIP, 2012) to further examine additional drought and habitat management measures that may be necessary to protect the endangered species. The EARIP produced a consensus-based water management plan in 2012 that is widely believed to be sufficient to sustain the endangered species, so long as the withdrawal reductions in Table 1 are achieved. It will be of interest to see whether the newest initiatives will reinforce or hamper the efficiency gains associated with water markets – a point discussed later in this paper.

In summary, water managers have been largely successful in managing water usage within cap limits in these two case studies; water use has exceeded the cap levels only occasionally. This was made much easier in the NCWCD because physical water availability (in C-BT Project deliveries) limits water use as much or more than the regulatory cap (allocation levels). In the Edwards Aquifer, water use has remained below cap levels primarily due to under-utilization of permanent water rights. With the capping strategy now shifting from a limit on the volume of permanent rights to a temporary limit on water use during droughts, it will be interesting to see whether the water managers of the Edwards Aquifer are able to maintain the aquifer at levels necessary to protect the endangered species.

### 2.2. *Effectiveness in protecting ecosystems and species*

A number of water markets have been initiated or utilized to achieve environmental restoration objectives. In some cases, the capping of water use – through either a limit on the volume of permanent water rights or by conditioning those water rights with variable annual allocations – is the primary mechanism to pursue environmental protection or restoration. In other cases, such capping is accompanied by a programme of 'buying down' the existing water rights, either through governmental buy-backs or through non-governmental water purchasing initiatives (e.g., by private conservation organizations). A buy-back programme is politically much more palatable than a regulatory reduction in water rights without compensation, which amounts to expropriation from water rights holders. Environmentally-driven acquisitions of water rights are expressly intended to make more water available in a freshwater ecosystem during critical seasons or droughts.

Acquisitions of water rights for environmental purposes have been made possible and increasingly common by an evolving legal doctrine that has recognized retaining water for in-stream use or even for off-stream use, such as for flooding, as a beneficial use. These legal adjustments have been important in preventing the loss of a water right or its seniority due to perceptions that the water right was not being used for any worthwhile purpose and should therefore be forfeited or re-allocated (Colby, 1990).

In the Edwards Aquifer of Texas, as discussed above, endangered species recovery plans are predicated on the ability to manage water withdrawals within prescribed limits and thereby leave enough water in the aquifer (or flowing in springs) to support targeted species. Even though the total volume of existing water rights remains 37% higher than the original cap level, it is believed that those rights can be adjusted through regulatory reductions in aquifer withdrawals during dry periods, such that endangered species and the overall health of aquifer and river ecosystems can be sustained (EARIP, 2012). This approach of withdrawal reductions has in fact been tested, in part, over the past decade by the San Antonio Water System (SAWS), the primary purveyor of water to San Antonio's metropolitan area and the dominant user of the groundwater. Since the late 1990s, SAWS has been implementing an aggressive water conservation programme that is keyed to specified water levels in the aquifer. As the aquifer level drops, increasing regulatory controls are exercised on water use in the metro area (SAWS, 2013). Monitoring of endangered species populations suggests a relatively steady or improving trend over recent years (Figure 6). Extensive ecological modeling conducted as part of the EARIP suggests that the aquifer management strategies already being employed by SAWS, to be supplemented substantially through EARIP recommendations, should sustain the species.

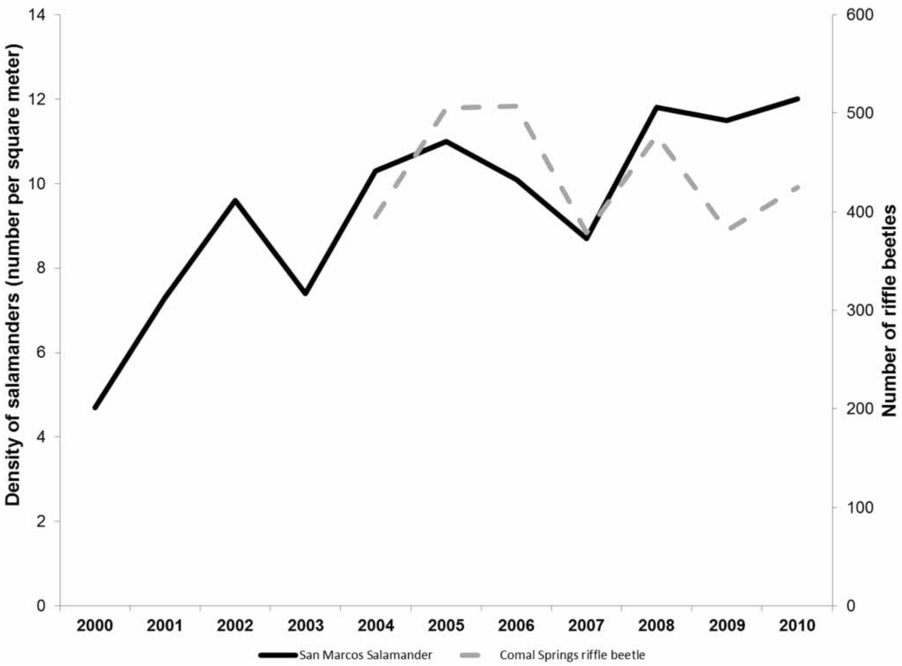


Fig. 6. Monitoring results for two endangered species (San Marcos salamanders and Comal Springs riffle beetles) dependent on the Edwards Aquifer (adapted from EARIP, 2012).

Similarly, capping the total volume of water rights as well as annual adjustments in water allocations have been used to protect and restore ecological health in the Murray–Darling Basin of southeastern Australia (Figure 7), where water is predominately used for irrigated agriculture. Australia's water markets are perhaps the most advanced and active water markets in the world. Australia has moved away from a strict engineering approach of water management towards economic principles and water markets (National Water Commission, 2011; Heberger, 2012). The recent 'Millennium Drought' that caught much of Australia in its grip between 1997 and 2009 helped develop and refine water markets that are concentrated in the Murray–Darling Basin, where the bulk of Australia's agricultural production takes place.

The Australian states sharing the basin stopped issuing new water rights as early as 1969, but a basin-wide cap on total water extractions was not adopted until 1995 and only formally institutionalized in 1997 (MDBA, 2011). A national water market was established soon thereafter. However, the cap in the Murray–Darling Basin was insufficient to prevent widespread ecological devastation when a drought of record began in 1997. With ecological and economic impacts mounting during the Millennium Drought, in 2007 the Federal Government passed a new Water Act calling for a basin-wide plan. A draft plan released in 2010 was based on an exhaustive assessment of more than 2,000 environmental assets (e.g., populations of target species, wetland habitats, etc.) spread across the basin, as well as more than 100 hydrologic indicator sites. The draft Basin Plan outlined three scenarios for reducing water diversions to protect ecological health, with reductions ranging from 27–37% (Murray–Darling Authority, 2010), but the final Basin Plan completed in 2012 set basin-wide reductions at 21% (a reduction of 2.8 billion ($2.8 \times 10^9$) $m^3$, or 2.2 million acre-feet), to be supplemented with a variety of water infrastructure modifications designed to facilitate environmental watering projects.

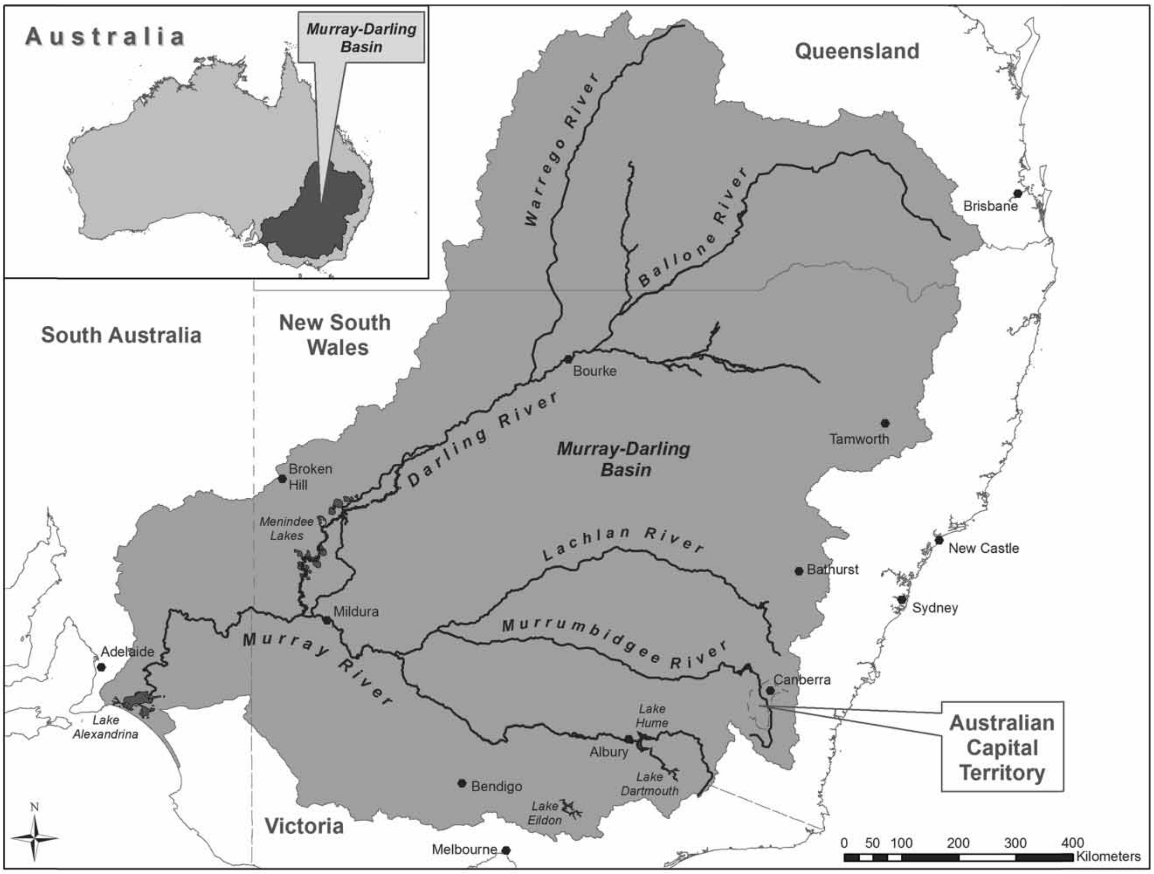


Fig. 7. Murray–Darling river basin of Australia.

The sequence of caps implemented by the states and by Federal Government in the Murray–Darling has been accompanied by commitments to buy back water rights to restore river flows and priority habitats. In 2003, AUS\$700 million was appropriated over a five-year period as part of a 'Living Murray Initiative' to purchase 960 million $m^3$ by 2009 (Australian Government, 2012). The federal government committed an additional \$3.1 billion ($3.1 \times 10^9$) for environmental water purchase in 2008 in its 'Restoring the balance' programme. As of September 2012, an additional 1.6 billion ($1.6 \times 10^9$) $m^3$ had been recovered, representing over half of the 2.8 billion ($2.8 \times 10^9$) $m^3$ of water use reductions called for in the final 2012 Basin Plan (Australian Government, 2012). The Australian case illustrates the challenge of reactively imposing a cap that *de facto* reduces water use permanently. It also brings to the fore the importance of a societal debate, as capping water use comes with governmental commitments of buying water rights to be paid for by taxpayer dollars.

The ecological benefits of the environmental water purchases in the Murray–Darling Basin are detailed in various reports (MDBA, 2011; Australian Government, 2012; National Water Commission, 2012, 2013). While considerable ecological benefit has clearly been achieved, given that the Basin Plan will not be fully implemented until 2019 it remains too early to tell whether the large suite of environmental targets identified in the Basin Plan will eventually be fully restored by the efforts to cap water consumption and buy back water rights for the environment.

### 2.3. Effectiveness in improving economic productivity

The benefits of competitive markets have long been appreciated, even before economists started analyzing their every aspect. In the context of our cap-and-trade approach, it is important to realize that the benefits that markets promise only materialize to the extent that water use is kept within an agreed-upon cap which takes into account environmental concerns and adverse third party effects. Effective water use cannot be greater than the total sum of (adjustable) water rights and, if so, governments and other agencies have to regulate or buy water rights down to the cap. Otherwise, negative externalities diminish the efficiency gains associated with water market transactions. Similarly, water markets can only be successful in addressing scarcity with proper governance (water metering and properly enforced water rights) and adequate competition. It is important to note here that a well-functioning market presupposes an holistic approach that defines an environmental cap for all sources of water. If it were the case that the cap on water use applied, for example, only to surface and not to groundwater, it is quite likely that attempts to limit the use of surface water would give way to the depletion of the other.

The main strength of competitive water markets derives from their pricing mechanism. As Hanemann (2006) points out, water is commonly subsidized in price, or priced at the service fee of delivering water. In the latter case, the price of water is what it costs to supply it to users – e.g., to build and maintain irrigation canals or distribution pipes, wells, dams or water treatment facilities – without accounting for supply or demand levels, externalities of water use, or the willingness of water users to pay for additional water. In other words, water users are often not paying the true cost of water. This underpricing leads to overuse, less conservation and resource depletion. Another serious problem in many water basins is the use of lots of water for low value purposes, to the exclusion of more economically productive uses.

An explicit goal of water markets advocates, then, is to let water markets price water more accurately and have them reallocate water resources to those who will use water more efficiently and maximize its economic value. The strength of markets to allocate water efficiently and to do so in a decentralized fashion appears most clearly when compared with traditional, non-market based regulations, or so-called 'command and control' policies. For instance, regulatory actions in droughts are common, calling for everyone to save $x$ litres of water. Because the cost of reducing water use differs considerably among water users, the ability to trade in water markets can decrease the economic cost of conservation significantly. Instead of incurring costly conservation, water users can buy water rights from those who save water easily and without much additional cost and who are willing to sell. In this light, it is important to ensure that any new water conservation regulations, such as those discussed above for the Edwards Aquifer, do not hamper the decentralized and efficiency-enhancing allocation of a cap-and-trade system.[3]

Figure 8 supplements Figure 2 in illustrating how markets reallocate water in such a way that it reaches the user whose income it increases most. Consistent with our focus on the cap-and-trade

[3] For carbon markets, researchers started studying the sometimes surprising interaction between regulations and a cap-and-trade system; see Goulder (2013).

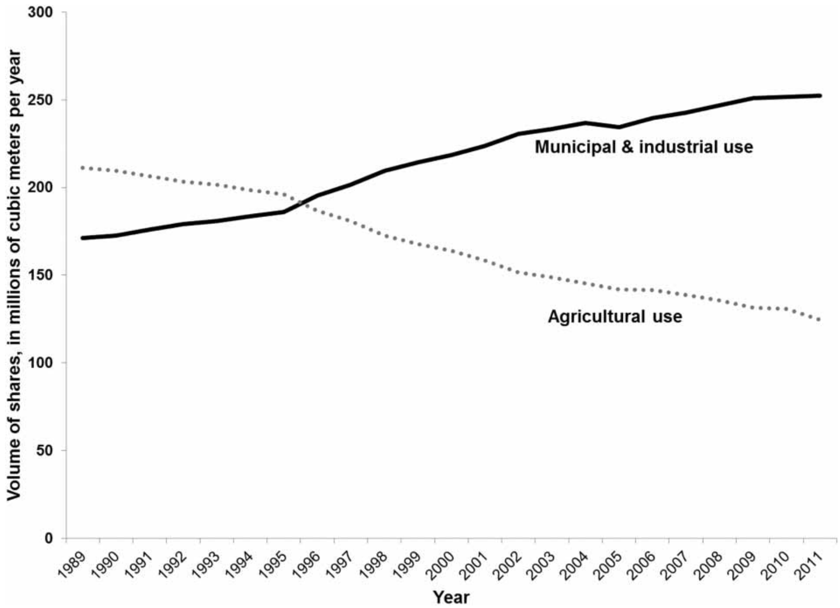


Fig. 8. Changing ownership structure of permanent water rights in the Northern Colorado Water Conservancy District (*Source:* NCWCD, 1997–2012).

aspect of water markets, we now assess the effectiveness of trade by examining the changing sectoral ownership structure of water rights in the Northern Colorado Water Conservancy District (NCWCD) since 1989. We focus on the net sales of permanent water rights between sectors rather than on the exchange of temporary rights (often referred to as leases) that eventually return to the original owner. Changes in permanent rights should best reflect the underlying, longer-run economic tradeoffs.

The Colorado-Big Thompson (CB-T) Project (Figure 3) involves an inter-basin water transfer from Colorado's west slope to its highly populated east slope. The NCWCD owns all rights to the transferred water. Three different types of shares have been allocated – agricultural, municipal and mixed – totalling 380 million $m^3$ (310,000 acre-feet). Mirroring the trend of the Edwards Aquifer, in Figure 8 we see a steady reallocation of water over roughly the last 20 years in the NCWCD away from agriculture towards municipal use. About 60,000 acre-feet, or about one fifth of all rights, have changed sector.

To illustrate and confirm that such inter-sectoral trades indeed enhance the efficient allocation of water is not an easy task. It would be highly desirable to directly document the differences in willingness to pay for additional water (and thus in the marginal value of water) between buyers in agriculture and in cities. However, beyond anecdotes of city dwellers willing to pay much more than farmers do, little systematic evidence exists.[4]

[4] According to Brewer *et al.* (2008), San Diego offered US$225 per acre-foot for water that cost farmers $15.50. Griffin & Boadu (1992) reported value differentials for water of 9–21 times between agriculture and cities. With US *Water Strategist* data, Brown (2006) and Brewer *et al.* (2008) found that prices across time and water basins for agriculture-to-agriculture trades were significantly lower than between agriculture and municipalities. Brewer *et al.* (2008) interpret this as evidence of sectoral differences in marginal productivity of water, assuming away the heterogeneity of trades; see Section 3: Key challenges of water markets.

Table 2. Water use and water productivity statistics for New South Wales, Australia.

| | Water consumption (megaliters, ML) | Gross value added (millions of Aus. dollars) | VA/WC (thousand $ per ML) | Employment (# of people) | Employment/WC (people per ML) |
|---|---|---|---|---|---|
| Agriculture, forestry, fishing | 2,896,397 | 7,226 | 2.49 | 73,600 | 0.03 |
| Mining | 77,544 | 13,032 | 168.06 | 36,500 | 0.47 |
| Manufacturing | 144,222 | 35,468 | 245.93 | 258,900 | 1.80 |
| Electricity, gas, water, waste | 995,250 | 11,788 | 11.84 | 44,700 | 0.04 |

*Sources:* Australian Bureau of Statistics (2011, 2012a, b), based on data for 2010–11.

Table 2 probably offers some of the best evidence at hand for why the observed inter-sectoral trades from agriculture to municipalities (or industries) might be an efficiency improvement as viewed through an economic lens. Table 2 lists data for Australia's Murray–Darling Basin on employment, value added, water consumption and water productivity for a number of sectors: (1) agriculture, forestry and fishing; (2) mining; (3) manufacturing; and (4) electricity, gas, water and waste treatment. These statistics illustrate the considerable inconsistency between the volume of water used in agriculture and its economic productivity or employment, as compared to other sectors. Water productivity as measured by employment per megalitre is 16 times higher in mining, and 60 times higher in manufacturing, than it is in agriculture. Measuring water productivity in terms of economic value added per megalitre of water, the contrast is even starker: mining is 67 times and manufacturing 99 times more productive than agriculture.

While water productivity numbers vary by the particular sectors or time periods considered, the order of magnitude of these numbers is in line with what we also observe in other river basins we have studied. The Lerma–Chapala–Santiago river basin of central Mexico is one of the country's most populous regions, and quite important to the country's agricultural and industrial production. Agriculture accounts for only 5% of the basin's GDP, but consumes 82% of its water (National Water Commission of Mexico, 2010). In other words, in the Lerma–Chapala–Santiago basin, agriculture uses 16.4 times more water per unit of value added as the rest of the economy.

Of course, the cited water productivity measures only capture average, not marginal, productivity. They do not tell us how much manufacturing income or employment are going up as we transfer water away from agriculture, nor do they reveal the significant variation in water productivity within agriculture and manufacturing or how other production factors such as labour, human capital, physical capital and land vary with water use. Still, these basic productivity measures are directly relevant in the dynamic context of the often contentious inter-sectoral redistributions of water. What all these data suggest is that the sectors other than agriculture can employ many more people and support much higher value creation than agriculture for the same amount of water used. A unit of water will support the employment of between 10 and 100 times as many people in municipalities than it will in agriculture.[5]

Water trading in the NCWCD illustrates the dynamic context of the reallocation of water. Over the last 20 years, roughly 60,000 acre-feet of water shares have been acquired by municipalities and industry

[5] Casual observation supports this conclusion: thousands of acres of irrigation for forages, grains, or cotton are required to support even a small town; see Young (1984).

from agricultural water users, and now the vast majority of shares are owned by sectors other than agriculture. For reference, in 1957, 95% of the water shares were allocated to agricultural irrigation (Hecox *et al.*, 2010). Consistent with this transfer of water rights, there has been a steady decline of employment in agriculture. The 1978 US Census for Agriculture records 22,480 people employed in agriculture for the relevant counties in the district (US Department of Commerce, 1989). This number declined to 14,951 in 1992, and to 13,768 in 2007. The picture for population growth in cities, on the other hand, tells a very different story. From 1950 to 2012, the population within district boundaries grew from 150,000 to 850,000, with most of this growth accounted for by an urban population boom (NCWCD, 2012). In other words, a relatively small decline in agricultural employment (less than 10,000 in the last 20 years) corresponds with an increase in the population, and by extension in employment, in the cities and industries that is in the hundreds of thousands.

This shift in rights among water use sectors is part of a common urbanization process that is driven by many factors other than water, such as forces of agglomeration, technological change, population growth, etc. The implicit message of such calculations, however, is clear. Because of the very significant differences in average water productivity between agriculture and the rest of the economy, a relatively moderate reduction in employment or value added in agriculture frees up enough water to support an increase in economic activity in cities and industries that is, in terms of employment and value added, at least ten times as large. This is not a new message, but it is one worth repeating because transfers between agriculture and cities are often controversial.[6] Needless to say, the economic gains only materialize when regions and countries can freely import and export food, and societies will have to decide whether such gains accord with their desire to protect local food sources or with their interest of protecting rural lifestyles. For these reasons, limits on the volume of agricultural water that can be transferred out of the sector are sometimes introduced, or exit taxes on sales of water from agriculture to other uses are charged.

In the margin of the inter-sectoral shift in water rights discussed here, a few additional caveats need to be raised. A change in ownership of permanent water rights does not necessarily mean that all water acquired is immediately used in non-agricultural activities. Many towns or cities rent some water back to agriculture on an annual basis (Howe & Goemans, 2003). In spite of this, and taking into account the current limited availability of comprehensive water data, the changing distribution of permanent rights is most likely the better guide to the longer-term sectoral shift, consistent with what economic efficiency predicts.

In the Murray–Darling Basin, as in other markets that we studied (e.g., NCWCD, Edwards Aquifer), far more trades are taking place within agriculture – among agricultural crops with varying water productivity – than between agriculture and manufacturing, whose productivity differences are more substantial. The fact that cities and agriculture are not always hydrologically connected explains some of this lack of water trading, but cities may also be reluctant to be seen buying water rights from farmers due to widespread perception that this disrupts rural communities. When cities do buy water from farmers, they often prefer to finance water conservation projects in agriculture instead and receive the saved water, rather than actively contribute to decreasing agricultural output and employment (Richter *et al.*, 2013). In addition, there is concern that by buying water rights, the cost of maintaining existing irrigation infrastructure may be left to the remaining farmers – the question

[6] Young (1984) is one of the earliest and still best articulated formulations of this message. Back then, he predicted that a 10–20% reduction in agricultural employment would be able to support water demands in cities between 1980 and 2020.

of stranded assets. While these concerns are real, an open question that ultimately has to be decided by societies themselves remains: should a dynamic market economy shelter agriculture from structural changes that improve overall economic efficiency and, if so, on what grounds? With economic development and international competition, some sectors wax and others wane. Moreover, as the NCWCD case illustrates, the trade-off is one of a relatively limited decrease in agricultural output or employment versus a sizeable increase in municipal and industrial employment and output. In sum, water markets enhance economic efficiency, and broader questions of equity and an equitable distribution of water rights should be part of a larger public debate.

## 3. Key challenges of water markets

Because of our cap-and-trade approach, we have so far focused on the implementation of a cap on water rights and how the changing distribution of water rights may reflect the more efficient allocation of water that economic theory suggests. In this last section, we analyze actual market transactions to draw out a few challenges that water markets face: the sometimes extremely localized nature of transactions, the high variability of prices, and the complex nature of water market transactions.

Before focusing on those challenges, the local context of water markets should be emphasized, which makes cross-market comparisons difficult, and caution should be taken in aggregating or comparing transaction volumes and prices across markets (not uncommon in the literature). The local context of water markets in no way implies that they are isolated from the regional, national or global economy. Water demand in California, for example, depends on the national and even global demand for its agricultural products. It does imply, however, that variation in the local conditions within the river basin also determine the prevailing market conditions – which is different, say, from oil, in that there is no integrated worldwide market for water. Hence, price differences between water markets among different countries, within countries, and even within a large river basin persist. During the week of 15 April 2013, the cost of a permanent water right in the Edwards Aquifer averaged US\$4.26 per $m^3$.[7] In the NCWCD, the cost was US\$19.70 (NCWCD, 2013b) and in the Murray–Darling Basin it was US\$1.20.[8] In stark contrast, crude oil was selling on the global market for \$528 per $m^3$. Because water is heavy and because transportation cost are high relative to water's value at the place of use, existing water price differences are not easily arbitraged away and, what is more important, price differences among water market regions do not necessarily reflect an inefficient allocation of water.

## 4. Geographical context matters

To compare the performance of water markets in different geographies is tricky. Not only do the environment and economic contexts vary but also the legal framework tends to differ. The regulation of water markets in the United States, for example, varies by state and sometimes by local entity. Even in Australia, where the responsibility for water management is more centralized, there is still significant variation in legal setup or definition of water rights across states (see for example the

[7] Price based on sales in Medina, Uvalde and Bexar counties (Edwards Aquifer Authority, 2013).
[8] Average sales prices from Waterfind (2013).

comparisons within the United States and Australia discussed in Grafton *et al.* (2012)). In this respect, Chile offers a relatively unusual case. The constitutional reform of 1980 and the National Water Code of 1981 created a nationwide framework that supported the emergence of water markets. Water was considered a natural resource for public use but the permanent right to use water could be granted to individuals. Moreover, water rights were recognized as private property independent from land ownership, and those water rights were allowed to be bought, sold, leased and mortgaged, just like private property, all across Chile (Hearne, 1998; Anderson *et al.,* 2012; Donoso, 2012). This uniform legal framework allows us to examine how regional variation in Chile's climate, its varying population density and industrial activity determine regional variation in market transactions (Figure 9; Table 3).

Limited trading occurs in Chile's lightly populated southern regions because in those wetter regions water is relatively abundant. Chile's five most northern regions and the more temperate regions immediately to the south are similar in population and economic weight. Even though water availability is quite limited in the north, few trades have taken place. A plausible explanation is that many of the available water rights are already owned by the high-value mining industry (Barrionuevo, 2009), and as a result there are fewer but more larger-volume and higher price transactions. Most transactions take place in Chile's central valley, where trading occurs between municipalities, industry and agriculture. It is here that we find many and varying demands for water in one location, along with an initial distribution of water rights that apparently deviates from the most economically-productive allocation of water use.

The existence of water trading suggests that buyers and sellers find water transfers beneficial. The volume and value measures of water trades are an indicator, albeit a fairly crude one, of potential gains of water trade. However, as our description of the varying local circumstances in Chile illustrates, the absence of or limited volume of water transactions does not necessarily imply that water is used inefficiently. To the extent that water is sufficiently abundant, there may not be a need for much water trading under current circumstances. And even when water is scarce, the initial distribution of water rights matters also. If the highest-value water users hold many water rights, there may be less need for water transactions. As Chile's central valley illustrates, water markets thrive and beneficial water trades take place under two conditions: first, that there is sufficient variation in the intensity of water demand across one and the same water basin; and second, that the distribution of water rights does not exactly map the distribution of water demand.

### *4.1. Localization of trading*

Table 4 documents the very strong localization in water markets and how it may impede an efficient allocation of water. The Edwards Aquifer runs under seven counties, with Bexar County the most urban county, which includes the city of San Antonio. The input–output Table 4 displays the trading pattern amongst the counties. The counties in the first column are the sellers and those in the first row the buyers. The first number of each cell reports the total trades (both purchases and leases) between 2000 and 2006. The second number in brackets reflects whether more or fewer trades occurred than would be expected if trading were frictionless and thus geographically random (positive values indicate more trades than expected, negative ones fewer trades than expected).[9] The data reveal a very strong

[9] Expected trades are obtained by multiplying the probability that a trade occurs between two counties by the total trades in the Edwards Aquifer from 2001–2006.


Fig. 9. Market trading regions in Chile.

Table 3. Water market regions in Chile.

| Region name (north to south) | Fig. 10 code | Population in 2012 | Population by percent (%) | Climate | GDP by region (%) | # of trades | Percent of trades (%) |
|---|---|---|---|---|---|---|---|
| Arica y Parinacota | XV | 213,595 | 1.29 | Desert | n/a | 1,203 | 2.30 |
| Tarapacá | I | 298,257 | 1.80 | Desert | 3.50 | 512 | 0.98 |
| Antofagasta | II | 542,504 | 3.27 | Desert | 6.10 | 450 | 0.86 |
| Atacama | III | 290,581 | 1.75 | Desert with winter rains | 1.90 | 504 | 0.96 |
| Coquimbo | IV | 704,908 | 4.25 | Dry with winter rains | 2.30 | 7,672 | 14.69 |
| Valparaiso | V | 1,723,547 | 10.40 | Temperate | 7.90 | 7,134 | 13.66 |
| Metropolitana | RM | 6,683,852 | 40.33 | Warm temperate | 43.30 | 14,390 | 27.55 |
| Libertador | VI | 872,510 | 5.26 | Temperate | 3.70 | 4,363 | 8.35 |
| Maule | VII | 963,618 | 5.81 | Warm temperate | 3.40 | 9,222 | 17.65 |
| Bio-Bio | VIII | 1,965,199 | 11.86 | Warm temperate | 9.00 | 4,809 | 9.21 |
| Araucanía | IX | 907,333 | 5.47 | Warm temperate | 2.40 | 1,136 | 2.17 |
| Los Rïos | XIV | 363,887 | 2.20 | Rainy temperate | n/a | 456 | 0.87 |
| Los Lagos | X | 785,169 | 4.74 | Warm temperate | 4.40 | 202 | 0.39 |
| Aisén del Gen D. C. | XI | 98,413 | 0.59 | Cold temperate | 0.60 | 158 | 0.30 |
| Magallanes | XII | 159,102 | 0.96 | Cold and wet | 1.20 | 24 | 0.05 |
| Totals | | 16,572,475 | | | | 52,235 | |

*Sources:* INE (2009); INE (2011); City Population Website (2013).

Table 4. Trades between counties in Edwards Aquifer (2000–2006) and (in brackets) the difference between observed and expected number of trades. Positive values indicate that more trades were observed than expected.

| Transferee → Transferor | Atascosa | Bexar | Comal | Guadalupe | Hays | Medina | Uvalde |
|---|---|---|---|---|---|---|---|
| Atascosa | 4 [4.0] | 3 [−0.4] | 0 [−0.4] | 0 [−0.1] | 0 [−0.4] | 0 [−1.8] | 0 [−1.0] |
| Bexar | 0 [−1.4] | 379 [156.5] | 14 [−9.2] | 8 [4.1] | 2 [−21.7] | 35 [−80.9] | 14 [−47.4] |
| Comal | 0 [−0.3] | 14 [−38.2] | 55 [49.6] | 4 [3.1] | 32 [26.4] | 1 [−26.2] | 0 [−14.4] |
| Guadalupe | 0 [0] | 0 [0] | 0 [0] | 0 [0] | 0 [0] | 0 [0] | 0 [0] |
| Hays | 0 [−0.1] | 0 [−22.2] | 3 [0.7] | 0 [−0.4] | 42 [39.6] | 0 [−11.5] | 0 [−6.1] |
| Medina | 0 [−1.9] | 252 [−51.2] | 10 [−21.6] | 1 [−4.3] | 4 [−28.3] | 329 [171.1] | 20 [−63.7] |
| Uvalde | 1 [−0.2} | 149 [−44.5] | 1 [−19.1] | 1 [−2.4] | 5 [−15.6] | 50 [−50.7] | 186 [132.6] |

*Source:* Edwards Aquifer Authority (2013).

local bias. Most transactions take place within a county, that is, along the diagonal of the table. Even though we find a similar pattern among the catchment areas in the Murray–Darling Basin, the extent of localization is particularly strong in Texas. This is unexpected given that, unlike buyers and sellers arranged along a linear river system, the spatial location of a buyer or seller using the same aquifer does not matter because they are sharing the same pool of water. Buyers and sellers live above the same aquifer in contiguous counties with limited variation in precipitation and temperature, and without fees punishing inter-county sales. Moreover, as Votteler (2011) and others have emphasized, the karst limestone aquifer is highly permeable, facilitating rapid water movements and minimizing local impacts of pumping.

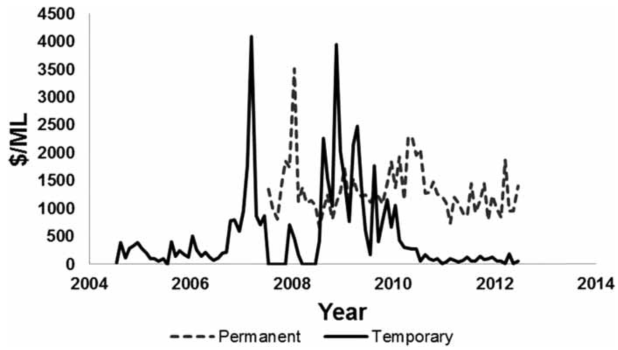


Fig. 10. Prices (in AUS$) for permanent and temporary water rights, Murray–Darling basin within New South Wales (*Source:* New South Wales Office of Water, 2013).

This highly localized nature of transactions does not necessarily imply inefficiencies. In a competitive environment, small transaction costs easily trigger localization. Anecdotal evidence from the Edwards Aquifer, however, suggests significant price variation for similar transactions, which is very suggestive of a lack of integration. A prime culprit for the insular market behaviour is a lack of transparency and information exchange that may directly challenge the markets' ability to allocate water efficiently. When information on water use and prices is readily available, water users can determine whether their own use is efficient or whether trading is desirable. Moreover, the lack of publicly available prices opens the door for bilateral, private deals that undercut competitive pricing. Unlike other commodity exchanges in which prices are readily and instantaneously reported through the internet, regulators of water markets do not always provide easy means to advertise one's interest in buying or selling. For example, while the Edwards Aquifer Authority does provide an online system for interested sellers to post notice of available water, prices are not listed. This lack of transparency also makes it very difficult to analyze market performance.[10] In the context of the Edwards Aquifer, cultural differences among farm irrigators and urban water suppliers may also play a role.[11] To the extent that water sales to other sectors or counties potentially negatively affect local communities, farmers may be reluctant to transfer water.

### *4.2. Temporal variability in water trading*

Temporal variations in water trades reveal additional challenges for markets. Figure 10 illustrates the variation in water prices for trading temporary and permanent rights in the Murray–Darling Basin within the Australian state of New South Wales.

On average, sales of permanent water rights that imply an ownership transfer of the water rights are higher priced than temporary rights. We also observe this pattern in the other basins considered in this paper, wherever data are publicly available. This regularity is in line with expectations, since the price of permanent rights should amount to the discounted value of the prices of the (expected) future temporary water rights. In addition, prices for temporary rights tend to be more volatile. In the case of the

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[10] Donoso (2012) attributes high standard deviation in Chilean water markets prices to a lack of integration.

[11] Kaiser & Philips (1998) surveyed potential buyers and sellers of water rights in Texas before effective trading started. The survey revealed a pronounced preference for trading among farmers and against inter-sectoral trades.

Murray–Darling Basin, for example, temporary prices range from virtually zero to over AUD $4000 per megalitre (~$3700 USD). The price fluctuation of permanent rights, on the other hand, has a narrower range. Temporary trades may occur after significant fixed costs have been incurred, which may account for why relatively high or low prices do materialize. To make sure a crop survives a growing season that is drier than expected, for example, farmers may have to buy additional (but more expensive) temporary rights. Alternatively, in a wetter than expected season, farmers can put unused water rights up for sale even when prices are already relatively low.[12]

What stands out is the extraordinary volatility of the (nominal) prices of water rights. While there is an expected cyclical pattern within a year, extraordinary variation occurs from year to year, with prices spiking during the years of Australia's notorious Millennium Drought which lasted from 1997–2009. Moreover, there is negative correlation for both the prices and the number of transactions of temporary leases with climatic measures such as precipitation (displayed) or deviation from the normal temperature. More water is traded during drier and hotter times and water prices tend to be higher during those periods as well. Such relationships between water trading and the climatic environment are quite important in light of recent writings about water scarcity. In spite of discussion about the stress on water coming from population growth and rising standards of living, variability in the local climatic environment seems to be a major factor behind variations in price and trade volume as population and GDP do not vary so extensively at high frequency. As one would expect, the link with current local conditions is less pronounced for permanent rights that incorporate expectations of water prices in future years, beyond the particular climatic conditions of a given year.[13]

In light of markets' overall objective to improve the efficient allocation of water, our analysis in Figure 10 presents an extraordinary challenge. The highly variable water prices reflect the uncertainty and rapidly changing circumstances in water markets. The changing water prices represent a constantly changing benchmark with which buyers and sellers have to compare the expected returns of their own water uses in order to decide to buy or to sell. As if this were not challenging enough, there are also multiple types of rights that can be bought or sold. In the Australian markets, for example, there are, in addition to permanent and temporary rights, also high-security and general-security rights that come with substantially differing expectations of allocation, as well as options to buy/sell water rights. There is no question that this expanded set of instruments allows for better fine-tuning of the allocation decision and an improved reduction of the risk of, say, a water shortfall. At the same time, the multiplicity of instruments requires a non-negligible degree of financial sophistication and literacy to make the optimal decision and make water markets work.

[12] Interviews by Brian Richter with farmers in Australia suggest that trading temporary rights is often related to short-term decisions, i.e., whether there is excess water versus an unanticipated shortage of water. The National Water Commission in Australia also concluded that water trading facilitated a higher level of water use during water-scarce periods, enabling irrigators to meet their full water needs by purchasing additional water, thereby serving as a hedge against water scarcity (Commonwealth of Australia, 2012).

[13] There is tremendous heterogeneity in water rights across regions. The price of acquiring water for a given duration should vary by date, location and buyer. Most statements about water prices in the literature, our own included, do, for lack of data, not control for all sources of heterogeneity. Similarly there are imprecisions in how trade volumes are reported. Permanent/multi-year trades are often underrepresented because the volume is often only reported for the initial year. Brewer *et al.* (2008) indicate that the only comprehensive data for US water trades, the *Water Strategist,* suffers from this flaw.

## 5. Conclusion: policy implications

Water markets are attracting increasingly more attention in the fight against water scarcity and for sustainable water use. Water markets hold promise in that they can help us move towards more efficient and sustainable water use. It is important to keep in mind, however, that while well-functioning markets have a proven record, it takes a lot to make a market for water rights function well. Therefore, water markets will rarely be the only policy tool to use. There are significant logistical hurdles that have to be cleared for markets to be successful, the most important of which are: the buyer and seller have to be hydrologically connected; they must have access to all relevant information pertaining to the water rights that are traded; the impacts on third parties have to be taken into account; the market instruments and different types of rights have to be transparent, and the market participants have to be literate enough to assess the uncertainty involved and to choose the most appropriate market instruments. Finally, and perhaps most importantly, water markets only function properly when a cap on the issued water rights is agreed upon. Imposing a cap is the primary way through which environmental protection will take place. Needless to say, the process of assigning water rights and of implementing an environmental cap has a political dimension. Therefore, as the Australian example shows clearly, implementing water markets and adjusting the cap on water use requires a broad societal debate.

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