

Substituting Freshwater: Can Ocean Desalination and Water Recycling Capacities Substitute for Groundwater Depletion in California?

ACCEPTED VERSION

Authors: Pierre Badiuzzaman^{a, b, c}, Eoin McLaughlin^a, Darren McCauley^a

- ^{a)} Department of Geography and Sustainable Development, University of St. Andrews, Irvine Building, St Andrews, KY16 9AL, Scotland, United Kingdom
- ^{b)} Present Address: 5a Mount Street, Taunton, TA1 3QB, United Kingdom
- ^{c)} First and Corresponding Author: pierre_badiuzzaman@outlook.com

Highlights

- Substitutability of natural resources is significant for sustainability monitoring.
- This case study sheds new light on water substitutability in California.
- Water recycling and ocean desalination are subject to socio-economic limitations.
- Demand-side measures can strongly support water substitutability in California.
- Social and institutional capital are pivotal in facilitating water substitutability.

Abstract

While the sustainability of resource depletion is a longstanding environmental concern, wider attention has recently been given to growing water scarcity and groundwater depletion. This study seeks to test the substitutability assumption embedded in weak sustainability indicators using a case study of Californian water supply. The volume of groundwater depletion is used as a proxy for unsustainable water consumption, and defined by synthesising existing research estimates into low, medium and high depletion baselines. These are compared against projected supply increases from ocean desalination and water recycling by 2035, to determine whether new, drought-proof water sources can substitute for currently unsustainable groundwater consumption. Results show that maximum projected supply of new water, 2.47 million acre-feet per year (MAF/yr), is sufficient to meet low depletion estimates of 2.02MAF/yr, but fails to come near the high baseline of 3.58MAF/yr. This does not necessarily indicate physical limitations of substitutability, but more so socio-economic limitations influenced by high comparative costs. By including capacities in demand-substitutability via urban water conservation, maximum predicted capacities reach 5.57MAF/yr, indicating wide room for substitution. Based on these results, investment in social and institutional capital is an important factor to enhance demand-side substitutability of water and other natural resources, which has been somewhat neglected by the literature on the substitutability of natural resources.

KEYWORDS: Substitutability; Weak Sustainability; Water; Desalination; Water Recycling; California.

1 Introduction

Major international development agencies have sought to expand the current conceptualisation of national accounts to include wider measures of wealth that are important for monitoring sustainability prospects (World Bank 2011; UNU and UNEP 2012, 2014). Both the World Bank and UNI agree, that the environment i.e. natural capital has been particularly neglected and needs to be included in total wealth accounts of nations (*Ibid.*).¹ Within their natural capital stocks, however, neither accounts for water resources although they constitute an “essential factor” in most economic activity (Perry 2012, p.216; Gleick 2001). The availability of freshwater forms an irreplaceable foundation for human life, ecosystem health and civilizational prosperity.

¹ The UN Inclusive Wealth Report additionally highlights the overarching importance of health for total wealth (UNU and UNEP 2012, 2014).

Predicted increase of regional water scarcity is a key challenge of the 21st century, likely to impose adverse effects on agricultural production, food security and a variety of economic activities (Savenije 2002; Postel 2000; Cooley *et al.* 2014; Seckler *et al.* 1999; IPCC 2014; DWR 2008; Rijsberman 2006; Famiglietti *et al.* 2011).

In the context of increasing scarcity, the question of substitutability, i.e. the ease with which to replace one resource with another, figures prominently in the economic debate of sustainability, which will be discussed in the theoretical context section (Neumayer 2013; Elkins 2002). Empirical work on the substitutability of water is very scarce regardless of its policy relevance. Measuring the economic value of water is a major challenge, making it seemingly impossible to 'test' its substitutability quantitatively (Drupps 2015; Atkinson *et al.* 2012). Therefore, this article seeks to shed new light on the matter by using a case study to assess feasible water substitution capacities within the State of California.

California presents an interesting and socially relevant case because it provides "common" water scarcity challenges faced by arid regions and "critical" conditions to analyse substitutability, given availability of data and implementation of new supply technologies (Yin 2014, p.50ff; IPCC 2014; UN 2012; Cooley *et al.* 2014; Seckler *et al.* 1999). Ongoing groundwater overdraft and adverse, climate change induced effects on water availability will likely exercise severe pressure on the State's water resources and its ability to sustain tremendous population growth, which is projected to rise from 38.4 to 51 million by 2050 (DWR 2014c, p.4; Famiglietti *et al.* 2011). With its main water resources already exploited to their ecological and physical limits, California seeks "state-wide water supply reliability and sustainability" (DWR2014b, p.9-5; Gleick and Palaniappan, 2010). Among other measures, water suppliers are legally mandated to evaluate desalination and recycling as options to meet the goals of their water resource management plans (*Ibid*; Cooley and Ajami 2014; USBR 2012).

Responding to the question, whether we can "supply our way out of scarcity?", this article analyses new water supply capacities from ocean desalination and water recycling to determine whether current water consumption can be sustained (Zetland 2014a, p.11). This research seeks to answer whether predicted capacities from those two sources can provide sufficient quantities of freshwater by 2035 to substitute for unsustainable groundwater depletion in California?

To provide a theoretical and analytical framework for the case study, section 2 outlines the discussion on the substitutability of natural resources and water. Methodological assumptions are stated in section 3. Section 4 contextualises the case study, compares Californian groundwater

depletion with water supply capacities of ocean desalination and water recycling, before introducing the impact of demand-side options and presenting socio-economic cost considerations. Finally, the discussion assesses the results and highlights the importance of social/institutional capital for water substitutability.

2 Theoretical Context: Water Substitutability

2.1 The Substitutability Assumption and Limits to Substitution

Economic thinking about sustainability focuses on accumulating and managing total wealth efficiently to ensure optimal consumption and welfare into the future (Barbier 2011, Hanley *et al* 2015, Arrow *et al* 2012).² Using total wealth as an indicator for sustainability implies optimal resource allocation and unlimited substitutability in monetary terms between different forms of capital. These form underlying assumptions within the economic model of weak sustainability (Pearce *et al.* 1989; Hanley *et al.* 2015; Hamilton and Hepburn 2014). The adequacy of these assumptions fundamentally depends on adequate monetary valuation or pricing of a good to indicate scarcity, the rate of technical progress and the possibilities of substitution between forms of capital (Lecomber 1975; Neumayer 2013; Hediger 2006; Hanley *et al.* 2013).

The relevance of the substitutability of natural capital was highlighted in the debate on Climate Change mitigation. It showed that different views often arise from economists focusing on substitution at the margin (in monetary terms) while most natural scientists assess ultimate physical limitations (Fenichel and Zhao 2014; Heal 2009; Drupps 2015; IPCC 2014).³ Economically, substitutability can be measured as the elasticity of substitution, which “captures the ease with which a decline in one input can be compensated by an increase in another, while holding output constant” (Markandya and Pedroso-Galinato 2007, p.298). Empirical studies of elasticities of substitution between K_P and K_N are scarce and build on “non-falsifiable beliefs” about technical progress and future substitution possibilities (Neumayer 2013, p.192f.; *Ibid*; Atkinsons *et al.* 2012; Dietz and Maddison 2009; Drupps 2015). In theory, proponents of weak sustainability (WS) assume that economic scarcity leads to price increases, which result in 4 different effects/propositions that support the substitutability of various capital forms, as outlined by Neumayer (2013) (See Appendix A.1 for detailed description):

² Total wealth is the sum of different capital stocks such as produced capital (K_P); natural capital (K_N); human capital (K_H); social capital and institutional capital (K_S) and intangible capital (Hamilton and Hepburn 2014).

³Substitutability at the margin means analysing substitutability for each incremental unit.

1. Scarcity makes substitutability with another resource economically viable due to its comparatively lower cost.
2. Prices signal economic scarcity and drive dynamic markets to adapt towards efficiency under new scarcity conditions.
3. Natural resources are substituted with produced capital if the elasticity of substitution is greater or equal to 1.
4. Technical progress affects substitutability through efficiency gains and via cheaper production techniques, which increase the economically available stock of less profitable resources.

This framework of arguments does not mention institutional or social capital, despite their great importance to overall wealth and their ability to improve factor productivity. Institutional and social capital can support intensive and structurally driven growth without further increasing natural resource use, thus potentially enhancing the substitutability of K_N (Hamilton and Hepburn 2014; Hamilton and Liu 2014; North 1990). The absence in the analysed literature is surprising, considering that intangible capital, which is assumed to be mostly social and institutional capital, accounted for 29% of comprehensive wealth in the USA in 2005 (World Bank 2011). The intangible character makes quantification difficult, but does not justify complete omission (Hamilton and Liu 2014; Putnam 2001). While institutional capital includes the capacity and effectiveness of legislative rules and institutions; social capital refers to local cooperation, trust, networks, and societal norms (Bottrill and Pressey 2012; Hearne 2007; North 1990; Lee *et al.* 2011).

2.2 Water and Substitutability

2.2.1 Water characteristics and usage types

Hydrologically, water has both renewable and non-renewable resource characteristics. The main renewable water components are river runoff and the groundwater inflow into rivers. Their flow rate determines the limits of water provision and indicates scarcity (Shiklomanov 2000; Perry 2012). Abstracting the total amount of water replenished in a watershed each year is termed 'peak renewable water' and severely damaging to ecosystems (Gleeson *et al.* 2012; Wilson and Carpenter 1999). According to Gleick and Palaniappan (2010), 'peak ecological water' would be

the maximum abstraction which avoids uneconomic ecosystem damages.

Non-renewable resources such as lakes, reservoirs, groundwater aquifers or mountain snow-pack are physically limited by their stock, which changes depending on in- and outflows. With recharge rates of up to 1500 years, some of these are “effectively non-renewable” (Gleeson *et al.*, 2010, p.379; *Ibid.*).⁴ The point of maximum abstraction in spite of greater costs is termed ‘Peak non-renewable water’ (Gleick and Palaniappan 2010).

Assessing water use and management strategies, it is important to distinguish between consumptive uses, which remove water from the watershed, and non-consumptive usages. The latter allows water to be reused downstream and includes for example domestic consumption, navigation, fisheries and hydro-power (Perry 2012; Gleick and Palaniappan, 2010).

2.2.2 Water Substitutability

It is a common conception that water is essential and does not have any substitutes (Rogers and Leal 2010, p.2; Barlow and Clarke 2002). Two types of responses can be identified in the literature. Fenichel and Zhao (2014) argue that although “the last drop of water, ... is undoubtedly essential”, marginal “drops” used for cleaning side-walks or irrigating crops can be substituted with brooms or alternative crops and irrigation technologies (p. 350). This demand-side perspective highlights that changes in the drivers of water consumption can reduce water demand by shifting it from low-value to high-value uses. It does not answer, however, whether existing water consumption patterns can be sustained by augmenting supply from alternative sources, which is one aim of this study and the second type of response.

Gleick and Palaniappan (2010) suggest different stages of water supply production, which increase with scarcity. They illustrate how demand for water from various sources increases incrementally until it reaches a point of “maximum cost-effective extraction of surface and groundwater”. When demand moves beyond this point, supply shifts to higher priced technologies like water transfers or desalination to meet demand, ultimately reaching a backstop price for water (*Ibid.* p.11157). Applying Nordhaus’ backstop technology concept to water, the authors point out:

“The ultimate water backstop is still water, from an essentially unlimited source — for example, desalination of ocean water. The amount of water in the oceans that humans can use is limited only by how much we are willing to pay to remove salts and transport it to the point of use, and by the environmental constraints of using it” (*Ibid.*, p.11157).

⁴ Fossil water is non-renewable groundwater that “entered the aquifer as recharge in past geologic periods” and which is not replenished through annual runoff (Pereira *et al.* 2009, p.136).

This example shows that water substitution through another resource (Neumayer's proposition 1) is possible, if one water source (groundwater) is replaced by another (desalinated seawater). Practically this involves substitution via produced capital and technical progress (propositions 3 and 4) because new supplies require large infrastructure investments (pipes, pumps, and treatment plants), labour, and technological innovation to increase economically available supplies (Neumayer 2013).⁵

The two major limiting factors of water substitutability outlined are provision costs and environmental impacts. More sceptical authors stress that substitutes may fare worse than ecosystem services in terms of resilience, life-span, cost-effectiveness and suitability. They underscore ultimate, but not marginal substitution limits (Brauman *et al.* 2007, p.81). Contrarily, the existence of a backstop technology would suggest that water can be created sufficiently without dependence on the hydrological cycle. These diverging views form the basis for the analysis, which will examine the ability of ocean desalination and water recycling to meet unsustainable groundwater supply rates in California. This article thus pursues an identified need for resource specific, empirical analysis of substitutability for water, whose availability significantly affects societal prosperity and well-being (Perry 2012, p.216).

3 Methodology

3.1 Research Design

The analysis seeks to operationalise the theoretical concept of resource substitutability by constructing a case study that compares the production capacities of two advanced water supply sources with groundwater depletion, which signifies unsustainable water use. The analysis synthesises the results of three key studies on groundwater storage change in the Central Valley and compares them against current supply predictions within government reports, policy studies and relevant literature on water supply.

To assess the applicability of the substitutability assumption, it is advisable to focus on one specific resource (Sterner and Persson 2008). Limited substitutability is generally more likely observed at the macro scale as it becomes increasingly difficult to import a resource externally (Drupps 2015; Stern 1997). Although California is part of a federal structure, the closest State

⁵ It must be noted that both proposition 3 and 4, involve also demand-side effects supporting substitutability, which are not considered here.

with surplus water capacities is Alaska. Since water is a regionally available good, whose transportation cost from these areas are widely considered excessively high, California presents a regionally limited system with a finite supply of surface and ground water (USBR 2012; Hogdes et al. 2014; Stockton 2015). Additionally, existing availability of data strongly encourages analysis at the State level. These factors make a State-focused case study a suitable research design to evaluate the practical feasibility of the WS theory.

To analyse supply side changes, water demand-effects such as water use efficiency gains or population growth are initially excluded from the analysis. Being faced with general inadequacy of water prices as signals of scarcity, the study utilises biophysical, rather than monetary data to analyse substitutability (World Bank 2011; Zetland 2014a; Equinox 2009). Monetary valuation being another key assumption of the weak sustainability framework, thus limits the scope of the results. However, monetary costs of providing the determined quantities will be presented to estimate economic feasibility. Combining ecological (biophysical) and economic (monetary) units, this research contributes towards “bridging” economic and ecological notions on substitutability (cp. Fenichel and Zhao 2014, p.350).

3.2 Estimating Unsustainable Groundwater Use

To establish a baseline against which to compare the predicted quantity from substitution water sources, the case study first aims to determine the quantity of unsustainable groundwater use. Groundwater depletion serves as a good proxy for unsustainable water use, not only because of the “widespread depletion” in California, but because its long-term overdraft reduces water availability for future generations and damages dependent aquatic and terrestrial ecosystems (Cooley *et al.* 2014, p.2; Gleick and Palaniappan 2010; Scanlon *et al.* 2012; Faunt 2009).⁶

A sustainable abstraction or pumping rate (P_s) requires long-term groundwater storage change to be zero, which means that abstraction plus total discharge (natural D_0 + induced dD_0) must equal total recharge (natural R_0 + induced dR_0) (Zhou 2009; Scanlon *et al.* 2012, p.9322; Gleeson 2012):

$$R_0 + dR_0 = P_s + (D_0 + dD_0)$$

⁶Contrary to groundwater stocks, total quantity of surface water storage remained almost constant between 2001 and 2010 because precipitation in ‘wet’ years filled storage quicker than groundwater, suggesting that groundwater is a better indicator for unsustainable water use (DWR 2014a, p.3-33; Faunt 2009).

The sustainable pumping rate is used here as a minimum requirement to sustain groundwater usage. In practice, avoidance of intolerable environmental, economic, and social consequences should also be accounted for (Zhou 2009; Devlin and Sophocleous 2005; Alley *et al.* 1999).

In California, 74% of groundwater abstraction is concentrated in the overcharged CV, and 20% the Coastal aquifer system, where three basins in the Central Coast are “being subject to critical conditions of overdraft” (DWR 2015a, p.30; DWR 2014b; Cooley *et al.* 2014; DWR 2014a; Maupin 2005; Gleeson *et al.* 2012, compare Table A.1). Because the CV is the only aquifer in California with substantial monitoring and data availability, its results will be extrapolated by applying them to the quantities in the Central Coast (Scanlon *et al.* 2012, p.9320).

Quantitative data from three different studies of the Central Valley (CV) are compiled and summarised in Table 1 to determine storage change. Based on 42-years of historic water level data from the US Geological Society (USGS), Faunt’s (2009) hydrologic modelling study is important to estimate long-term groundwater trends. Observations periods for unsustainable use should be at least 40-50 years because short-term depletion is an acceptable buffer practice against droughts, if recharged thereafter (DWR 2015b; Pereira *et al.* 2009). California’s Department of Water Resources (2015) collected groundwater elevation data from various monitoring wells, while Famiglietti *et al.* (2011) utilised satellite data from NASA’s Gravity Recovery and Climate Experiment (GRACE) which measures total water storage change.

Variable recharge rates; unrecorded, private well pumping; and a natural time-lag of aquifer storage cause uncertainties in the underlying data, which nevertheless belongs to the best estimates worldwide (Pereira *et al.* 2009; McMahon *et al.* 2011; Faunt 2009).

3.3 Estimating Water Supply Prospects of Desalination and Water Recycling

Presented capacity projections are based on policy or water supplier studies, plans and project proposals. They represent current socio-political, environmental and economic considerations of feasibility, rather than theoretical and biophysical supply capacities. Aiming for the longest assessment period with sufficient data availability, the analysis looks at supply capacity predictions until 2035.

Desalination is defined as “the removal of salts from water to produce a water of lesser salinity than the source water”, with ocean water having a salinity between 3% and 5% that is reduced to below 0.1% (DWR 2014b, p.10-6). Treatment of brackish water of lower salinity is not considered here due to its limited supply capacity. Reverse Osmosis (RO) is currently the most relevant

technology considered in California (*Ibid*; Fritzmann *et al.* 2007). Municipal water recycling can be categorized into direct potable use as drinking water; indirect potable use via recharging groundwater or augmenting reservoirs; and non-potable water for non-drinking purposes like irrigation, which requires a separate piping distribution network. Because upstream recycling reduces downstream water availability, increase in ‘new water’, rather than total capacity, will be used as the appropriate measure to determine additional supply capacities (DWR 2003).

3.4 Limitations

The nature of the question and the absence of reliable long-term estimates (50-100 years presumably) for water supply capacities limit the capacity to analyse water supply substitutability to reasonable estimates within limited timeframes (Neumayer 2013). To accommodate for estimated future water declines, the impacts of climate change inform the selection of groundwater depletion baselines. Climate Change effects on water availability, however, are not comprehensively modelled to include impacts on sea level rise. Neither are water quality issues or dynamic ecosystem effects such as changing recharge quantities considered (DWR 2008). Both would exceed this research’s scope, as would considering the potentials of other management options such as water banking, conjunctive water management and surface storage (cp. Scanlon *et al.* 2012; Faunt 2009). Using aggregated data, this study assumes that new water supply in coastal regions can translate into greater availability in the Central Valley (DWR 2014a). Given the nature of the case study, findings are not generalizable, although they hope to contribute to a better understanding of the practical substitutability of water.

4 Case Study

4.1 Californian Water Context

California is the most populous US American State (28.7million) with a nominal GDP of \$ 2.31trillion (2014), comparable to Brazil, the 7th largest economy in the World (CADOF 2015). California is also the USA’s greatest water user, accounting for 11% of US total use and 16% of US groundwater withdrawals (Maupin 2010). Human water appropriation of approximately 43 million acre-feet per year (MAF/yr) is used to 20% for urban uses and 80% for agricultural uses. (DWR 2015a, 2014b). Despite its importance in US fruit, nuts, vegetables and dairy production, agriculture contributes only 2% to the State’s GDP and 4% to employment (Hanak *et al.* 2012, p.1; USDA 2014). The major water users are the irrigated agricultural regions, the Central Valley (CV) and the Imperial Valley, as well the growing population centres Los Angeles, Orange County and

San Diego. Thus, 75% of state-wide water demand is in Southern California, while roughly 75% of annual precipitation to recharge river runoff and groundwater is available in the Northern California. (DWR 2014b; Rogers and Leal 2010; Hanak *et al.* 2012, p.5).

The State's water distribution infrastructure comprises seven aqueducts, most importantly the State Water Project (SWP) and the Central Valley Water Project (CVWP), which transport water from the Sacramento-San Joaquin Delta (the Delta) to Southern California and the CV. Transportation to Southern California creates supply vulnerability and high energy costs (LAEDC 2008, p.4). The two other, major water source are the Colorado River, supplying the Imperial Valley and the South Coast, and groundwater abstraction, comprising on 38% of total water use with particularly high shares of 46% during dry periods (Hogdes *et al.* 2014; DWR 2014b). California's complex surface water rights system is based on timing of appropriation ("first-in-time, first-in-right") and neighbourhood to a water body ("riparian right"). Groundwater can be regulated locally or not at all, which complicates cooperative agreements and conjunctive management of both (CWC 2014).

Overdraft has exhibited severe ecosystem damages from loss of streamflow and wetlands in the Delta and the Colorado River; aquifer compaction, up to 9m of subsidence and consequent infrastructural damages in the Tulare basin; salt water intrusion in coastal aquifers; and social conflict among water users in the Central Valley (CV) (Gleeson *et al.* 2010, p.378; Scanlon *et al.* 2012; Wada *et al.* 2010; Barlow and Clarke 2002; World Bank 2015).

Water is managed by over thousand mostly public, but also private agencies (Federal, State and local), with the Californian Department of Water Resources (DWR) as the State level authority and the US Bureau of Reclamation as the federal authority to ensure water supply, (Hanak *et al.* 2012). Pursuing an Integrated water management approach, California aims for a "diversified portfolio of water strategies" (DWR 2014b, p.1-5). Desalination and wastewater recycling were identified as important supply sources in terms of creating local of "new", i.e. additional, water supply; greater "overall supply reliability" from diversification; and more long-term reliability from droughts (DWR 2014b, p.10-33; USBR 2012).⁷

⁷ High reliability and technical feasibility risks, as well as transportation costs, associated with bulk imports from an envisioned Alaskan submarine pipeline or from water bags towed behind tankers, as well as more exotic options like weather modification via silver particles, disqualify these options for this analysis (USBR 2012; Hogdes *et al.* 2014; Gleick 2010; Stockton 2015).

4.2 Groundwater Depletion in the Central Valley Aquifer

The long-term USGS hydrologic modelling study calculates an average total storage decrease of 1.4MAF/yr between 1962 and 2003 (Faunt, 2009).⁸ The DWR estimates average storage depletion to lie between 1.08 and 2.62MAF/yr from 2005 to 2010 (2015a, p.2). Over a 6.5-year study period, Famiglietti *et al.* (2011), estimate an average annual loss of 2.53MAF in total water storage, which is comparable to the upper range of the DWR estimates. Combining Faunt's and Famiglietti's data for the time of a 12-year dry period, lasting from 1998 to 2012, an average loss of 3.28MAF/yr can be observed (Famiglietti *et al.* 2011).

Closer analysis of the data shows that temporal variability, depending on climatic conditions such precipitation and drought, strongly affect water tables. For example, storage remained nearly constant between 2003 and 2006, before decreasing rapidly during drought conditions from 2006 to 2010, with depreciation reaching 4.86MAF/yr (*Ibid.*). This phenomenon stems from lower surface water recharge and higher abstraction in dry periods, resulting in declines with partial recovery in wet years. Faunt (2009) shows that permanent losses occur only in the Tulare Basin, showcasing high regional variability. Receiving higher precipitation, northern aquifers receive sufficient recharge and have renewable qualities, whereas use in the Tulare Basin shows non-renewable characteristics and occurrence of groundwater mining, i.e. overdraft (Scanlon *et al.* 2012, p.9323).

Despite interchanging dry and wet periods average groundwater levels continually declined over the past 50 years. Climate change predictions further indicate that droughts will become more severe, persistent and frequent (Cayan *et al.* 2010; MacDonald 2010) in addition to an average of 20-60% less surface water flows and 50% lower groundwater recharge. Hanson *et al.* (2012), projects continual groundwater declines of 1.78MAF. A predicted minimum of 25% snowpack loss in the Sierra Nevada Mountains would result in an annual loss of 3.75MAF (DWR 2008, p.4). This exhibits a peak non-renewable water effect and causes permanent supply decreases of 2.41MAF by 2035, if a linear average decline is assumed.⁹ It is difficult to say to what extent groundwater declines already incorporate this loss, but interesting to see that these predictions align with existing results.

⁸ Conversion Factor: 1 acre-foot per year (AF/yr) = 1,233 cubic meter per year (m³/yr)

⁹ Over period of 42 years from 2008 to 2050, total predicted decline of 3.75MAF was multiplied by $(27/42)$ to get the cumulative linear increase by 2035.

Table 1: Groundwater Storage Change in the Central Valley by Study						
Source, Method & Special Climatic Periods	Study Period	Study Years	Storage Change Total, Confidence Interval	Storage Change Total	Storage Change Avg/yr, Confidence Interval	Storage Change Avg/yr
Faunt (2009) Hydrologic Modelling	1962 - 2003	41		-57,7		-1,4
drought period	1974 - 1976	2				-9,97
drought period	1985 - 1989	4				-6,65
dry period	1998 - 2003	5				-3,97
Famiglietti et al. (2011) Satellite Geometry	2003 - 2010	7	[-13,38 – -19,54]	-16,46	[-2,06 – -3,00]	-2,53
variable/wet period	2003 - 2006	3	[+3,49 – -4,29]	-0,4	[0,94 – -1.16]	-0,13
drought period	2006 - 2010	4	[-14,91 – -19,85]	-19,38	[-3,68 – -6,02]	-4,86
Combined data (Faunt and Famiglietti)	1998 - 2010	12		-39,32		-3,28
DWR (2015a) Water Table Observation	2005 - 2010	5	[-5,4 – -13,1]		[-1,08 – -2,62]	-1,85

Table 1: The table compares the storage change estimated by three key studies covering different time periods with varying methods. Water quantities are given in MAF and were taken from quoted studies. They may not add up due to rounding. The author estimated the annual average for the DWR study (2015a) by calculating the median of the range provided in the study. Faunt's average data is taken from the Water Balance on p.63.

On these grounds, it is reasonable to assume that the above-average dry periods 1999-2003 and 2006-2010, exhibits 'chronic shortages', which will likely become the new average climate in California (Hanson *et al.* 2012; Scanlon *et al.* 2012; Barton 2015). This study will therefore utilise Famiglietti's estimates of 2.53MAF as a medium, the 12-year combined dry period with 3.28MAF as a high, and the DWR mean average of 1.85MAF as a low baseline of unsustainable water loss under existing dry conditions and likely future average conditions. Adjusted for Central Coast depletion quantities, this results in State-wide average depletion baselines of 2.76MAF (medium); 3.58MAF (high); and 2.02MAF (low).¹⁰

4.3 Supply Capacity of Ocean Desalination

Having had only negligible capacity in California so far, the largest desalination facility of the Western Hemisphere with a capacity of 56,000 AF/year (50 million gallons/day), started producing water in Carlsbad in December 2015. The plant aims to supply 7% to 10% of San Diego

¹⁰ Calculation: $dS_{(CV+CC)} = [(dS_{CV} / CV \text{ groundwater abstraction}) \times (CC \text{ groundwater abstraction})] + dS_{CV}$.
CV abstraction = 12.13MAF; CC abstraction = 1.1MAF, based on DWR 2014a data.

County's water demand for 3.1million people (SDCWA 2015, 2016).¹¹ Table B.1 (Appendix) gives an overview of 15 currently proposed projects that are under consideration along the South and Central Coast. By implementing all projects, the total seawater desalination capacity would rise approximately to 438,353 AF/year (DWR 2014b, p.10-27f). Including two further plants located in Mexico, raises the proposed capacity to about 575,000AF/year. The USBR study (2012) gives lower yield estimates of ocean desalination in the Pacific Ocean of 256,000AF/year by 2035. Including capacities in the Gulf of California, however, a similar total of 450,000AF/year is estimated.¹²

Experiences from Carlsbad, which took 18 years to be completed, show that the planning and permitting process can be cost and time intensive (Garret 2014). Predicted completion dates of 2015/16 were documented by Cooley *et al.* (2012) for several plants, but will not be reached by most of them. Considering an estimated planning and implementation duration of at least 6 to 10 years, a much greater capacity is not foreseeable for the next 20 years (LAEDC 2008). With total State-wide water use standing at 42MAF/yr, the estimated capacity would account for 1 percent of California's water. ¹³

4.4 Supply Capacity of Water Recycling

An estimated 669,000AF of water were reused in California in 2009, mostly in the Central Coast and the Central Valley. Given restrictions in existing public health legislation, none of this water is used for direct potable purposes, but most of it is applied in agricultural or landscape irrigation (SWRCB and DWR 2012; DWR 2014b).

¹¹ Conversion Factors: 1 gallon (gal) = 3.785 liter (L); 1 million gallons (Mgal) = 3.07 acre-foot (AF); 1 million gallons per day (Mgal/d) = 1.121 thousand acre-feet per year (AF/yr)

¹² For California to access water sources on Mexican ground, involves physical transportation from Mexico or paper transfers of water rights to the Colorado River.

¹³ In longer time-scales, a USBR study (2012) estimates that ocean desalination could potentially contribute 600,000AF/year in the Pacific Ocean in California, 56,000AF/year in Mexico and 1.2MAF/yr in the Mexican Gulf of California by 2060. Yet, not all of this would be available to the State of California. If 50% of the capacity from the Gulf was transferred to the State, which would involve great costs and bilateral agreements, seawater desalination could provide 1.2MAF/yr to the water supply in 2060 (*Ibid.*).

Table 2 summarises the recycling potential predicted by three key documents. Converted to 2009 levels, a comprehensive assessment in 2003 predicted 1.0-1.23MAF/yr of new water by 2030. The California Water Plan 2013 estimated between 1.4 and 1.9MAF/yr of new water, which roughly aligns with the SWRCB 2013 policy plan. Both show an increase in feasible potential over time. Theoretical capacity limits for water recycling under perfect efficiency are subject to the quantity of wastewater effluent produced, which can be predicted to rise with population growth from 5MAF/yr in 2002 to 5.75MAF/yr by 2035 (DWR 2003; DWR 2014c).¹⁴

Table 2: Potential Water Recycling Capacity in California by 2030 (MAF/yr)				
Study	Year	Potential Increase (over 2002 levels)	Potential Increase (over 2009 levels)	Estimates of Increase in New Water (2009 levels)
Recycled Water Task Force	2003	1.4 - 1.67	1.23 - 1.5	1.0 - 1.23
SWRCB	2012	At least 2	At least 1.83	-
California Water Plan Update	2013	-	1.8- 2.3	1.4 -1.9

Table 2: The comparison of water recycling capacities was compiled by the author, utilising predictions presented in the three stated policy documents (DWR 2003; SWRCB 2013; DWR 2014b).

¹⁴ Water Task Force estimates of 6.5MAF have been adjusted downwards by 50% to accommodate for updated population growth predictions of 4.5 million (DWR 2014c).

4.5 Results and Costs:

4.5.1 Can New Supply Substitute for Groundwater Depletion?

The results show that it is possible to meet the lower groundwater depletion baseline of 2.02MAF/yr with lower desalination and high recycling estimates, as illustrated by Figure 1. A combination of the full estimated capacity for water recycling (1.9MAF/yr) and desalination (0.57MAF/yr) (4th column) falls short to meet the medium baseline of 2.76MAF. In combination with other strategies that provide smaller or less reliable supply, however, supply substitution appears possible (DWR 2014b). The high depletion baseline of 3.58 is comparable to the predicted decline of snowpack by 2050, but reaching it with analysed supply strategies does not seem feasible by 2035. In this case, current water demand is unsustainably high. Only when considering a longer timescale e.g. 2060, and including the construction of plants in the Mexican Gulf of California as well as further increases in water recycling potential, these two supply sources might possibly reach the predicted decline in groundwater.

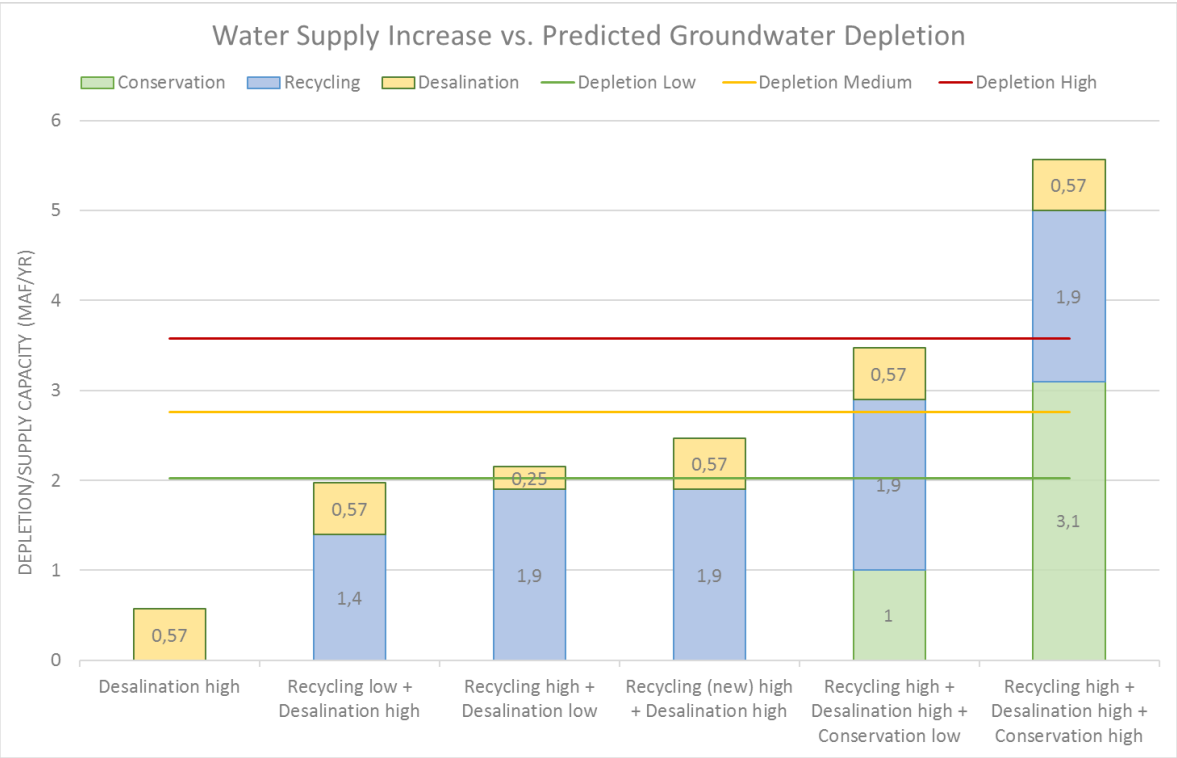


Figure 1 [Colour]: The Graph above shows how different combinations of potential water increase compare against a low, medium and high groundwater decline scenario (in MAF/yr). While desalination and water recycling can substitute for a low depletion scenario, the medium and high depletion scenario, show the importance of adding conservation as a demand-side measure to avoid unsustainable water use.

The final two columns highlight the importance of substitutability in demand when included in the analysis. Urban and industrial water conservation measures hold an estimated capacity to decrease overall demand between 1.0 and 3.1MAF/yr by 2030 (DWR 2014b, p.1-9).¹⁵ By including demand-side water reduction, total capacities reach 5.57MAF/yr., underlining California's potential to meet the high depletion baseline.

4.5.2 Estimating the Economic Costs

Estimating the monetary costs of new water supply is relevant to assess the economic and social feasibility under current conditions, but can only be done in brevity, foregoing a comprehensive cost-benefit analysis (CBA).

Cost accounts of for desalinated water are largely inconsistent in their scope. The costs are often specific and highly variable to geographic location, depending on the choice of water intake and composition, desalination technology, infrastructural and permitting costs and energy price variability, and the selected reporting method. Most available cost estimation tools provide few details on how to account for these differences (Reddy and Ghaffour 2007; Ghaffour *et al.* 2013). Based on two comparative articles Table 3 provides an indication that annualised capital costs account for approximately 40% of total annualised costs. Energy costs for desalination plants make up the largest annual operational cost, but depending on location, their average varies substantially between 19% to over 36% of total annualised costs, as detailed in Table 3. In comparison, costs for drinking water from groundwater in California are mainly incurred at the treatment stage, amounting to 71% (Cooley and Phurisamban 2016).

The unit costs of groundwater vary substantially by region and can be difficult to quantify as many farmers rely on private wells whose costs are unknown. Marginal unit costs are relatively high in mainly urban San Diego County, ranging between \$375 and \$1100 per AF (Equinox 2010, costs in 2010 USD). Across California, recent studies and survey indicate that the energy costs for pumping groundwater from wells are \$40 per AF on average and that total pumping costs are around \$100 per AF. The costs for treatment to achieve drinking water quality have been estimated to be \$240 per AF, resulting in total costs for drinkable groundwater at approximately \$340 per AF (USDA 2013, Cooley and Phurisamban 2016).

¹⁵ Agricultural irrigation efficiency is not considered here, due to its ambiguous effects on groundwater recharge (Perry 2007; Scanlon et al. 2012).

Table 3: Composition of Project Costs for Desalination and Groundwater					
	Desalinated Water Costs (Reverse Osmosis)			Groundwater Costs	
Source		Ghafour <i>et al</i> (2013)	Cooley and Ajami (2014)		Cooley and Phurisamban (2016)
	Composition	Typical Cost Breakdown		Composition	Breakdown
Direct	Capital Costs	41%	37%	Capital Costs - Pumping***	15%
	Constructions Costs: buildings, equipment, piping, civil works on site, infrastructure for intake and brine discharge, roads, laboratories, land costs	21-35%		Well drilling, pump, tanks, piping system, electrical wiring	
Indirect	Planning, permitting, designing costs: capital costs and interest, overheads, insurance, freight, permits and import duties, fees for specialists (architects, engineers, project managers, lawyers)	6-20%		Designing, capital costs, permits (if needed), fees for specialists,	
Fixed	Operating Costs	59%	63%	Operating Costs - Pumping	15%
	Insurance, Amortisation, Labor, Maintenance, Monitoring	14%	10%	Insurance, Amortisation, Labor, Maintenance***	3%
Variable	Energy	19%	36%	Energy	12%
	Membrane replacements	16%	4%		
	Chemicals	6%	12%		
	Spare Parts	4%	1%		
	Total	100%	100%	Capital & Operating Costs - Treatment	71%
				Total	100%

* The cost breakdown is based on a global sample of reverse osmosis plants.
**The typical costs assume a reverse osmosis plant with a capacity of 50 million gallons per day; "constant energy costs at \$0.07 per kilowatt-hour; membrane life of five years; nominal interest rate of 5 percent; and a depreciation period of twenty-five years" (Cooley and Ajami 2014, p.96).
*** These are estimates as no specific information is provided by the source.

Table 3 shows the composition and breakdown of project costs for desalinated water based on two estimates of a typical desalination project, as well as of production costs of groundwater in California. Groundwater costs are divided into pumping and treatment costs and are based on average estimates from the literature.

To avoid bias from heavily subsidised and undervalued existing supplies, marginal unit costs of water provisions are used in Figure 2 to compare alternative water management strategies. Unit costs are arrived at by dividing the sum of annualised capital costs and annual operating costs, by the average annual production volume. The comparatively high unit costs of ocean desalination are clearly visible, ranging between \$1500 and \$3000/AF (Cooley and Ajami 2014, p.99). In cases like the Carlsbad plant, the lower end figure would not include the integration into the existing infrastructure network, which amounts to an additional 16% of the capital costs. Unit costs of recycled water are lower, but vary significantly between \$300 and \$1300/AF. Local wastewater quality, infrastructure and transportation costs to user or recharge sites, permitting and financing conditions affect unit provision costs differently (DWR 2014b). For non-potable irrigation water, the costs of fitting a dual-piping system vary strongly (Sheehan 2009; CSD 2012; Equinox 2010).

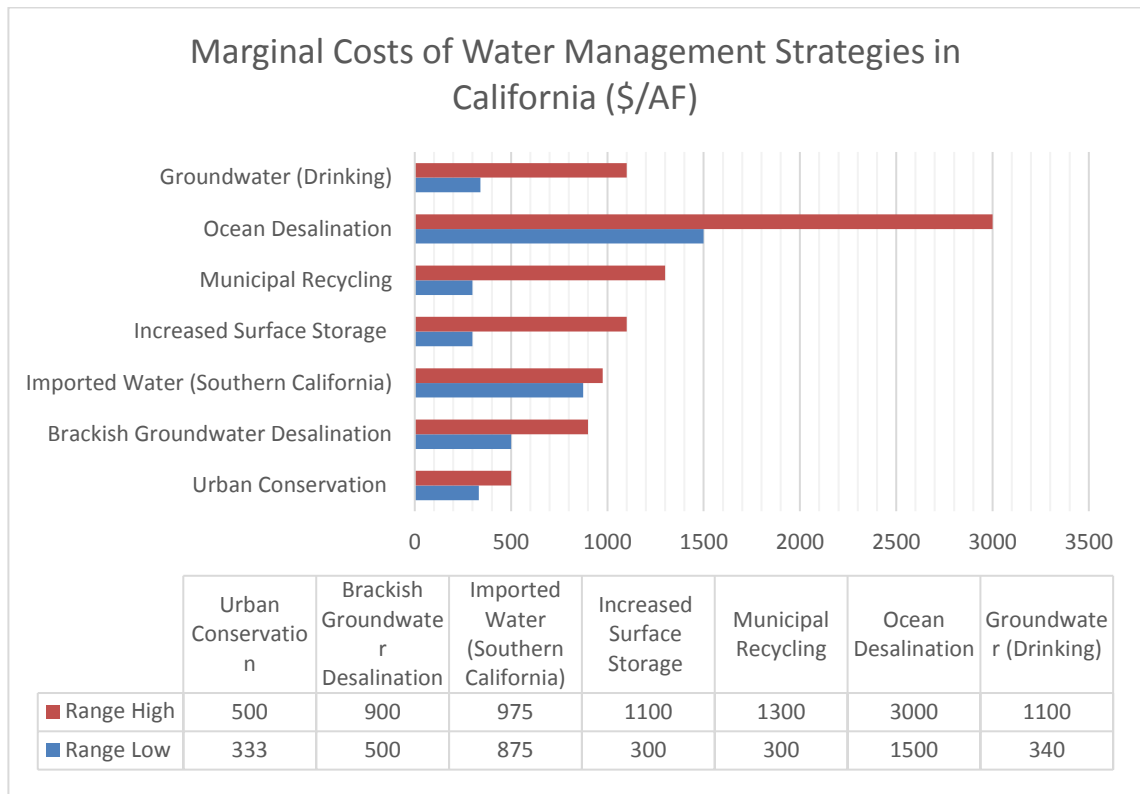


Figure 2 [Colour]: The marginal costs of water vary significantly, not only between chosen strategies, but also between location and type for the same strategy. Data compiled by the author from these sources: DWR 2014b, p.1-12, p.3-32; Equinox 2010; Cooley and Ajami 2014; Cooley and Phurisamban 2016.

To determine the economic feasibility, both the costs of alternative supply sources and the costs of not increasing water supply ideally need consideration. A few other relevant alternative supply options such as brackish groundwater desalination and new surface storage are included in Figure 2, showing unit costs for projected capacities of 0.22MAF/yr and 1.1MAF/yr. Surface water storage and some other options are limited by the availability of surface water for recharge, making them a less reliable future water source (DWR 2014b, p.9-27; USBR 2012).

The most cost-effective option shown is urban water conservation, ranging between \$333 and \$500/AF. This involves changes in human lifestyle, existing water consumption patterns and associated opportunity costs of not enjoying them anymore. The range of conservation measures involves replacing lawns and gardens with drought-resistant landscape, prohibiting hosing of pavements, assigning water budgets, installing greywater systems, low-flush or composting toilets, rain barrels, bucket showers and hot water recirculation pumps (DWR 2014b; Orbach 2014). It is important to be aware that “conservation programs could start to run into diminishing returns over the next two decades as the easiest and least costly options for water users are implemented” (Equinox 2010, p.7). The political decision whether to increase supply or to consume

less, fundamentally involves “estimating just how much the public is willing to change their water use habits versus how much they are willing to pay to avoid having to do so” (Orbach 2014, p.72).

In light of existing legislative orders preventing further exploitation of surface flows beyond peak ecological limits, not acting at all invokes further depletion of groundwater, leading to declining water availability for agricultural users in the future (Moyle and Bennet 2008; Faunt 2009). Fallowing of lands is a trend, which was observable in the 2015 drought in the order of 2.2% of cultivated land (Howitt *et al.* 2015; The Economist 2014).

The Californian water rights market (1.4MAF transferred in 2011) gives an indication of the value of water, even though subsidies and environmental regulations distort prices and no overview of prices exists (Hanak and Stryjewski 2012; Cooley and Ajami 2014). Interviews show that western San Joaquin Valley farmers paid “routinely more than \$100 per acre-foot” for additional water (Hanak *et al.*, 2009, p.22). With subsidised contract prices for agricultural water ranging between \$10 and \$65, however, prices of alternative supplies starting at \$300/AF exceed most farmer’s willingness to pay (*Ibid.*). Potential economic costs resulting from water supply interruptions in a major north-south pipeline can also be accounted for as one LAEDC study did (2013).

4.5.3 Influencing and limiting factors

Since the potential production capacity is based on socio-economic considerations of government agencies and water suppliers, the technically and environmentally feasible capacity is likely to be substantially higher than the estimated 2.47MAF/yr. Limiting factors might arise from significant, uneconomical damages to marine ecosystems and energy demand of desalination at larger scales, as well as limits in the quantity of available wastewater which is currently needed both for water recycling and to mix salty brine discharges (Sheehan 2009; Cooley and Ajami 2014; Lattemann and Höppner 2008; Poseidon Water 2005). With legislative prohibition of once-through water cooling systems for power plants, availability of wastewater provides a potential limiting factor on feasible construction sites for ocean desalination in the absence of improved discharge technologies (DWR 2014b). The high energy intensity of desalination technologies and associated GHG emissions in the light of California’s commitments under the 2006 Global Warming Solutions Act to reduce emissions to 1990 levels, might pose an additional political and environmental limitation (ARB 2014; Poseidon Water 2005, 1-18). Table 4 summarises average GHG emissions for different water sources in California. Based on an average energy-use of 4880 kWh/AF, the maximum projected desalination capacity would increase California’s

total energy use by 1.08% over total 2013 demand (EIA 2015), raising GHG emissions by 0.22% of California’s total emissions in 2013 (Cooley and Heberger 2013).¹⁶ To put this into perspective, 73% of California’s water associated electricity use is spent in end-uses, including pumping, heating and wastewater treatment, and not in production or conveyance. Consequently, the increased water treatment costs from desalination do not seem unacceptably high from a GHG perspective, as other saving opportunities exist (CEC 2006, p.16). Contrarily, water recycling is a much lower energy intensive supply option with the potential to reduce water stress in rivers from sewage pollution and intensive abstraction (Table 4; Cooley *et al.* 2014; Equinox 2010; LAEDC 2008; DWR 2014b).

High unit and investment costs associated with new supplies might, however, seriously limit the implementation of analysed proposals, if more economical alternative opportunities exist. Implementation will firstly depend on the value which water agencies and authorities place on reliability over costs considering local conditions e.g. political will for growth and reliable local supplies influencing the Carlsbad Desalination Plant in San Diego (Zetland 2014b; DWR 2014b; SDCWA 2015). Secondly, with urban conservation being on average more cost-effective than most supply increase options (Figure 2), new water supply might be limited by questions of optimal decision making and people’s willingness to pay for additional supplies. Social opposition has been a limiting factor in the city of Santa Cruz, where strong environmental activism prevented new water supply from desalination or recycling in return for a strict conservation plan, resulting in 50% of the Californian average water consumption (Rogers 2014; Orbach 2014).

Table 4: Energy Intensity for Different Water Sources in California (kWh/AF)		
	Low Estimate	High Estimate
Ocean Desalination	3900	5800
	2550	4550
Imports to South California (via SWP)	3170 ¹	
Recycling potable	1070	2700
Imports to North California	690 ¹	
Recycling non-potable	320	590
Conservation	Decrease	0

Table 4: The table shows low and high estimates and, where not available, the average energy intensity of water from different supply sources. Californian average surface imports don’t include an average of 36kWh/AF for treatment (CEC 2006) Modified from Cooley/Heberger 2013 and CEC 2006.

¹⁶CO₂ equivalent, based upon IPCC Fourth Assessment Report’s Global Warming Potentials and local energy mixes in 2013.

5 Discussion

5.1 How do the Results relate to the Substitutability of Water Supply?

The results agree with existing evaluations that stress the need for a portfolio of water strategies, rather than one silver bullet solution, to balance California's water household (LAEDC 2008; USBR 2012). To infer the substitution capacity of all feasible water supply sources clearly requires a more comprehensive analysis. The results show, however, that the theoretically conceivable capacity of ocean desalination to provide a backstop technology for freshwater, as suggested by Gleick and Palaniappan (2010), currently lacks practical credibility in light of a projected water supply share of 1% by 2035. The implementation of new water supply technologies suggests that California has crossed or is at least near the point of "maximum cost-effective extraction of surface and groundwater" (*ibid.*, p.11157). Yet, limitations of economic feasibility give evidence that a backstop price for water has not been reached. The state of current technology and environmental limitations cast serious doubt on the ability of desalination to provide "virtually infinite" amounts, or even sufficient quantities to replace the entire surface water supply of California in a foreseeable timeframe (Nordhaus *et al.* 1973, p.548). This finding supports existing assessments of non-conventional water resources, which argue that their quantities are not sufficient to support agriculture and local food security in water scarce countries (Qadir *et al.* 2007, p.2). Yet, as determining backstop capacities of a technology requires very long time-scales, and because future possibilities are veiled by uncertainty, no final judgement can be made (Neumayer 2013).

The high baseline draws up possible limits in water substitution with current technologies and produced capital capacities, but it does not preclude the ability of future supply capacities to adjust to the challenge and manage water substitution at high levels by using greater produced and human capital, as described by propositions 3 and 4 and by Fenichel and Zhao (2014) in chapter 2. These reservations are underlined by the inability to analyse the role which price signals play in balancing resource consumption in accordance with scarcity, as described by proposition 2 (Neumayer 2013). The high potential of demand-side solutions, highlighted in Figure 1 and 4, urge the debate on substitutability to move beyond produced and human capital to take account of institutional and social capital. Both are key components in increasing systemic efficiency and reducing transaction costs, which can encourage customers to adopt conservation measures and accept the outlined trade-offs more readily.

5.2 The Relevance of Institutional and Social Capital for Water Substitutability

Since social/institutional capital can reduce uncertainty and transaction costs (North 1990), it arguably represents “total factor productivity”, which means that it can enhance the productivity and value of produced, natural, and human capital (Hamilton and Liu 2014, p.70). Water conservation, water pricing and regulation of groundwater in California can serve to exemplify that social/Institutional capital might compensate declining natural resources.

While agreement on optimal decisions concerning conservation measures can be difficult to reach, Lee *et al.* (2011) argue that greater social capital makes consensus for relevant public goods easier (Orbach 2014). Water is often considered to be a public good. Free-rider problems can prevent effective conservation measures in the absence of a) social capital in the form of reciprocal relationships, cultural norms and trust, or b) institutional capital in the form of regulation, sanctions or economic incentives (Hardin 1968; Ostrom 1990). Wells (1998) highlighted the importance of “institutions in designing and implementing effective incentive measures” for conservation, thus encouraging substitutability via efficient demand reductions (p.815).

Considering irrigation inefficiency of subsidised water in agriculture for example, adequate water pricing can “send an important signal to internalize the scarcity level of water and change the behaviour of consumers”, thus providing a cost-effective incentive for conservation (Dinar quoted in Miller 2015; Zetland 2014a; Equinox 2009; The Economist). A variety of pricing mechanisms, such as tiered rates, are feasible and can be designed to establish fair structures without threatening basic water demands from low-quantity households (Dinar *et al.* 2015; Equinox 2010; Cuniff 2015). Reductions between 17-36% have so far been observed in some districts (Equinox 2009).

The described depletion of water tables occurred in the absence of any State regulation or management of water use rights, particularly related to groundwater, with most groundwater wells being under local ownership (Chappelle 2015; CWC 2014). During recent droughts, table depletion has led users to drill deeper wells, creating increasing social conflict over water (Rogers and Leal 2010; Pretty and Ward 2001). Acknowledging the problem, California has passed the Sustainable Groundwater Management Act (SGMA) in 2014. This provides a legislative framework, which requires the registration of individual groundwater wells and annual extraction monitoring. It sets limits on extractions and raises usage fees to achieve sustainable use within aquifer yields over the next 20-years (ACWA 2014). While the outcomes remain to be seen, Pretty and Ward’s (2001) present supportive findings, highlighting that greater water efficiency and equity

achieved by local water-user groups with locally established rules and sanctions, in comparison to individuals who operate alone or in competition.

These examples underline the great relevance of institutional and social capital in mitigating future water scarcity in California. They can facilitate demand reduction, which suggests that $K_{S/I}$ should receive greater attention in the discussion on the substitutability of natural capital. Investment in social or institutional capital to reduce water consumption can be described in terms of “capital maintenance” i.e. ensuring that renewable natural capital assets can provide services in perpetuity (Helm 2014). According to Helm, the consumption of natural capital without ensuring its maintenance gives testimony to excessive usage pattern for which the next generation is bound to pay. New water supply strategies can similarly achieve this, by financing the higher costs of new water to prevent unsustainable groundwater depletion. Turning back to the costs of provision and optimal economic decisions making, conservation appears to be the most sensible option, when unknown social costs of behaviour change are not included.

5.3 Conclusion

The results of the case study in California show that water substitutability from analysed supply sources is viable for the low depletion baseline, but limited in case of the medium and not feasible for the high depletion baseline. Assessing economic costs of provision in the face of alternative supply sources, the marginal value of water in agriculture, and opportunities of substitutability in demand via conservation, raises questions of economic efficiency and optimal decision-making (Zetland 2014a). California seems to have crossed a point of “cost-effective surface and groundwater extraction”, but the high price of new supply sources limits feasible supply quantities (Gleick and Palaniappan 2010).

Introducing substitutability in demand to the analysis has shown that California is more than capable to meet the high depletion baseline, suggesting full substitutability of groundwater overdraft in California for the analysed timeframe. Although limited to groundwater depletion, this result supports the ‘resource optimistic’ view and the substitutability assumption of weak sustainability (Neumayer 2013). Nevertheless, the nature of the substitutability question and its dependence on future capacities to overcome scarcity, limits the temporal applicability of these results. Within the given context, they point to limits in maintaining a ‘supply without limits’ driven mentality. By including institutional and social capital, the results simultaneously support the substitutability assumption for the presented case study on water in California.

Further research should expand capacities of unsustainable water use and predicted future shortages beyond groundwater to adequately take account of precipitation and snow-level loss, as well as ecological costs of reduced streamflow in the Colorado River. Similarly, a more comprehensive analysis of supply capacities would ideally include the full range of realistic options. Further case studies on substitutability of water in other locations could help to generate a broader picture about the nature of water substitutability.

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Appendix A.1:

Discussion of Neumayer's 4 propositions of weak sustainability

Neumayer (2013) outlined that the Weak Sustainability theory assumes 4 different propositions that support the substitutability of various capital forms:

1. Substitutability with another resource which becomes economically viable due to its comparatively lower cost. Ultimately, this depends on the availability of a 'backstop technology' – a resource that has a nearly infinite resource base and which can meet demand at a constant marginal price (Nordhaus *et al.* 1973). No reliable calculation attempts exist for non-energy resources so far (Neumayer 2013; Ayres 2007).
2. Prices signal economic scarcity and drive dynamic markets to adapt towards efficiency under new scarcity conditions (Neumayer 2013). Yet, according to Dietz and Maddison (2009), the values of these parameters are largely unknown. This is due to great difficulty to empirically measure either resource rents, which should indicate scarcity, or their surrogate indicators, unit extraction costs and relative resource prices (Hotelling 1931).
3. Natural resources can be substituted with produced capital, if the elasticity of substitution is greater or equal to 1. Originally a single, constant elasticity of substitution was assumed by Hicks (1932) and Solow (1974). Yet, recent studies suggest a non-constant elasticity function, which displays high marginal substitutability in abundance, but decreasing substitutability below a certain boundary point induced by suggested technical, economic or social limits (Gerlagh and van der Zwaan 2002; Drupps 2015; Traeger 2011).
4. Technical progress, driven by human capital, can affect substitutability through efficiency gains in production, as well as via cheaper production techniques which make lower-profitability resources economically available and thus increase their stock. Whether technical progress can continue indefinitely for a resource is uncertain (Neumayer 2013; Lecomber 1975). Investing rents in knowledge/ingenuity (K_H) can reduce dependency on certain resources, but replacing the full range of ecosystem services might be impossible or too expensive (Fenichel and Zhao 2014; Krautkraemer 2005; Brauman *et al.* 2007).

Table A.1: Groundwater Withdrawals in California

Groundwater Withdrawals in California by Hydrologic Region (Annual Average 2005-2010)

Hydrologic Regions	Total Withdrawals (MAF)	Percentage of Total Withdrawals	Agriculture (MAF)	Public Supply (MAF)	Watershed Management (MAF)
Sacramento River	2742.9	17%	2294.2	428.6	20.1
San Joaquin	3198.4	19%	2591.8	415.9	190.7
Tulare Lake	6184.8	38%	5551.8	604	28.9
Central Valley Aquifer System	12126	74%	10437.8	1448.5	239.7
North Coast	364	2%	301.3	60.3	2.5
San Francisco	259.5	2%	76.1	183.5	0
Central Coast	1119.5	7%	906.1	213.3	0
South Coast	1605	10%	385.4	1219.6	0
California Coastal Aquifer System	3348	20%	1668.9	1676.7	2.5
North Lahontan	166.2	1%	118.4	37.1	10.7
South Lahontan	440.9	3%	270.6	170.3	0
Colorado River	379.7	2%	50.1	329.7	0
Other Aquifer Systems	986.8	6%	439.1	537.1	10.7
2005-2010 Annual Average California Total	16460.8	100%	12545.7	3662.2	252.9

Numbers may not add up due to rounding. Source: DWR 2014a

Table B.1: List of proposed Desalination Plants in California

Production Capacities and Theoretical GHG Emissions of Proposed Desalination Plants in California

Project Partners	Location	Capacity (TAF per yr)	Emissions (TT CO2e per yr)
East Bay Municipal Utilities District, San Francisco Public Utilities Commission, Contra Costa Water District, Santa Clara Valley Water District, Zone 7 Water Agency	Pittsburg	22,2	30
City of Santa Cruz, Soquel Creek Water	Santa Cruz	5,6	7
Deep Water, LLC	Moss Landing	2,8	3
The People's Moss Landing Water Desal Project	Moss Landing	28,0	30
California American Water	North Marina	11,2	10
California Water Service Company	unknown	10,1	10
Ocean View Plaza	Monterey	0,3	3
Monterey Peninsula Water Management District	Monterey, Del Monte Beach	2,2	0,3
Seawater Desalination Vessel	Monterey Bay	22,4	60
Cambria Community Services District/U.S. Army Corps of Engineers	Cambria	0,7	0,8
Arroyo Grande, Grover Beach, Oceano Community Services District	Oceano	2,2	30
West Basin Municipal Water District	El Segundo	20,2	30
Poseidon Resources	Huntington Beach	56,0	80
Municipal Water District of Orange County, City of Oceanside	Dana Point	16,8	30
Poseidon Resources, San Diego County Water	Oceanside	11,2	20
San Diego County Water Authority	Carlsbad	56,0	90
NSC Agua	Camp Pendleton	168,2	300
San Diego County Water Authority	Rosarito, Mexico	112,1	80
San Diego County Water Authority	Rosarito, Mexico	28,0	300
TOTAL		576,2	1114,1

Data modified from Cooley and Herberger 2013. TAF = thousand acre-feet, TT = thousand tonnes