Environmental Science & Technology

Article

pubs.acs.org/est

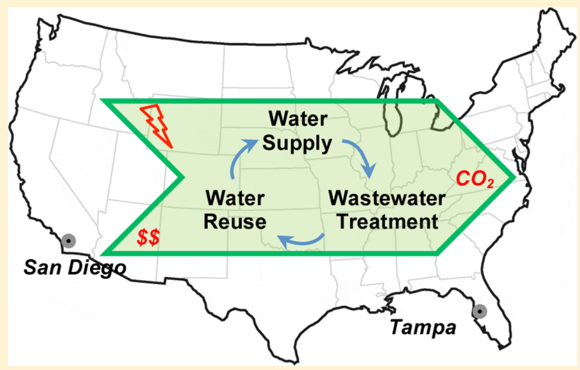
Energy−Water Nexus Analysis of Enhanced Water Supply Scenarios: A Regional Comparison of Tampa Bay, Florida, and San Diego, California

Weiwei Mo,[†] Ranran Wang,[‡] and Julie B. Zimmerman*[,†,‡]

[†]Department of Chemical and Environmental Engineering and [‡]School of Forestry and Environmental Studies, Yale University, New Haven, Connecticut 06520, United States

Supporting Information

ABSTRACT: Increased water demand and scarce freshwater resources have forced communities to seek nontraditional water sources. These challenges are exacerbated in coastal communities, where population growth rates and densities in the United States are the highest. To understand the current management dilemma between constrained surface and groundwater sources and potential new water sources, Tampa Bay, Florida (TB), and San Diego, California (SD), were studied through 2030 accounting for changes in population, water demand, and electricity grid mix. These locations were chosen on the basis of their similar populations, land areas, economies, and water consumption characters as well as their coastal locations and rising contradictions between water demand and supply. Three scenarios were evaluated for each study area: (1) maximization of traditional supplies; (2) maximization of seawater desalination; and (3) maximization of nonpotable water reclamation. Three types of impacts were assessed: embodied energy, greenhouse gas (GHG) emission, and energy cost. SD was found to have higher embodied energy and energy cost but lower GHG emission than TB in most of its water infrastructure systems because of the differences between the electricity grid mixes and water resources of the two regions. Maximizing water reclamation was found to be better than increasing either traditional supplies or seawater desalination in both regions in terms of the three impact categories. The results further imply the importance of assessing the energy−water nexus when pursuing demand-side control targets or goals as well to ensure that the potentially most economical options are considered.



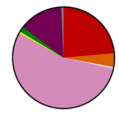
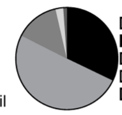
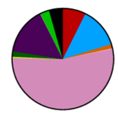
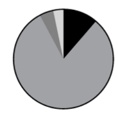
■ INTRODUCTION

During the past decade, water shortage and water scarcity have led to a change in urban water supply in many areas and particularly in densely populated regions.[1−3] These challenges are exacerbated in coastal communities, where population growth rates and densities in the United States are the highest. Alternative water sources such as desalinated seawater, reclaimed water, and even rainwater harvesting have been introduced to augment existing water supplies. For example, despite relying on water importation since the early 1900s, southern California is currently pursuing rapid development of seawater desalination plants, with six in operation and 16 in design and construction.[4] In response to similar drivers of rapid population growth, economic development, and land use changes, southwest Florida currently operates the largest desalination plant in the United States to augment the regional water supply.[5] Capturing and treating water from nonconventional sources such as industrial or municipal wastewater for various reuses is another promising solution to urban water shortages.[6] However, the implementation of water reclamation is still far below its full potential, particularly in water-scarce regions.[7] For example, San Diego County, California, reclaims around 83 000 m^3 of water daily (22 MGD), which is less than 10% of its wastewater treatment capacity,[8] while the daily nonpotable water demand can exceed 750 000 m^3, a significant portion of which is used for residential irrigation.[9] While both desalination and water reclamation and reuse are potentially viable strategies to increase water supply, both technologies present costs and benefits in terms of life-cycle energy, cost, and environmental impacts. This suggests the need to systematically evaluate scenarios for enhancing water supply considering the complete system, including treatment, distribution, and discharge.

Over the past decade, there has been a proliferation of life-cycle studies on drinking water[10−20] and wastewater[21−27] systems reflecting an enhanced understanding of water−energy interdependence and the associated management implications for water and energy infrastructure systems. While these studies stemmed from the needs of management support of a certain community or region, there is a lack of studies comparing the impacts of geospatial conditions on water management

Received: December 18, 2013
Revised: March 25, 2014
Accepted: April 14, 2014
Published: April 14, 2014

Table 1. Baseline Information for the Tampa Bay Water Planning Region (TB) and San Diego County Water Authority (SD) Related to Population, Area, Economy, Water Demand, and the Compositions of the Energy Grid and Freshwater Supply[§]

Region	Popu -lation (million)	Area (km^2)	Regional GDP* ($ billion)	Water demand in 2010 (million m^3)	Projected demand increase by 2030	Grid mix[33]	Water source[5,31]
Tampa Bay Water Planning Region	2.7[34]	5180[34]	116[35]	1.7[5]	21%[5]		
San Diego County Water Authority	3.1[34]	3845[34]	173[35]	2.1[31]	45%[31]		

[§]Carbon emission factors and energy costs for the two regions are included in Table 3. [*]GDP: gross domestic production.

decisions, including the impact of different water source availability and composition of the regional energy grid. Furthermore, although in previous studies water reclamation was overwhelmingly favored over water desalination[14−16,19,20] and importation,[14,15,20] these studies considered water reuse as either part of water source analysis or part of wastewater system impact assessment[14−16,19−21,24−26] but did not consider the entire water infrastructure system (including supply, treatment, and reuse) in a single model. This can provide limited understanding of holistic water management by not effectively considering linkages and feedbacks within the system. Hence, one further step can be taken to close the loop of the anthropogenic water cycle to provide understandings and guidance for regional water management. Closed-loop life-cycle studies avoid problems such as inconsistent system boundaries when comparing different water supply sources.[28] To date, only three studies have combined water supply with wastewater treatment, specifically focused in Australia,[29] Spain,[16] and Belgium,[23] and only one of these also included water reclamation as a potential mechanism to augment supply.[16] In view of the importance of geospatial conditions to planning and implementing future water management strategies, this study is aimed at evaluating the impact of current and various future water supply portfolios, including supply augmented by water reclamation/reuse and desalination, on energy demand, greenhouse gas (GHG) emissions, and costs while considering the current and future regional energy grid mix.

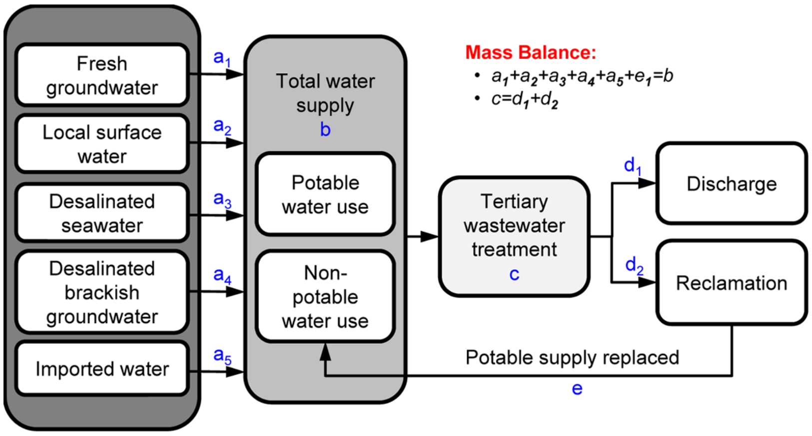
■ STUDY AREAS AND WATER SUPPLY SCENARIOS

Study Areas. Because population growth rates and densities in the United States are highest in coastal counties, exacerbating water supply and demand gaps,[30] the Tampa Bay Water Planning Region (TB) and the San Diego County Water Authority (SD) were chosen. With similar socioeconomic characteristics, water demands, and water scarcity profiles, TB and SD are vigorously developing alternative water supplies; however, these communities have significantly different water resources and regional energy grid mixes (Table 1).

Water resources vary significantly between the two regions. SD has very limited local freshwater resources[4] and thus relies on imported water for approximately 81% of the total water supply.[31] Water importation, however, is not considered a reliable long-term water source because of its extensive energy requirement and limited water availability, especially under the impact of climate change, such as decreasing snowpack storage and increasing drought frequency.[4] TB, on the other hand, is very dependent on local groundwater and surface water sources that together constitute 82% of the total supply.[5] These groundwater and surface water sources are becoming more and more restricted because of concerns about minimum flow levels for ecological systems and seawater intrusion.[5] Therefore, both TB and SD are currently aggressively pursuing a variety of strategies and technologies to augment the existing water supply. Currently, reclaimed water represents 15% and 4% of the total water supply in TB and SD, respectively. Furthermore, desalination of brackish groundwater and seawater, although generally considered to be more energy-intensive than water reclamation,[14,20] has or is being established widely in both regions. For instance, TB is planning a 10 MGD (38 thousand m^3/day) expansion of the current 25 MGD (95 thousand m^3/day) seawater desalination plant, as well as another 25 MGD facility.[5] SD alone has three seawater desalination projects under consideration with one estimated to operate fully by 2020 with a capacity of 50 MGD (189 thousand m^3/day).[4,32]

The electricity grid mixes in the two regions are also very different (Table 1), further affecting the environmental and economic impacts associated with energy use in water systems. The electricity grids in both regions rely heavily on fossil fuels, but SD has a higher and more diverse nonfossil composition than TB. More specifically, among the dominant energy types that are used for electricity generation, natural gas (over 50%) and nuclear (14−15%) are equally important in both regions, while coal use is much higher in TB (24% vs 7%) and hydroelectricity generation is much higher in SD (13% vs 0.01%).[33]

Water Management Scenarios. To understand the impacts of future water management strategies to augment water supply on energy resources, GHG emissions, and costs, three scenarios were developed to transition from the current (2010) water supply portfolio and energy grid mix baselines to 2030. Collectively, these scenarios were designed to highlight the current water management dilemma among exhausting surface water and groundwater sources, perceived energy-intensive desalination technologies, and underdeveloped non-



		a_1	a_2	a_3	a_4	a_5	b	c	d_1	d_2	e
TB	Current	848	541	42	19	0	1685	852	492	360	235
	S1	848	833	94	23	0	2033	973	613	360	235
	S2	848	723	227	0	0	2033	973	613	360	235
	S3	848	549	0	0	0	2033	973	0	973	636
SD	Current	0	208	0	61	1491	1843	1136	1052	83	83
	S1	0	208	568	106	1949	2990	1344	1185	159	159
	S2	0	208	1325	106	1192	2990	1344	1185	159	159
	S3	0	208	0	106	1332	2990	1344	0	1344	1344

Unit: thousand m^3/day

• Note: reclaimed water use efficiency (e/d_2) is 0.65 in the TB and 1 in the SD.

Figure 1. For Tampa Bay, Florida, and San Diego, California, the amounts of water supplied and treated by each type of water infrastructure in 2010 and the projected regional water demand in 2030 for three different water supply portfolio scenarios.

potable water reclamation that exists in many coastal regions, including TB and SD. The three scenarios are as follows:

Scenario 1 (S1): Utilization of currently dominant water supplies (groundwater and surface water in TB and imported water in SD) is maximized; baseline levels of desalination of brackish groundwater and reuse of nonpotable water are maintained; remaining water demand is met by seawater desalination.

Scenario 2 (S2): All of the planned seawater desalination capacities are in full use; the current level of nonpotable water reuse is maintained; the remaining demand is fulfilled in order of least to most energy-intensive[18,20,36] water supplies, moving from groundwater to surface water to brackish groundwater to imported water as sources are exhausted.

Scenario 3 (S3): Tertiary treated wastewater for nonpotable purposes is maximized; the remaining demand is fulfilled in order of least to most energy-intensive[18,20,36] water supplies, moving from groundwater to surface water to brackish groundwater to imported water as sources are exhausted.

The amounts of water supplied or treated by the different types of water infrastructure are provided in Figure 1. The current (2010) supply and potential availability and limits of each water source, including wastewater generation for the two regions, were obtained or estimated from the regional planning agencies' documents.[5,31]

METHODOLOGY

Research Scope and Functional Unit. The current study is a water−energy assessment with system boundaries including the operation, maintenance, and construction phases of the anthropogenic water cycle (water supply, wastewater treatment, and water reuse) for TB and SD. The life span of all water infrastructures was assumed to be 100 years, as used in previous life-cycle assessments of water infrastructure systems.[18,20] The disposition of the water infrastructure and equipment at end of life was not considered, as their associated environmental impacts were shown to be insignificant in previous studies.[11,12]

Overall, six types of water infrastructure supply systems were examined in this study: (1) fresh groundwater, (2) local surface water, (3) desalinated seawater, (4) desalinated brackish groundwater, (5) imported water, and (6) reclaimed water. Except for the reclaimed water supply, all of the water supply systems included water intake, conveyance, treatment, and delivery, and water was assumed to be treated to meet potable standards with typical technologies associated with each type of water in the region (see the Supporting Information for detailed treatment trains). Reclaimed water supply systems included additional treatment of the wastewater plant effluent and water redistribution. Reclaimed water supplies were assumed to be used for nonpotable uses in this study. Wastewater systems included collection, treatment, and discharge (if any). All wastewater treatment systems were assumed to have tertiary treatment ability, given the increasingly stringent wastewater discharge requirements.[1] The functional unit for the analysis is 1 m^3 of water delivered through the anthropogenic water cycle for beneficial (both potable and nonpotable) uses.

Table 2. Direct and Indirect Energy Consumption (MJ of Primary Energy/m³ of Finish Water) of Different Water Infrastructure Systems for the Regions of Tampa Bay, Florida, and San Diego, California

		TB			SD		
type of water infrastructure based on water source	energy consuption	typical	low	high	typical	low	high
fresh groundwater	direct	5.7[36]	5.1[a]	6.2[a]		n.a.	
	indirect	4.0[g]	3.6[d]	4.4[d]		n.a.	
local surface water	direct	4.8[h]	4.5[b]	5.0[b]	4.8[36]	4.3[a]	5.3[a]
	indirect	8.9[g]	8.4[e]	9.5[e]	8.7[g]	7.0[d]	10.6[d]
desalinated seawater	direct	58.0[h]	55.1[b]	60.9[b]	64.7[c]	57.1[42]	72.3[43]
	indirect	14.3[g]	13.5[e]	15.2[e]	34.5[20]	17.6[f]	57.9[f]
reclaimed water	direct	4.0[h]	3.6[a]	4.4[a]	16.0[c]	4.2[42,44]	27.9[20]
	indirect	1.8[g]	1.6[d]	2.0[d]	8.6[20]	1.3[f]	22.5[f]
desalinated brackish groundwater	direct	22.1[c]	6.3[37]	37.8[37]	20.9[c]	4.3[42,45]	37.5[20]
	indirect	5.9[g]	2.3[g]	9.6[g]	14.5[20]	1.7[f]	39.0[f]
imported water from northern CA	direct		n.a.		34.3[42,43]	30.8[a]	37.7[a]
	indirect		n.a.		17.9[20]	10.4[f]	26.9[f]
imported water from Colorado River	direct		n.a.		21.1[43,45]	19.0[a]	23.2[a]
	indirect		n.a.		11.0[20]	6.4[f]	16.6[f]
wastewater	direct	8.9[h]	8.5[b]	9.4[b]	10.2[c]	6.9[43,45]	13.4[42]
	indirect	8.5[g]	8.0[e]	9.0[e]	9.3[g]	5.6[d]	13.6[d]

[a]An uncertainty of 10% was assumed for recent literature data. [b]An uncertainty of 5% was assumed for recent primary data. [c]Average of the range provided by the literature. [d]10% uncertainty for recent literature data and 1% uncertainty for the input–output data. [e]5% uncertainty for project-specific data and 1% uncertainty for the input–output data. [f]10% uncertainty for recent literature data, 35% uncertainty for the relative difference between the different estimation methods, and 1% uncertainty for the input–output data. [g]Estimated through input–output-based hybrid analysis. [h]Data obtained from real systems or estimated on the basis of the primary data.

Energy–Water Impact Assessment. Three types of impacts were assessed in this study: embodied energy, GHG emissions, and cost of energy to deliver a functional unit of water. Embodied energy is the life-cycle energy use in water and wastewater systems, including direct energy (energy used directly for operation and maintenance) and indirect energy (energy used indirectly to provide materials and services during operation and maintenance and energy used during construction). GHG emissions and energy costs are both associated with the embodied energy consumption of the water and wastewater systems.

For TB, direct energy consumption of the different water infrastructure systems was primarily obtained from representative real systems in the region through personal communications,[18,27] except for the fresh groundwater supply[36] and desalinated brackish groundwater supply[37] (Table 2). For SD, direct energy consumption of the different water infrastructure systems was primarily based on literature values (Table 2). In cases where there was no consensus on typical energy consumption by a certain type of water infrastructure, minimum, average, and maximum values were used in the assessment for a comprehensive understanding.

Indirect energy consumption of water infrastructure systems in TB was estimated through an input–output (IO)-based hybrid analysis, which entailed a structural path analysis to substitute the energy paths derived from IO tables with system-specific data[17,18] (see the Supporting Information for details). Indirect energy consumption of water infrastructure systems in SD was adopted from Stokes and Horvath,[20] who adopted a process-based hybrid analysis relying on detailed direct material and service inputs and then estimated the embodied energy of each input through IO tables. On the basis of the same inventory data, it has been reported that process-based hybrid analysis generally underestimates building embodied energy by 18–52% compared with IO-based hybrid analysis,[38] and the latter is favored because of its reduced truncation error and more comprehensive approximation.[38,39] In view of the variations in system boundaries and the inventory data, an uncertainty of 35% was assigned to the indirect energy obtained from Stokes and Horvath[20] to account for the differences between the two methods.

In order to improve the overall robustness of the results, recommendations from Lloyd and Ries[40] were adopted such that uncertainties of 5% and 10% were assigned to data obtained from real systems and from recent literature, respectively. For data estimated from the IO-based hybrid analysis, an uncertainty of 1% was assigned to represent errors in the IO table, as indicated by Bullard and Sebald[41] and Lenzen,[39] in addition to the uncertainties associated with the real system or literature data used for structural path analysis. For data obtained from Stokes and Horvath,[20] in addition to the 35% uncertainty assigned to represent the method, a 1% uncertainty was assumed for using IO tables and a 10% uncertainty was assigned for its raw inventory data.[38]

To compare energy, carbon, and cost for the water infrastructure systems and scenarios in TB and SD, energy was divided into four types: coal, natural gas, petroleum, and electricity. These fuel types were converted into primary energy through a series of primary energy factors, which take into account energy losses during generation and transmission.[17,46] While the primary energy factors for coal, natural gas, and petroleum were assumed to be the same for the two regions, the primary energy factors for electricity were estimated for TB and SD on the basis of the appropriate energy grid mix using a method provided by the U.S. Energy Information Administration (EIA)[47] (details are provided in the Supporting Information). Similarly, GHG emissions were estimated on the basis of the carbon emission factors associated with the different types of energy.[48] Emission factors for coal, natural gas, and petroleum were assumed to be the same for the two regions and were based on the national averages,[27,48] while carbon emission factors for electricity were estimated on the basis of U.S. Environmental Protection Agency (EPA) eGRID2012 data for TB and SD[27,33] (details are provided in the Supporting Information). Energy costs were estimated on the basis of energy prices and the energy breakdown among fuel types with region-specific energy prices obtained from EIA databases using 2010 as the base year.[49−52] Table 3 summarizes the values of the primary energy factors, carbon emission factors, and energy prices used in this study.

Table 3. Primary Energy Factors, Carbon Emission Factors, and Energy Prices for Different Fuel Types for the Regions of Tampa Bay, Florida, and San Diego, California

parameter	region	coal	natural gas	petroleum	electricity
primary energy factor	TB	1.13	1.05	1.42	3.50
	SD				3.62
carbon emission factor (ton CO_2e/TJ)	TB	88.2	50.1	69	148.9
	SD				83.3
energy price (10^4 \$/TJ)	TB	0.47	0.51	1.59	2.78
	SD	0.38	0.45	1.77	3.61

Equation 1 elucidates the calculation of embodied energy, GHG emission, and energy cost for a certain type of water infrastructure:

$$T_w = \sum_{\alpha=1}^{4} (E_{\alpha,o}^{w} \times f_\alpha) + \sum_{\alpha=1}^{4} (E_{\alpha,c}^{w} \times f_\alpha)/t_w \quad (1)$$

where T_w is the total primary energy (TJ), carbon emission (ton CO_2e), or energy cost (10^4 \$) of water infrastructure type w; $E_{\alpha,o}^{w}$ is the amount of energy type α in its original form used for operating and maintaining water infrastructure type w each year (TJ/year); $E_{\alpha,c}^{w}$ is the amount of energy type α in its original form used for constructing water infrastructure w (TJ); α is the energy-type index; t_w is the life span of water infrastructure type w (here assumed to be 100 years for all w); and f_α is the primary energy factor (dimensionless), carbon emission factor (ton CO_2e/TJ), or energy price of energy type α (10^4 \$/TJ).

For each water management scenario, embodied energy, GHG emissions, and energy costs were assessed and normalized to the functional unit proportional to the different types of water infrastructure systems (water supply, wastewater treatment, and water reuse if applicable) implemented to meet projected water demand in 2030 for TB and SD. To do this, each impact category was assumed to be linearly related to the amount of water supplied on the basis of the fact shown by previous studies that economies of scale are negligible when the systems reach certain capacities.[53−55] Equation 2 provides the calculations of the impact categories:

$$U = \sum_{i} (t_w^i \times Q^i)/Q_d \quad (2)$$

where U is the unit primary energy (TJ/m^3), carbon emission (ton CO_2e/m^3), or energy cost (10^4 \$/m^3) for the given scenario; t_w^i is the unit primary energy (TJ/m^3), carbon emission (ton CO_2e/m^3), or energy cost (10^4 \$/m^3) of system type w treating water type i (wastewater included); Q^i is the total usage of water type i (m^3) in the given scenario; i is the water-type index; and Q_d is the total water demand (m^3) in the given scenario.

RESULTS AND DISCUSSION

Comparison of Different Types of Water Infrastructures in 2010 (Baseline). For both regions, TB and SD, direct energy is more significant than indirect energy in most types of water infrastructure systems, except for local surface water supply and wastewater treatment (Table 2). The primary contribution to this is the more extensive chemical use

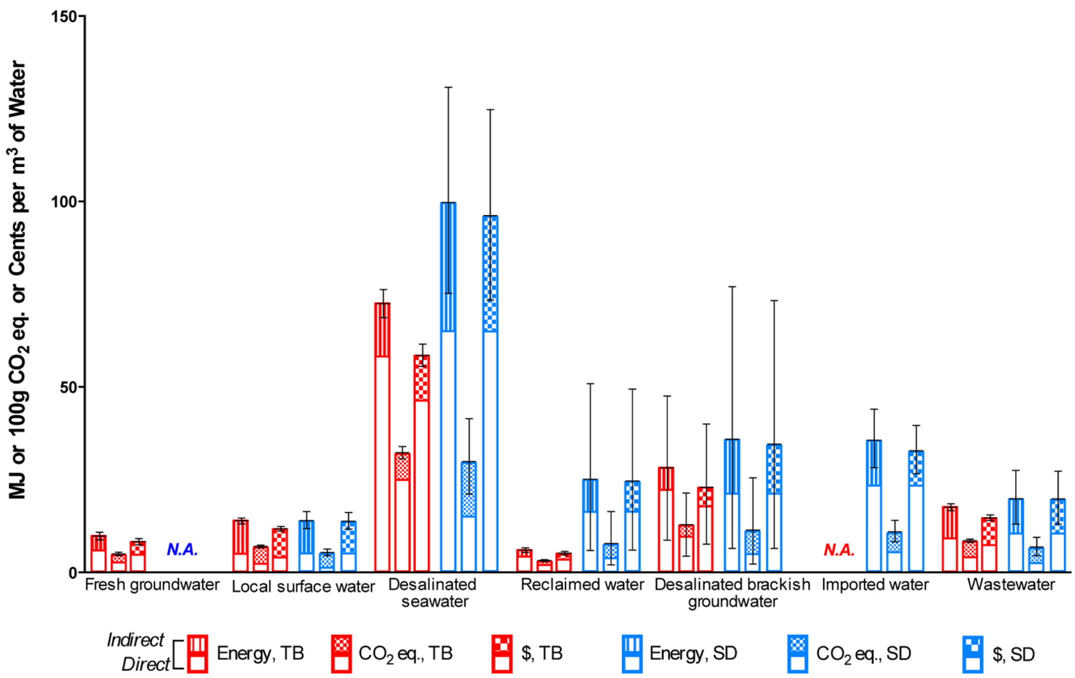


Figure 2. Embodied energy, GHG emissions, and energy costs of water infrastructure systems in the regions of Tampa Bay, Florida (TB), and San Diego, California (SD). (Note: the impacts of imported water were calculated as 15% for water from northern California and 85% for water from the Colorado River, as reported by Cohen et al.[42] as the proportional mix in 2010.)

dx.doi.org/10.1021/es405648x | Environ. Sci. Technol. 2014, 48, 5883−5891

in these two types of water infrastructure systems. Most of the SD systems have generally higher energy intensities than TB systems, especially in terms of seawater desalination and water reclamation. Seawater desalination is less energy-intensive in TB primarily because the salinity of the source water in TB (26 000 ppm) is substantially lower than that of typical seawater (33000–40000 ppm).[56] The higher water reclamation energy in SD is mainly caused by the difference between the topographies of the two regions,[57] as TB has relatively little elevation change while SD is more mountainous, leading to additional energy for pumping to redistribute the water to users.

Despite the more energy-intensive water in SD, GHG emissions are lower in most of the SD systems, except for water reclamation (Figure 2). More illustratively, replacing 1 m^3 of desalinated seawater with reclaimed water has a larger energy benefit in SD (75 MJ) than in TB (66 MJ) but has a larger carbon benefit in TB (2.9 kg CO_2e) than in SD (2.2 kg CO_2e). This is a result of the higher renewable contribution to the electricity grid in SD. While energy assessments are highly motivated by concerns over climate change and fossil fuel depletion, the dramatic difference between the trends of energy consumption versus GHG emissions for the two regions shows the undeniable importance of renewable energy utilization in the regional grid mix for electricity-intensive sectors, such as water infrastructure. Increasing the renewable energy composition is also desirable in terms of substantially reducing GHG emissions, as energy efficiency improvements through equipment upgrades and modifications in water infrastructure have been limited by technology innovation and water quality requirements.[58]

Energy costs are much higher in SD than in TB not only because of the higher energy intensity but also because the base electricity price is higher as a trade-off of the high renewable contribution to the grid mix[59] (Figure 2). This has two implications. First, optimized water infrastructure systems management can result in energy savings as well as a disproportionate cost savings. Again, upon replacement of 1 m^3 of desalinated seawater with reclaimed water, the energy savings in SD will be 12% higher than in TB, with a cost savings in SD that is 25% higher than in TB. Second, in the case of a carbon tax or a formal carbon market, optimized SD systems may gain carbon credits with the potential to generate additional revenue to offset costs. In that case, costs in the SD systems can be reduced, and they may be able to outperform the TB systems in both GHG emissions and economics.

Comparison of Future Water Supply Scenarios. The embodied energy, GHG emissions, and energy costs are higher in SD than in TB regardless of the future water supply source scenario (Figure 3). It is not unexpected for embodied energy and energy costs to be higher because these are higher for all types of water infrastructure systems in SD than in TB (Figure 2). While GHG emissions are lower in most types of SD water infrastructures, overall GHG emissions are higher because the low-GHG sources of groundwater and surface water cannot meaningfully contribute to meet the demand driving SD to rely on the more energy-intensive water sources, particularly importation (Figure 3).

Upon disaggregation of these general trends, a comparison of the scenarios shows that for each region, scenario S2 (maximizing desalination) has the highest embodied energy, GHG emissions, and energy costs while scenario S3 (maximizing water reclamation) has the lowest, with scenario S1 (maximizing traditional supplies) falling in between. This means that if water supply management is pursued along S3 rather than S2, totals of 5550 and 35 540 TJ of primary energy can be saved per year in TB and SD respectively. These represent 0.2% and 1.3% of the total electricity generation (in primary energy form) in Florida and California, respectively. Furthermore, totals of 0.25 and 1.05 million metric tons of GHG emissions can be reduced annually in TB and SD, respectively. On the basis of the average conversion reported by EPA,[60] these are equivalent to 48 000 and 206 000 typical passenger cars annually in TB and SD, respectively. Additionally, totals of \$45 million and \$341 million in annual energy costs can be saved in TB and SD, respectively. As these regions represent significant diversity in the energy, GHG emissions, and costs for these water infrastructure systems, it is likely that water reclamation is a more sustainable strategy than desalination in many other regions, as has been previously reported.[14]

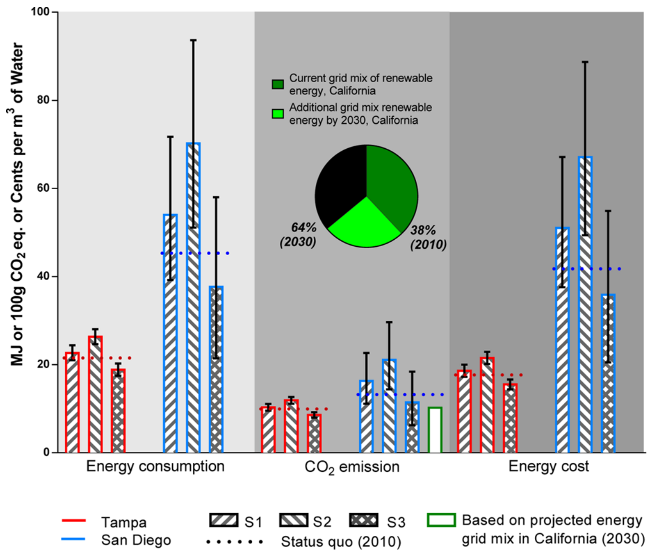


Figure 3. Embodied energy, GHG emissions, and energy costs of the future water management scenarios in the regions of Tampa Bay, Florida, and San Diego, California.

Additionally, S1 and S2 exhibit higher values than the current status quo in embodied energy, GHG emissions, and energy costs, which is primarily driven by the need to utilize more energy-intensive water sources to meet growing demand. Because of the energy–water nexus, the growing water demand exacerbates the stresses on energy resources and the environment in addition to the impacts on finite water sources. On the other hand, S3 has the potential to meet the increased demand with lower energy consumption, GHG emissions, and costs than the current status quo as a result of its reliance on wastewater reclamation, a lower-carbon water source.

Comparison of Future Energy Grid Scenarios. For the analysis presented in the previous section, the regional grid mixes in TB and SD were assumed to be the same in 2030 as in 2010. However, because the future water supply scenarios were carried in 2030, it is important for a holistic energy–water nexus analysis to consider the role of potential future electricity grid mix changes. This is especially pertinent to SD, given the state's vigorous plan to increase the renewable energy composition of the grid mix based on California's Global Warming Solutions Act of 2006 (AB32)[61] and Executive Order

S-3-05.[62] In order to comply with these legislative targets, California needs to reduce its GHG emissions to 80% below 1990 levels by 2050 while accommodating projected growth in its economy and population.[63] In order to achieve this goal, an 7.5% average annual increase of eligible low-carbon (i.e., renewable) feedstocks must be added to the grid mix, resulting in an average of 64% of low-carbon energy composition by 2030. With the assumption that GHG emissions are linearly related to the amount of fossil energy used (Table 3), the carbon emission factor of SD will be approximately 34.3 kg CO_2e/GJ in 2030. For the purpose of comparison, a new grid mix was established for 2030, and using the new carbon emission factor, the GHG emissions will be approximately 11% lower than the status quo calculated using the current grid mix for 2010.

It should be noted that this GHG emission reduction from the water infrastructure systems is much less significant compared with the fossil fuel composition reduction in the grid mix. That is, a 42% fossil fuel reduction in the grid mix has to be made in order to achieve this 11% GHG emission reduction. This is the result of the direct uses of coal, petroleum, and natural gas during the life cycle of water infrastructure systems (e.g., petroleum for material transportation and natural gas for heating) that are not reflected in changes to the electricity grid.

Uncertainty and Sensitivity Analyses. It is important to evaluate the impact of the uncertainties associated with different water infrastructure systems detailed in Table 2 on the results presented. Considering the extreme cases in which the impacts of one type of water infrastructure are maximized while the others are minimized allows for the choice of "winning" and "losing" water supply sources. Table 4 provides energy consumption results for such cases when the embodied energy of the water infrastructure indicated at the beginning of each row was assumed to have the maximum possible value in the range while all of the other water sources were assumed to have the lowest value in the range. The energy consumption trends for the three scenarios in each of the winner/loser cases still replicate the results reported in Figure 3 for both TB and SD, except when reclaimed was the "winner" (i.e., when the embodied energy for this water source was maximized). When the energy use of water reclamation is at its highest potential value and the energy uses of other water supply types (including seawater desalination) are at their lowest potentials, S3 (maximizing water reclamation) is almost comparable to S1 (maximizing traditional supply) in SD. However, even in this particular extreme case, maximizing water reclamation may still be preferable on the basis of other sustainability and resiliency criteria not captured in this analysis, including vulnerability of long-distance pipelines, depletion of finite resources, and impacts on ecological systems function.

Table 4. Extreme Cases of the Embodied Energy Consumptions of Future Scenarios Based on Uncertainties Associated with Each Type of Water Infrastructure

"Winning" water infrastructure type	Energy intensity in each extreme case (MJ primary energy/m^3 water)					
	Tampa Bay			San Diego		
	S1	S2	S3	S1	S2	S3
Fresh groundwater	22.3	26.0	18.8	*n.a.*	*n.a.*	*n.a.*
Local surface water	22.1	25.7	18.4	42.6	54.5	24.9
Desalinated seawater	21.8	26.0	18.0	52.9	78.8	24.6
Reclaimed water	21.7	25.3	18.5	44.7	56.6	44.8
Desalinated brackish groundwater	21.9	25.1	18.0	44.8	56.7	27.1
Imported water	*n.a.*	*n.a.*	*n.a.*	52.6	60.5	31.6

Furthermore, a sensitivity analysis on the water infrastructure life span was performed, and the percentage changes of embodied energy under life spans of 20, 50, and 150 years were examined (Figure 4). The embodied energy of water infrastructures increases exponentially with the reduction of their life span. When the life span is between 50 and 150 years, insignificant changes in embodied energy (<10%) in all types of water infrastructures in both regions were observed because the construction energy is minor compared with the operation and maintenance energy over this range; however, when the life span is further reduced to 20 years, the embodied energy increases significantly in most types of water infrastructures, except for the ones with high direct energy consumptions, such as seawater and brackish groundwater desalination in both regions and water reclamation in SD. The results imply that investing in the maintenance of existing infrastructures to extend their life spans is more energy efficient than constructing new infrastructures.

IMPLICATIONS

Region-specific conditions play an important role in determining the embodied energy, GHG emissions, and energy costs of water infrastructure systems for a given location, and therefore, the highest achievable benefits of water reclamation vary among regions. However, the current analysis suggests that maximizing water reclamation as a source augmentation strategy to meet future water demand can be more sustainable from an energy—water nexus perspective than relying on increasing traditional supply or seawater desalination. In view of the fact that the energy intensities of most water infrastructure systems fall well within the uncertainties of the systems modeled for SD,[64] this conclusion can be reasonably extended to other coastal regions. Further considerations for water reclamation include social considerations and emerging contaminants. Public acceptance and potential health risks have been the major obstacles in reusing treated wastewater for potable uses. Hence, to make the proposed future water supply scenarios more realistic, this study considered only nonpotable water reclamation, which has already taken place in the two study regions. However, further studies considering the life-cycle costs of quality control as well as the social effects of vast water reclamation, particularly for potable uses, are needed.

Considering not only future water supply scenarios but also future energy grid mix scenarios, it is suggested that increasing renewable energy sources in the regional electricity grid mix can significantly reduce GHG emissions for water infrastructure systems. However, this GHG reduction is limited by the water sector's dependence on fossil-fuel-based industries such as transportation and steel and cement production. Thus, in addition to striving for a "greener" grid, it is also important to consider investment in the electrification of current high-carbon, fossil-fuel-based industries and to continue to improve the fossil fuel use efficiency.

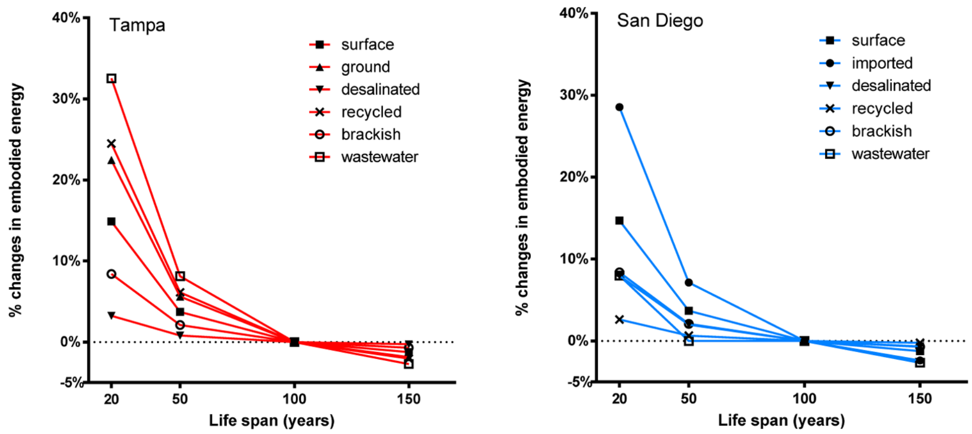


Figure 4. Percentage change in embodied energy for life spans of 20, 50, and 150 years in Tampa Bay, Florida, and San Diego, California. [Note: % change = (new embodied energy − original embodied energy)/(original embodied energy)].

It is also important to note that the same GHG emission reductions can be achieved by water efficiency improvements (e.g., water conservation) with less financial investment[44] than would be needed to reduce the carbon footprint of the energy sector. On the basis of the status-quo conditions (2010) for both the energy and water infrastructure systems in SD, the amount of water that needs to be conserved is estimated to be 197 000 m^3/day, which is well within the water conservation goal for the region.[31] This further demonstrates the benefits and critical need to assess the energy−water nexus when pursuing conversation and efficiency targets or goals for the individual infrastructure systems to ensure that the potentially most economical options are considered, even if they are outside the individual system boundaries.

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

Introduction of IO-based hybrid analysis; details of sample water infrastructures in TB and SD; calculation of electricity primary energy factors and electricity carbon emission factors in TB and SD; and life-cycle inventory for structural path analysis in IO-based hybrid analysis. This material is available free of charge via the Internet at http://pubs.acs.org.

■ AUTHOR INFORMATION

Notes

The authors declare no competing financial interest.

■ REFERENCES

(1) Bischel, H. N.; Simon, G. L.; Frisby, T. M.; Luthy, R. G. Management experiences and trends for water reuse implementation in Northern California. *Environ. Sci. Technol.* **2012**, *46*, 180−188.

(2) *Water−Energy Interactions of Water Reuse*; Lazarova, V., Choo, K.-H., Cornel, P., Eds.; IWA Publishing: London, 2012.

(3) Yüce, S.; Kazner, C.; Hochstrat, R.; Wintgens, T.; Melin, T. Water reuse versus seawater desalination—Evaluation of the economic and environmental viability. In *Water−Energy Interactions of Water Reuse*; Lazarova, V., Choo, K.-H., Cornel, P., Eds.; IWA Publishing: London, 2012; pp 243−256.

(4) *California Water Plan Update 2009*; California Department of Water Resources: Sacramento, CA, 2010; http://www.waterplan.water.ca.gov/cwpu2009/.

(5) *2010 Regional Water Supply Plan—Tampa Bay Planning Region*; Southwest Florida Water Management District: Brooksville, FL, 2011.

(6) Cheng, H.; Hu, Y.; Zhao, J. Meeting China's water shortage crisis: Current practices and challenges. *Environ. Sci. Technol.* **2009**, *43*, 240−244.

(7) National Academy of Sciences. *Water Reuse: Potential for Expanding the Nation's Water Supply through Reuse of Municipal Wastewater*; The National Academies Press: Washington, D.C., 2012; p 277.

(8) *2012 Annual Report, Real Value Water—Sustaining Our Qualitiy of Life, Driving Our Economy*; San Diego County Water Authority: San Diego, CA, 2013; http://www.sdcwa.org/annualreport/2012/.

(9) *Water Smart Landscapes for California—AB 2717 Landscape Task Force Findings, Recommendations, & Actions: Report to the Governor & Legislature*; California Urban Water Conservation Council: Sacramento, CA, 2005.

(10) Vince, F.; Aoustin, E.; Bréant, P.; Marechal, F. LCA tool for the environmental evaluation of potable water production. *Desalination* **2008**, *220*, 37−56.

(11) Raluy, R.; Serra, L.; Uche, J. Life cycle assessment of desalination technologies integrated with renewable energies. *Desalination* **2005**, *183*, 81−93.

(12) Friedrich, E. Life-cycle assessment as an environmental management tool in the production of potable water. *Water Sci. Technol.* **2002**, *46*, 29−36.

(13) Landu, L.; Brent, A. C. Environmental life cycle assessment of water supply in South Africa: The Rosslyn industrial area as a case study. *Water SA* **2007**, *32*, 249−256.

(14) Lyons, E.; Zhang, P.; Benn, T.; Sharif, F.; Li, K.; Crittenden, J.; Costanza, M.; Chen, Y. Life cycle assessment of three water supply systems: Importation, reclamation and desalination. *Water Sci. Technol.: Water Supply* **2009**, *9*, 439−448.

(15) Stokes, J.; Horvath, A. Life cycle energy assessment of alternative water supply systems. *Int. J. Life Cycle Assess.* **2006**, *11*, 335−343.

(16) Munoz, I.; Milà-i-Canals, L.; Fernández-Alba, A. R. Life cycle assessment of water supply plans in Mediterranean Spain. *J. Ind. Ecol.* **2010**, *14*, 902−918.

(17) Mo, W.; Nasiri, F.; Eckelman, M. J.; Zhang, Q.; Zimmerman, J. B. Measuring the embodied energy in drinking water supply systems: A case study in the Great Lakes Region. *Environ. Sci. Technol.* **2010**, *44*, 9516−9521.

(18) Mo, W.; Zhang, Q.; Mihelcic, J. R.; Hokanson, D. R. Embodied energy comparison of surface water and groundwater supply options. *Water Res.* **2011**, *45*, 5577−5586.

(19) Hancock, N. T.; Black, N. D.; Cath, T. Y. A comparative life cycle assessment of hybrid osmotic dilution desalination and established seawater desalination and wastewater reclamation processes. *Water Res.* **2012**, *46*, 1145−1154.

(20) Stokes, J. R.; Horvath, A. Energy and air emission effects of water supply. *Environ. Sci. Technol.* **2009**, *43*, 2680−2687.

(21) Rodriguez-Garcia, G.; Molinos-Senante, M.; Hospido, A.; Hernandez-Sancho, F.; Moreira, M.; Feijoo, G. Environmental and economic profile of six typologies of wastewater treatment plants. *Water Res.* **2011**, *45*, 5997−6010.

(22) Peters, G.; Rouse, K., Environmental sustainability in water supply planning—An LCA approach for the Eyre Peninsula, South Australia. Presented at the Fourth Australian Conference on Life Cycle Assessment, Melbourne, Australia, 2005.

(23) Lassaux, S.; Renzoni, R.; Germain, A. Life cycle assessment of water from the pumping station to the wastewater treatment plant. *Int. J. Life Cycle Assess.* **2007**, *12*, 118−126.

(24) Pasqualino, J. C.; Meneses, M.; Castells, F. Life cycle assessment of urban wastewater reclamation and reuse alternatives. *J. Ind. Ecol.* **2011**, *15*, 49−63.

(25) Meneses, M.; Pasqualino, J. C.; Castells, F. Environmental assessment of urban wastewater reuse: Treatment alternatives and applications. *Chemosphere* **2010**, *81*, 266−272.

(26) Ortiz, M.; Raluy, R.; Serra, L. Life cycle assessment of water treatment technologies: Wastewater and water-reuse in a small town. *Desalination* **2007**, *204*, 121−131.

(27) Mo, W.; Zhang, Q. Can municipal wastewater treatment systems be carbon neutral? *J. Environ. Manage.* **2012**, *112*, 360−367.

(28) Mo, W.; Zhang, Q. Energy−nutrients−water nexus: Integrated resource recovery in municipal wastewater treatment plants. *J. Environ. Manage.* **2013**, *127*, 255−267.

(29) Lundie, S.; Peters, G. M.; Beavis, P. C. Life cycle assessment for sustainable metropolitan water systems planning. *Environ. Sci. Technol.* **2004**, *38*, 3465−3473.

(30) Oki, T.; Kanae, S. Global hydrological cycles and world water resources. *Science* **2006**, *313*, 1068−1072.

(31) San Diego County Water Authority. Fact Sheets. http://www.sdcwa.org/publications (accessed Nov 15, 2013).

(32) San Diego County Water Authority. Seawater Desalination—The Carlsbad Desalination Project. http://www.sdcwa.org/sites/default/files/files/publications/desal-carlsbad-fs-single.pdf (accessed Sept 15, 2013).

(33) *Emissions & Generation Resource Integrated Database (eGRID) 2012*, version 1.0; U.S. Environmental Protection Agency: Washington, DC, 2012.

(34) U.S. Census Bureau. State & County QuickFacts. http://quickfacts.census.gov/qfd/states/12000.html (accessed Sept 16, 2013).

(35) Bureau of Economic Analysis. Economic Growth Widespread across Metropolitan Areas in 2012. http://www.bea.gov/newsreleases/regional/gdp_metro/2013/pdf/gdp_metro0913.pdf (accessed Oct 10, 2013).

(36) Goldstein, R.; Smith, W. *Water & Sustainability (Volume 4): U.S. Electricity Consumption for Water Supply & Treatment—The Next Half Century*; Electric Power Research Institute: Palo Alto, CA, 2002.

(37) Hill, C. P. Florida brackish water and seawater desalination: Challenges and opportunities. *Florida Water Res. J.* **2012**, 7.

(38) Crawford, R. H. Validation of a hybrid life-cycle inventory analysis method. *J. Environ. Manage.* **2008**, *88*, 496−506.

(39) Lenzen, M. Errors in conventional and input−output-based life-cycle inventories. *J. Ind. Ecol.* **2000**, *4*, 127−148.

(40) Lloyd, S. M.; Ries, R. Characterizing, propagating, and analyzing uncertainty in life cycle assessment: A survey of quantitative approaches. *J. Ind. Ecol.* **2007**, *11*, 161−179.

(41) Bullard, C. W.; Sebald, A. V. Monte Carlo sensitivity analysis of input−output models. *Rev. Econ. Stat.* **1988**, 708−712.

(42) Cohen, R.; Wolff, G.; Nelson, B. *Energy Down the Drain: The Hidden Costs of California's Water Supply*; Natural Resources Defense Council: Oakland, CA, 2004.

(43) Wilkinson, R. *Methodology for Analysis of the Energy Intensity of California's Water Systems, and an Assessment of Multiple Potential Benefits through Integrated Water−Energy Efficiency Measures*; California Institute for Energy Efficiency: Santa Barbara, CA, 2000.

(44) Klein, G.; Krebs, M.; Hall, V.; O'Brien, T.; Blevins, B. B. *California's Water−Energy Relationship*; California Energy Commission: Sacramento, CA, 2005.

(45) *Refining Estimates of Water-Related Energy Use in California*; California Energy Commission, PIER Industrial/Agricultural/Water End Use Energy Efficiency Program, CEC-500-2006-118; Navigant Consulting, Inc.: Sacramento, CA, 2006.

(46) Gowdy, J.; Miller, J. Technological and demand change in energy use: An input−output analysis. *Environ. Plann., A* **1987**, *19*, 1387−1398.

(47) *Measuring Energy Efficiency in the United States' Economy: A Beginning*; U.S. Energy Information Administration: Washington DC, 1995.

(48) U.S. Environmental Protection Agency. Unit Conversions, Emissions Factors, and Other Reference Data. http://www.epa.gov/appdstar/pdf/brochure.pdf (accessed Oct 29, 2013).

(49) U.S. Energy Information Administration. Annual Coal Report 2011. http://www.eia.gov/coal/annual/pdf/acr.pdf (accessed Oct 15, 2013).

(50) U.S. Energy Information Administration. Natural Gas Prices. http://www.eia.gov/dnav/ng/ng_pri_sum_dcu_nus_m.htm (accessed Sept 15, 2013).

(51) U.S. Energy Information Administration. Weekly Retail Gasoline and Diesel Prices. http://www.eia.gov/dnav/pet/pet_pri_gnd_a_epm0_pte_dpgal_w.htm (accessed Sept 15, 2013).

(52) U.S. Energy Information Administration. Electric Power Monthly. http://www.eia.gov/electricity/monthly/pdf/epm.pdf (accessed Oct 15, 2013).

(53) Côté, P.; Masini, M.; Mourato, D. Comparison of membrane options for water reuse and reclamation. *Desalination* **2004**, *167*, 1−11.

(54) McGivney, W.; Kawamura, S. *Cost Estimating Manual for Water Treatment Facilities*; John Wiley & Sons: Hoboken, NJ, 2008.

(55) Gumerman, R. C.; Culp, R. L.; Hansen, S. P.; Lineck, T. S. *Estimating Water Treatment Costs*; Environmental Protection Agency, Office of Research and Development, Municipal Environmental Research Laboratory: Washington, DC, 1979.

(56) Cooley, H.; Gleick, P. H.; Wolff, G. *Desalination, with a Grain of Salt: A California Perspective*; Pacific Institute: Oakland, CA, 2006

(57) U.S. Geological Survey. National Elevation Dataset. http://ned.usgs.gov/index.asp (accessed Nov 24, 2013).

(58) *Energy Efficiency in Water and Wastewater Facilities: A Guide to Developing and Implementing Greenhouse Gas Reduction Programs*; U.S. Environmental Protection Agency: Washington, D.C., 2013.

(59) Bryce, R. *The High Cost of Renewable-Electricity Mandates*; Manhattan Institute for Policy Research: New York, 2012; http://www.manhattan-institute.org/html/eper_10.htm.

(60) *Greenhouse Gas Emissions from a Typical Passenger Vehicle*; Office of Transportation and Air Quality, U.S. Environmental Protection Agency: Washington, DC, 2011.

(61) California Environmental Protection Agency Air Resources Board. Assembly Bill 32: Global Warming Solutions Act, 2006. http://www.arb.ca.gov/cc/ab32/ab32.htm (accessed April 25, 2014).

(62) State of California. Executive Order S-3-05, 2005; gov.ca.gov/news.php?id=1861 (accessed April 25, 2014).

(63) Greenblatt, J.; Wei, M.; Yang, C.; Richter, B.; Hanegan, B.; Youngs, H. *California's Energy Future—The View to 2050*; California Council on Science and Technology: Sacramento, CA, 2012.

(64) Sanders, K. T.; Webber, M. E. Evaluating the energy consumed for water use in the United States. *Environ. Res. Lett.* **2012**, *7*, No. 034034.