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Spatially-explicit water balance implications of carbon capture and sequestration

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ABSTRACT

Implementation of carbon capture and sequestration (CCS) will increase water demand due to the cooling water requirements of CO_2 capture equipment. If the captured CO_2 is injected into saline aquifers for sequestration, brine may be extracted to manage the aquifer pressure, and can be desalinated to provide additional freshwater supply. We conduct a geospatial analysis to determine how CCS may affect local water supply and demand across the contiguous United States. We calculate baseline indices for each county in the year 2005, and project future water supply and demand with and without CCS through 2030. We conduct sensitivity analyses to identify the system parameters that most significantly affect water balance. Water supply changes due to inter-annual variability and projected climate change are overwhelmingly the most significant sources of variation. CCS can have strong local effects on water supply and demand, but overall it has a modest effect on water balances.

1. Introduction

Carbon dioxide (CO_2) capture and sequestration (CCS) is increasingly discussed as a means to reduce greenhouse gas emissions and limit climate destabilization (IEA, 2013; IPCC, 2014). The implementation of CCS is expected to have varied effects on local water balances. For example, power plants equipped with CO_2 capture will require more cooling water than plants without CO_2 capture. Water withdrawal and consumption per kWh for power plants with CO_2 capture could be double that of plants without CO_2 capture, and power plant cooling already comprises 49% of all US water withdrawals (Macknick et al., 2011; Kenny et al., 2009). Conversely, if the captured CO_2 is injected into saline aquifers for long-term sequestration, brine may need to be extracted from the aquifer to manage the pressure within the geologic formation (Buscheck et al., 2012). The extracted brine may be desalinated and used as a freshwater resource, partially offsetting the increased water demand for power plant cooling.

CCS must be implemented on a very large scale to contribute significantly to climate change mitigation (Herzog, 2011). To avoid unintended consequences from deployment of CCS, decision-makers must consider its interrelations with water use (Sathre et al., 2012). Water stress varies geographically—both within the United States (US) and globally. Areas with greater water stress may be less suited for expansion of water-intensive activities such as electricity production and CO_2 capture. Water supply and demand also vary over different time scales, as technologies and behaviors change. Spatially and temporally dynamic factors that affect a region's water balance include demand from agriculture, industry, and domestic consumption; demand for electricity and the share of electricity produced by fossil fuels and other sources; the rate and extent of CCS deployment; potential extraction of brine to manage pressure in saline aquifers used for CO_2 sequestration; and the effects of climate change on water supply patterns.

Prior literature provides partial insight into the local relationships between CCS implementation and water supply and demand. NETL (2010b) estimated the increased aggregate water use due to several projected CCS deployment scenarios in the US through 2035, though provided no breakdown by geographic region, which

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is critical to understanding local water supply and demand. Tidwell et al. (2013) analyzed the water use implications of CCS retrofitting in water-stressed regions of the US. They showed that water consumption could increase by almost 1 million m^3 day^{-1}, but did not consider the potential additional water supply from brine extracted from saline reservoirs. Talati et al. (2014) examined the change in overall water use that would result from proposed CO_2 emission standards in the US, finding significant differences in water use between power plants using different fuels, cooling systems, and CO_2 capture percentages. However, they provided no information on regional differences, potential local water constraints, or water supply from extracted brine. Schakel et al. (2015) studied the effect of CCS deployment on water stress in Europe, finding potentially significant impacts by 2050 in some areas including the UK, Spain, and central and eastern Europe.

In this analysis, we explore how CCS implementation in the electrical power sector may affect the balance of water supply and demand in the US. We conduct a spatially-explicit analysis at the US county level to understand regional variations in water supply and water demand. Given the dynamic nature of the system across temporal and spatial dimensions, we explore how potential variations in local water balances due to CCS (resulting from both increased cooling water demand and brine availability) compare with other sources of variability such as climate change, population change, and water demand in other sectors. This analysis allows us to identify the main drivers of uncertainty and variability within the system, and begin to define the solution space of possible outcomes.

2. Methods

2.1. Modeling framework

We develop and apply a spatially explicit system model of current and projected water supply and demand. The geographic scope of the analysis is the contiguous US, with spatial resolution at the county level (identified using 3109 Federal Information Processing Standard (FIPS) codes). The temporal scope includes a baseline analysis of 2005 conditions, as well as scenario projections to 2030 to identify potential future water-related implications of CCS deployment. We analyze the local interactions between CCS deployment, water supply and water demand. On the demand side, we compare the magnitude of water flows associated with CCS to the water requirements of other sectors such as irrigation, industry and public water supply. On the supply side, we use water supply projections and saline aquifer characteristics to estimate potential water availability. We determine how scenario conditions and system parameters affect the water balance and water stress of each county through 2030. The modeling framework is shown schematically in Fig. 1.

We analyze the water flows within each county under three scenarios: (1) no CCS implementation; (2) CCS deployment without brine extraction; and (3) CCS deployment coupled with brine extraction and desalination. The water balance implications of CCS deployment are estimated based on prospective mass and energy flow modeling of fossil fuel-fired electricity generation systems with and without CCS integration. An example of this modeling output is shown in Fig. 2, which describes selected mass and energy flows associated with the operation of an illustrative 500 MW coal-fired power plant equipped with CCS for 1 min. This example figure serves to illustrate the interactions between energy systems and water supply and demand. In our analysis, each individual case is quantitatively modeled based on the unique characteristics of the power plant and saline aquifer, as described below.

2.2. Water supply

County-level freshwater supply (*WS*) is calculated as:

$$WS = WS_{WaSSi} + \Delta WS_{brine} + \Delta WS_{return} \quad (1)$$

where WS_{WaSSi} is the projected county-level groundwater and surface water supply for a given climate scenario based on the WaSSi model, ΔWS_{brine} is the change in county-level water supply due to brine extraction and desalination, and ΔWS_{return} is the change in county-level water supply due to altered water return flows as a consequence of changes in water demand. *WS* is calculated in units of m^3 day^{-1}.

Local water supply projections are based largely on the WaSSi model (Sun et al., 2008). The WaSSi model estimates local water supplies at the level of US Geologic Survey (USGS) 8-digit Hydrologic Unit Code (HUC) watershed regions. Water supply for each HUC region is the sum of surface water supply, groundwater supply, and return flows. Surface water supply is estimated using a hydrological water balance model that predicts water supply as a function of monthly precipitation received, potential evapotranspiration, land use type, canopy interception capacity, soil moisture content, and plant rooting depth (Zhou et al., 2008). Groundwater supply is based on historical annual groundwater withdrawal records from the USGS. Return flow is based on historical return flow rates (Solley et al., 1998) multiplied by water use in different sectors such as domestic, industrial, irrigation, and thermoelectric power generation.

Future water supply varies according to climate change projections, based on emission scenarios from the IPCC Special Report on Emissions Scenarios (SRES) (IPCC, 2000). Climate change projections in the WaSSi model are based on downscaled climate modeling by Coulson et al. (2010) comprising monthly precipitation, monthly means of daily maximum air temperature, and monthly means of daily minimum air temperature. Our base-case climate projection uses the SRES B2 emission scenario, while the projected climate change associated with the SRES A1B emission scenario is also analyzed in a sensitivity analysis. The WaSSi model uses data from the MK2 climate model of Australia's Commonwealth Scientific and Industrial Research Organisation (CSIRO). We consider a nine-year window around the nominal analysis year (i.e., the four preceding years, the nominal year, and the four following years) to incorporate the effects of inter-annual variability in water supply. The “average annual water supply” and the “minimum annual water supply” for the nine-year span are used in water balance calculations. Intra-annual (e.g. seasonal) variability is not considered in this analysis.

The geographic resolution of WaSSi water supply data is HUC watershed regions, while the water demand analysis is resolved into political county regions (FIPS regions). Accordingly, we intersect two national scale geographic information system (GIS) map coverages, one comprised of 3109 FIPS regions and the other comprised of 2106 HUC regions. ArcGIS software is used to first reproject the county coverage so that both county and HUC coverages are in the same coordinate system. The HUC and county coverages are combined in a single feature resulting in more than 15,000 separate polygons, and the area of these polygons is calculated. Analysis is limited to HUC polygons larger than 10 km^2 that lie within a given county, in order to reduce the number of polygons to a manageable number. Because the average area of a county in the contiguous US is 2572 km^2, we expect the error introduced by this simplification will be minimal. The proportion of each HUC that lies within a particular county is calculated by dividing HUC area by county area. A matrix is then created to allow the conversion of water supply at the HUC level to approximate water supply at the

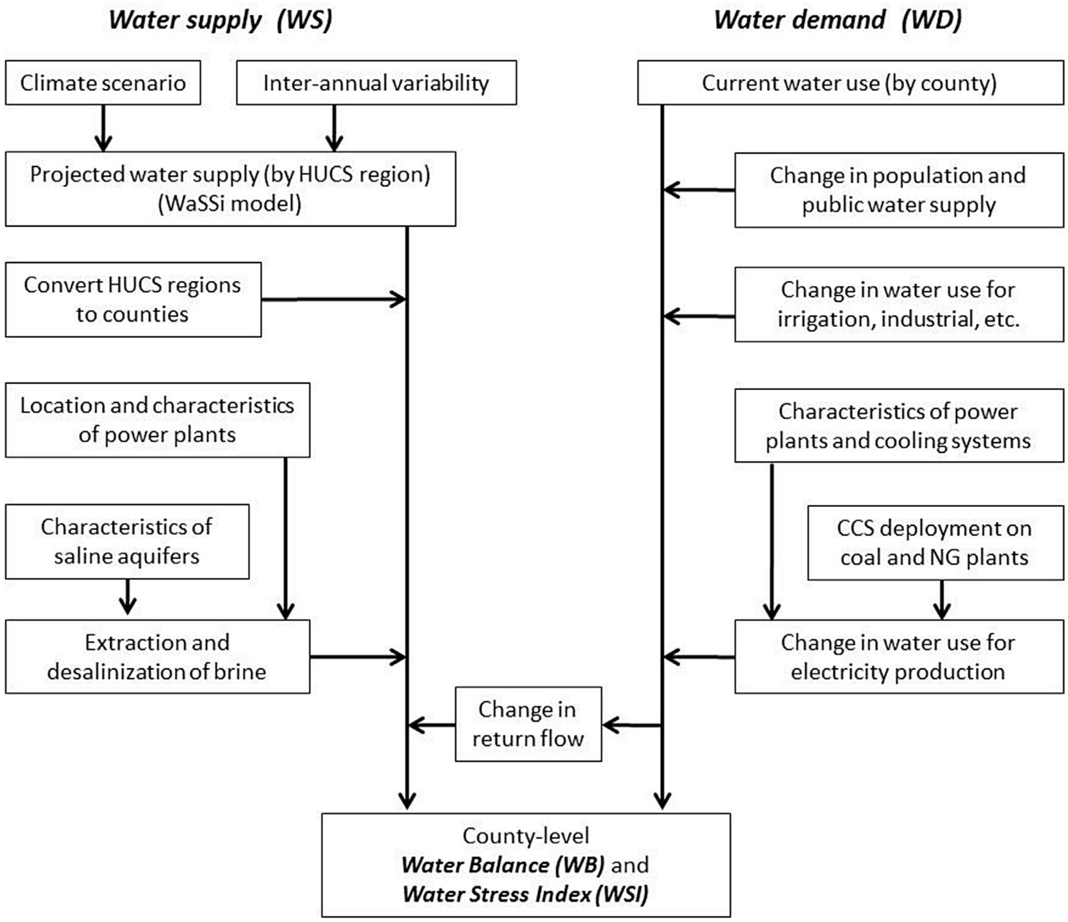

Fig. 1. Schematic diagram of the system modeling framework.

county level, based on the proportion of coinciding land area in each.

An additional potential source of freshwater is desalination of brine extracted from saline aquifers used for CO_2 sequestration. Locations of saline aquifers in the US are based on the NATCARB GIS database (NETL, 2012). Aquifers are included that underwent initial

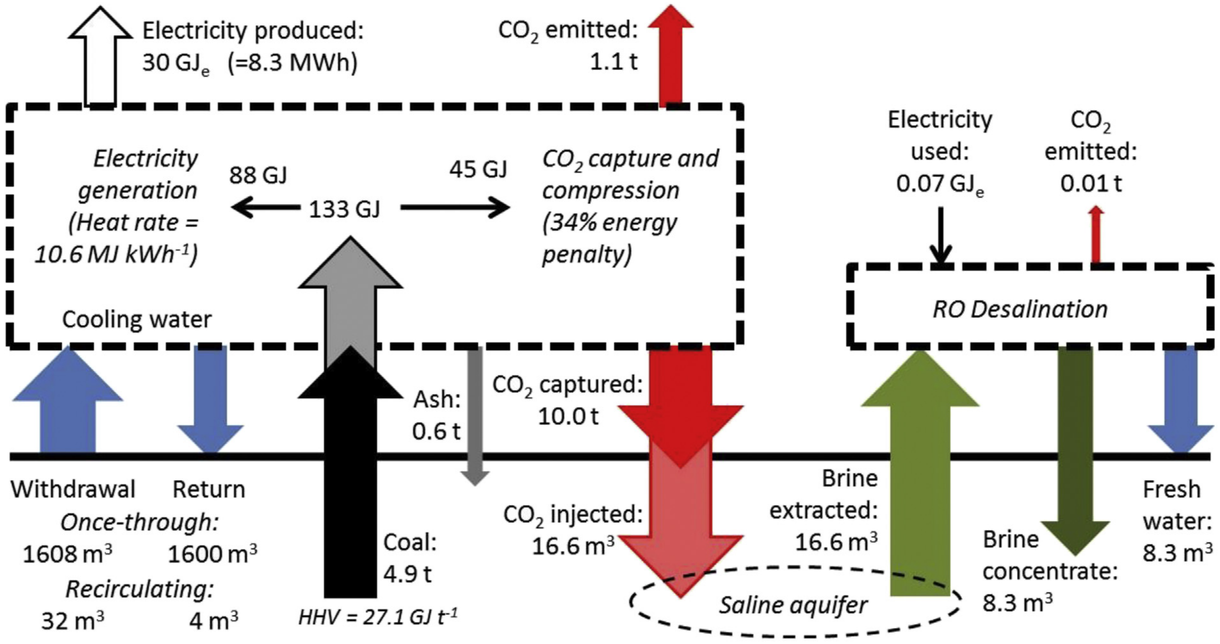

Fig. 2. Selected mass and energy flows associated with the operation of an illustrative 500 MW power plant with CCS for 1 minute. The dashed box on the left is the power plant with CO_2 capture, the dashed box on the right is the RO desalination plant, and the horizontal line is the earth's surface.

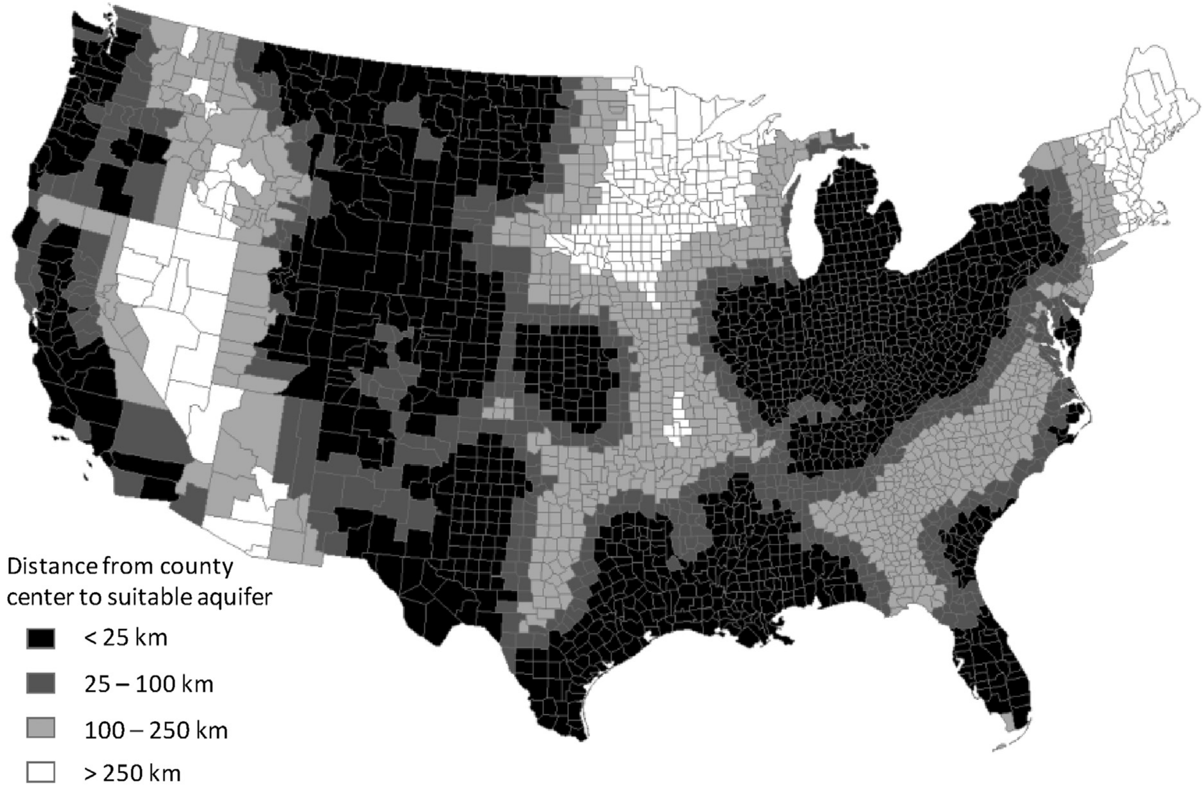

Fig. 3. Distance from the center of each county to the edge of the nearest saline aquifer suitable for CO_2 sequestration.

site screening by NATCARB; these aquifers have been assessed for storage potential and are listed by NATCARB as possible candidates for use in geologic CO_2 injection. Additional screening, selection, and characterization is necessary to ensure aquifers meet technical and non-technical criteria; NATCARB limits its initial screening to the geological criteria of capacity, injectivity, and integrity. Storage capacity is estimated based on the pore volume that can be occupied by injected CO_2. Formations that store supercritical CO_2, which has a critical point of 32 °C and 7.4 MPa, reside at different depths due to thermal and hydrodynamic conditions, but typically occur between depths of 800 and 3000 m (Bachu and Adams, 2003; NETL, 2010a). We characterize the total dissolved solids (TDS) concentrations within the aquifer using USGS data on produced water sampled at appropriate depths.

A review of CCS life cycle assessments found that reference distances for pipeline transportation of CO_2 ranged from 50 to 500 km (Corsten et al., 2013). While distances upward of 500 km have been reported for Norwegian projects directing CO_2 to oil and gas fields for enhanced recovery (Khoo and Tan, 2006), it is unlikely that the storage of CO_2 in saline aquifers would warrant such distances since there is no monetary incentive. Hasan et al. (2014) used an upper bound of 200 miles (322 km) in their supply chain network optimization model of CO_2 utilization options, including enhanced oil recovery, arguing that pipeline lengths greater than this were unlikely to be a part of the most economical supply chains in the United States. In our base-case analysis we limit CCS deployment to counties within 100 km of a suitable aquifer, beyond which we do not consider CCS as an option for power plants located in the county. We consider this appropriate for sequestration in saline aquifers where there is no enhanced fuel recovery to offset transportation costs. To determine the significance of this parameter, we also consider threshold distances of 25 km and 250 km in a sensitivity analysis. We use spatial analysis tools in ArcGIS software to determine the distance from the nearest region of a suitable saline aquifer to a county's center point. Fig. 3 shows the counties that are within 25 km, 100 km, and 250 km from a suitable aquifer.

To estimate quantities of extracted brine, we assume a density of supercritical CO_2 of 0.60 t m^{-3}, and we assume a brine-to-CO_2 displacement ratio of 1:1 by volume; in other words, for each m^3 of supercritical CO_2 injected into a saline aquifer, 1 m^3 of brine is extracted (Buscheck et al., 2012). A brine-to-CO_2 displacement ratio of 0.5:1 is also explored in a sensitivity analysis. The salinity of the extracted brine, measured as TDS, limits the fraction of freshwater that can be produced via reverse osmosis (RO) desalination. In our desalination analysis, we adapt data from Bourcier et al. (2011) and Aines et al. (2011) and assume that brines with TDS greater than 100 g L^{-1} cannot be treated by standard RO. We make a conservative assumption that brines with TDS less than 50 g L^{-1} can be treated with a 50% recovery fraction of freshwater, meaning that for every 2 L of brine treated, 1 L of freshwater and 1 L of concentrated brine are produced. For brines with TDS less than 100 g L^{-1} and greater than 50 g L^{-1} we use Equation (2) to determine the recovery fraction:

$$Recovery\ Percent = \left(1 - \frac{[TDS]}{[100\ g/L]}\right) \times 100\% \qquad (2)$$

where *Recovery Percent* is the maximum recovery fraction achievable using current RO technology, and [*TDS*] is the average TDS concentration of brine sampled in an aquifer. This equation reflects current RO membrane thresholds for osmotic pressure (Bourcier et al., 2011). It implies, for example, that brine with TDS of 80 g L^{-1} can be treated with a 20% freshwater recovery fraction. We assume that the concentrated brine produced by the desalination process is disposed of in a manner that does not affect the freshwater balance, e.g., by reinjection. For each county, quantities of freshwater recovered from brine desalination are added to quantities of other freshwater from the WaSSi model to determine the total water supply.

2.3. Water demand

County-level freshwater demand (*WD*) is calculated as:

$$WD = WD_{current} + \Delta WD_{electricity} + \Delta WD_{population} + \Delta WD_{other} \tag{3}$$

where $WD_{current}$ is the latest available actual county-level water demand, $\Delta WD_{electricity}$ is the change in water demand due to thermoelectric power plant deployment with and without CCS, $\Delta WD_{population}$ is the change in county-level public and self-supplied domestic water demand due to projected population change, and ΔWD_{other} is the change in water demand due to projected changes in water use for irrigation, industrial, livestock, aquaculture and mining. *WD* is calculated in units of m^3 day^{-1}.

Annual water use in 2005 within each county is based on USGS data describing water withdrawals in 2005 for public supply, self-supplied domestic use, irrigation, livestock, aquaculture, industrial, and mining (Kenny et al., 2009). The USGS report also includes coarse data on water used for thermoelectric power generation, though we choose to instead use power plant data from Ventyx (2012), as described below, due to its greater granularity. In 2005, water withdrawals in the US averaged approximately 1.5 billion m^3 per day (Kenny et al., 2009). Domestic and commercial uses made up only 12% of total US withdrawals. The sectors responsible for the majority of water use are thermoelectric power generation and agriculture, contributing 49% and 31% of total water withdrawals, respectively. Other sectors contribute smaller fractions, including industrial facilities (4%), aquaculture (2%), mining (1%), and livestock (<1%) (Kenny et al., 2009).

We adapt the 2005 annual data to account for projected future changes in county-level water use by 2030. Exploring how water use in various sectors has changed in recent history and the factors driving those changes may offer insight into how water demand will evolve in the coming decades. Between 1950 and 1980, total US water withdrawals grew at a rate that significantly outpaced population growth, peaking at 1.6 billion m^2 per day. More recently, however, water use intensity (measured, for example, in m^3 of water per unit of economic output) has decreased steadily in most sectors (Brown, 2000). This has resulted in total water use remaining fairly stable despite increased economic activity. For example, water resource limitations and federal regulations resulted in increased use of recirculating cooling systems for power plants, which withdraw only about 3% of the water used in once-through cooling systems, per unit of power output (Macknick et al., 2011). Industrial water use has been similarly impacted. Because water is often used to transfer heat within an industrial facility and remove waste heat, energy efficiency improvements provide the indirect benefit of reducing water requirements (Ellis et al., 2001). The result of these changes has been an overall decrease in US water withdrawals since 1980, despite increasing population and economic activity.

For water used for public supply and self-supplied domestic use in 2030, we estimate changes based on population and *per capita* water use. County population projections are based on Zarnoch et al. (2010) who developed three sets of population growth projections (low, medium, and high). We use the medium growth projection in our base-case modeling, and the low and high growth projections in a sensitivity analysis. Our base-case scenario assumes that *per capita* water use remains constant, thus county-level water use varies linearly with changes in county population; in a sensitivity analysis we consider the effects of a 20% increase in *per capita* water use. Our base case further assumes that county-level water use for irrigation, livestock, aquaculture, industrial, and mining purposes remain constant, based on past trends of water-use efficiency (Brown, 2000; Kenny et al., 2009). To determine the significance of changes in water use in these sectors, we increase water use in each of these sectors by 20% in a sensitivity analysis.

To estimate the current and future water use in the thermoelectric power sector, we develop a dataset of coal- and natural gas-fired power plants in the contiguous US, based primarily on Ventyx (2012) data. The dataset includes all power plants that emitted at least 100,000 t CO_2 in 2005. Carbon capture is less economically feasible at scales smaller than this plant size (IPCC, 2005). In total, 757 power plants are included, of which 355 are fueled by coal and 402 are fueled by natural gas. Collectively, the coal-fired plants produced a total of 1826 TWh of electricity and emitted 1785 Mt of CO_2 in 2005, for an average emission intensity of 0.98 tCO_2 MWh^{-1} The gas-fired plants collectively produced a total of 683 TWh of electricity and emitted 384 Mt of CO_2 in 2005, for an average emission intensity of 0.56 tCO_2 MWh^{-1}. We assume future power generation will be located at the same sites, due to infrastructural inertia within the sector. Modeled future changes in overall electricity production are based on Annual Energy Outlook (AEO) regional projections for 2030 (EIA, 2011). We scale the current electricity production in each county in proportion to the AEO projections for the NERC region the county is in, for both coal- and gas-fired production.

We develop a set of criteria for determining which of the 757 power plants are suitable for retrofitting with CO_2 capture equipment (MIT, 2009; IEA, 2012). Plants that satisfy all of the following four criteria are deemed suitable:

1. Nameplate generating capacity of 200 MW or more;
2. Capacity factor in 2005 of 50% or greater (capacity factor of a power plant is defined as the total amount of electricity produced during a period of time, divided by the amount of electricity the plant would have produced if operated continuously at full capacity);
3. Average heat rate of 12.7 MJ kWh^{-1} or less (heat rate of a power plant is defined as the heat value of fuel required to produce a unit of electricity);
4. Construction year of 1960 or later.

The first criterion is due to the economy of scale of CCS installations, making larger power plants more attractive than smaller ones. The second is because CO_2 capture facilities are best operated continuously, rather than intermittently. The third restricts CCS retrofitting to more efficient power plants, which can best benefit from the investment. The fourth criterion removes older power plants from retrofitting consideration, because they are close to retirement age.

Of the total number of 355 coal-fired plants, 288 have adequate nameplate capacities, 286 have adequate capacity factors, 292 have adequate heat rates, 238 were built in 1960 or later, and 168 plants meet all four criteria for CCS retrofitting. Of the 402 gas-fired plants, 349 have adequate nameplate capacities, 84 have adequate capacity factors, 336 have adequate heat rates, 349 were built in 1960 or later, and 49 plants meet all four criteria. Thus, a total of 217 plants meet all criteria for CCS retrofitting, of which 168 are coal-fueled and 49 are natural gas-fueled. Collectively these plants emitted about 1350 Mt of CO_2 in 2005, or about 62% of total emissions from all 757 plants. Fig. 4 shows the locations and CO_2

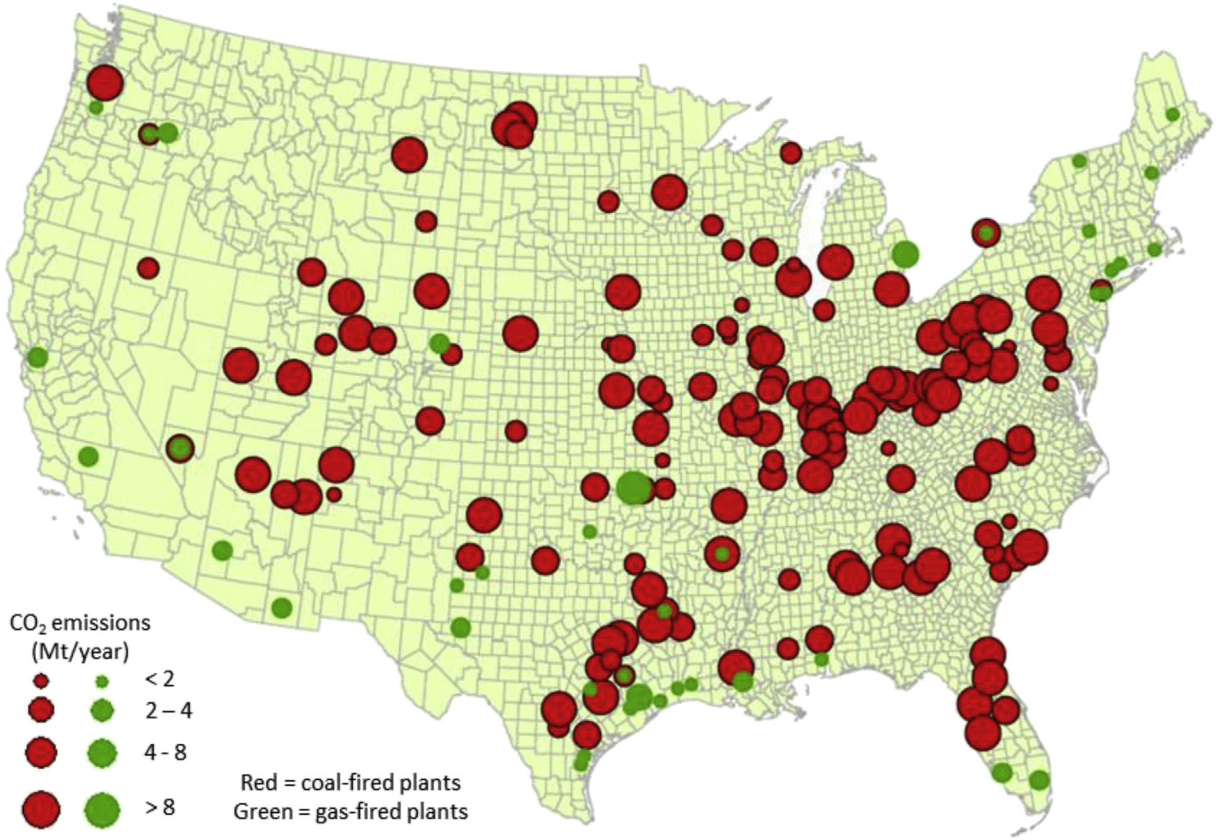

Fig. 4. Locations of 217 coal-fired (red) and natural gas-fired (green) power plants that meet plant-level criteria for CO_2 capture retrofitting. Size of circle corresponds to amount of CO_2 emission in 2005. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

emissions of these power plants. In addition to these plant-level criteria, the distance from the county to the nearest suitable sequestration formation (Fig. 3) also determines suitability for CO_2 capture in any given plant. A total of 141 power plants meet all four plant-level criteria for CCS retrofitting and are within 100 km from a suitable aquifer; these plants emitted about 1025 Mt of CO_2 in 2005.

These selection criteria are relaxed in a sensitivity analysis to determine the significance of a greater number of retrofitted plants, including all plants larger than 100 MW nameplate capacity (instead of 200 MW), with a capacity factor greater than 40% (instead of 50%), a heat rate less than 13.7 MJ kWh^{-1} (instead of 12.7 MJ kWh^{-1}), and built after 1955 (instead of 1960). Under these less stringent criteria, an additional 69 coal-fired plants and 51 gas-fired plants can be retrofitted.

Our power plant dataset also includes information on types of plant cooling systems, such as rates of water withdrawal, discharge, and consumption, and the source of the cooling water (Ventyx, 2012). Cooling system data for 2005 are available for plants responsible for 94% of the total CO_2 emissions. 2005 cooling system data are not available for the remaining plants, so we use USGS data for plants responsible for 4% of CO_2 emissions and proxy average cooling data for the remaining 2% of plants. We assume that water withdrawals per kWh for plant cooling will increase by 80% when CO_2 capture equipment is installed based on Zhai et al. (2011), which is broadly consistent with the findings of Macknick et al. (2011). We further assume that the CCS energy penalty, defined as the percent increase in fuel input per unit of delivered electricity, is 35% (Sathre et al., 2011). The cooling water increase and the energy penalty are varied in a sensitivity analysis. The total CO_2 production changes proportionally with the fuel use, and we assume that 90% of the CO_2 in the flue gas is captured and sequestered. Water use by nuclear power plants is assumed to remain unchanged. Water use by power plants that use saline cooling water is not accounted for in the analysis.

A portion of all increased water use is assumed to return to the hydrologic system for further use. This return flow amount is based on Solley et al. (1998) and is 80.8% for public supply and self-supplied domestic water, 85.2% for self-supplied industrial water, 39.3% for irrigation, livestock and aquacultural water, 85.2% for mining water, and 97.5% for thermoelectric cooling water. These return water flows are included as a water supply component in Equation (1).

2.4. Water balance and Water Stress Index

For each county under each scenario we calculate the freshwater balance, which we define as:

$$WB = WS - WD \tag{4}$$

where *WB* is the water balance, *WS* is the water supply, and *WD* is the water demand.

Following the approach developed by Sun et al. (2008), we also calculate a Water Stress Index (*WSI*) defined as:

$$WSI = \frac{WD}{WS} \tag{5}$$

The WSI is dimensionless, and is the ratio of county-level freshwater demand and freshwater supply. A lower value implies less water stress while a higher value implies greater water stress. A WSI of unity means all available freshwater is used; a value greater than unity implies that water is imported into the region from outside sources.

3. Results

Fig. 5 shows the 2005 WSI for US counties, considering average annual water supply and minimum annual water supply. Not surprisingly, water stress is higher in the minimum annual water supply scenario. The area most critically affected by water stress is the desert southwest US, but highly populated counties in other regions also experience local water stress. Comparison between Figs. 4 and 5 show that many isolated cases of high water stress are likely due to cooling water use by thermoelectric power plants.

Fig. 6 shows the projected WSI for all counties in 2030 with average and minimum annual water supplies, not including the effects of CCS implementation. Overall, the projected future changes in water stress through 2030 are modest. There are some cases of increased water stress, particularly in the southwest. Increased water stress is more commonly seen in years of minimum water supply. There are also several counties in which the WSI is projected to decrease by 2030.

Fig. 7 shows the increase in WSI due to the additional water use for the implementation of CCS, not including brine extraction and desalination, in counties in 2030 under average and minimum annual water supply conditions. There are several counties, primarily in the southwest, that may experience significant local water stress due the implementation of CCS. This effect is more pronounced under conditions of minimum water supply.

Fig. 8 shows the reduction of WSI due to the additional water supply from brine extraction and desalination, relative to CCS use without brine extraction, in counties in 2030 under average and minimum annual water supply conditions. The additional freshwater available from extraction and desalination of brine is shown to have a greater impact on WSI under conditions of minimum water supply. In absolute terms, the change in WSI due to brine extraction is less than the change due to CCS implementation.

Table 1 lists the results of a sensitivity analysis of various system parameters. Four indicators are quantified for each change in parameter value: nationwide water balance (km^3 day^{-1}), defined as the total water supply minus the total water demand; nationwide mean and median of Water Stress Indices of all counties; and the number of counties with WSI greater than 1.

The largest source of variation in water stress, by far, is natural variability in water supply between years of average precipitation

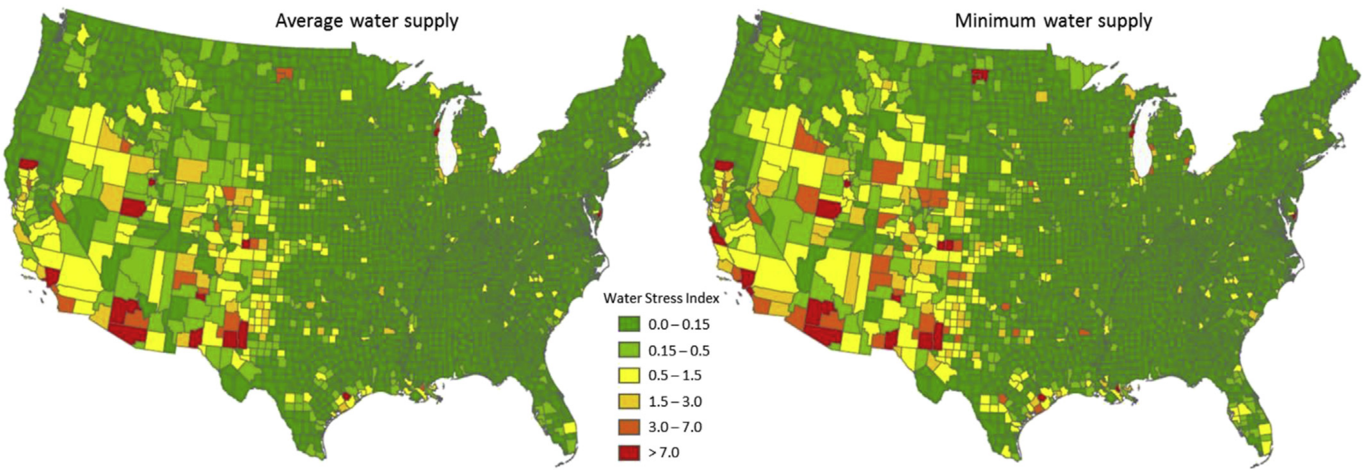

Fig. 5. Water Stress Index for counties in 2005, with average annual water supply (left) and minimum annual water supply (right).

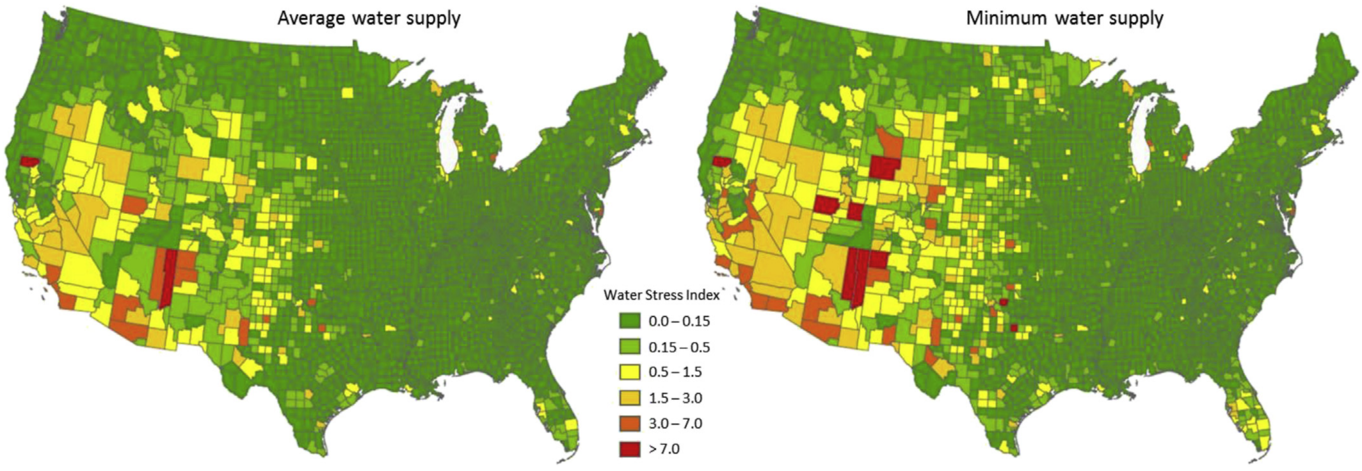

Fig. 6. Water Stress Index for counties in 2030 with no CCS implementation, with average annual water supply (left) and minimum annual water supply (right).

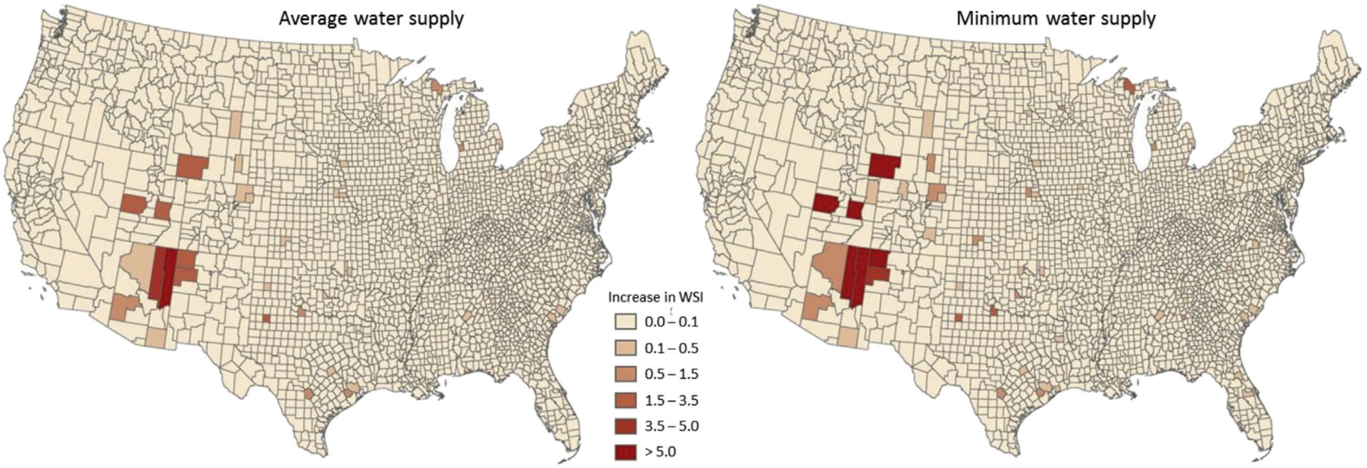

Fig. 7. Increase in Water Stress Index for counties in 2030 due to implementation of CCS without brine extraction and desalination.

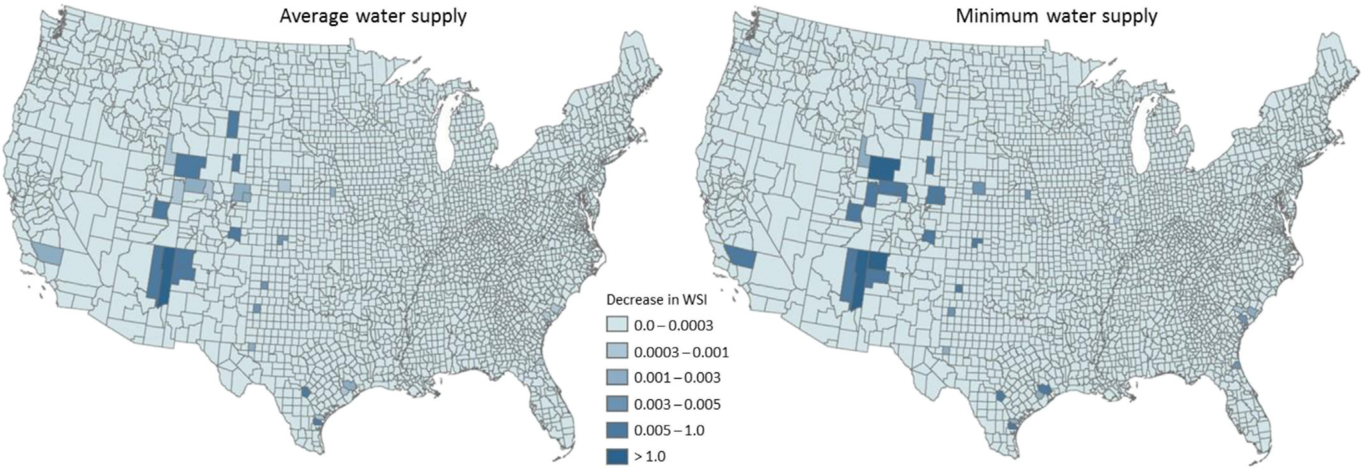

Fig. 8. Decrease in Water Stress Index for counties in 2030 due to brine extraction and desalination (relative to WSI of CCS without brine extraction).

Table 1
Values of key indicators under base-case conditions, and change from base case due to variation of one parameter at a time.

Parameter	Base-case parameter value	Adjusted parameter value	Water balance (km^3/day)	Water Stress Index		Number of counties with WSI > 1
				Mean	Median	
Base-case	**See below**	**N/A**	**95.040**	**0.117**	**0.0081**	**75**
Change from base case due to variation of each parameter:						
Annual precipitation	Average	Minimum	−34.9571	+0.09255	+0.00723	+89
Climate change scenario	SRES B2	SRES A1B	+4.6167	−0.02696	−0.00112	−23
Irrigation use	Unchanged	+20%	−0.0686	+0.00362	+0.00022	+9
Per capita public and domestic use	Unchanged	+20%	−0.0099	+0.00110	+0.00073	0
CCS deployment	Yes	No	+0.0092	−0.00833	−0.00013	0
Plants retrofitted with CCS	217	337	−0.0062	+0.00126	+0.00004	0
CO_2 capture water use	+80%	+40%	+0.0053	−0.00052	−0.00004	0
Aquaculture use	Unchanged	+20%	−0.0043	+0.00324	+0.00008	0
Industrial use	Unchanged	+20%	−0.0022	+0.00014	+0.00014	0
Population growth	Medium	High	−0.0022	+0.00041	+0.00010	0
Population growth	Medium	Low	+0.0014	−0.00037	−0.00011	0
Brine extraction and use	Yes	No	−0.0013	+0.00005	0.00000	0
Livestock use	Unchanged	+20%	−0.0011	+0.00017	+0.00005	0
Brine:CO_2 volume ratio	1:1	0.5:1	−0.0007	+0.00003	0.00000	0
Max distance to aquifer	100 km	250 km	+0.0005	−0.00002	0.00000	0
Mining use	Unchanged	+20%	−0.0003	+0.00008	+0.00003	0
Max distance to aquifer	100 km	25 km	−0.0002	+0.00001	0.00000	0
CCS energy penalty	35%	20%	−0.0001	+0.00001	0.00000	0

and minimum precipitation. Years of minimum precipitation have a national total water balance that is one-third lower than average years, with more than double the number of counties with WSI greater than 1. Another important source of variation is the extent of future climate change. Altering the climate change projection from the IPCC SRES B2 (i.e., the base case) to the A1B emission scenario results in a significant increase in water balance. This is due to the difference in future climate patterns between the scenarios, with the A1B scenario having higher average precipitation.

Changing the water use for irrigation has a moderately large effect, followed by the *per capita* water use of public and domestic supplies. Population growth rate, CCS energy penalty, and variation in water use for livestock, aquaculture, and industrial purposes have little impact on the system indicators. Broadening the criteria for which plants are retrofitted with CO_2 capture equipment results in a modest decrease in water balance and increase in water stress. The allowable distance between the power plant location and the geological sequestration formation is found to be less significant. At our base-case threshold of 100 km, CO_2 capture can be implemented at 141 of the 217 plants that meet the technical criteria established for suitability for retrofitting. If the maximum distance to an aquifer is extended to 250 km, CO_2 capture can occur at 179 plants. At a threshold distance of 25 km, CO_2 capture can occur at only 112 of the plants.

4. Conclusions

In this analysis we have considered how CCS implementation affects water stress, and identified areas in which water stress may constrain CCS implementation. Because water stress varies from place to place, we have conducted a geospatial analysis detailing the county-level balances of water supply and demand across the contiguous United States. Our focus has been to identify and understand the major sources of uncertainty and variation regarding the water-CCS nexus.

We find that CCS can strongly affect freshwater supply and demand in specific locations, but overall it has a moderate effect on water balances. Water supply changes due to inter-annual variability and longer-term climate change effects are the most significant sources of water stress. The use of extracted brine to overcome local water constraints may reduce local water stress compared to what would be possible without brine extraction. The importance of extracted brine increases as the water supply becomes more limited, a condition that is increasingly likely to occur as future climate becomes more unstable.

There are several issues that introduce uncertainty and may affect the results of this analysis. For example, the WaSSi water supply model does not consider water storage (e.g. in reservoirs), which could moderate inter-annual variability in water supply and thus reduce the differences in results between average and minimum water supply years. Spatially-explicit data on US dams are available (US Corps of Engineers, 2015)) and could be added to this analysis to improve validity of results. Furthermore, the contributions of return flow to total water supply are held constant in our modeling, though return flow quantities may vary due to future changes in water use efficiency.

We have used regional climate projections from the CSIRO MK2 climate model. To determine the significance of the particular climate model, updated projections from other climate models could be compared. In all cases, however, the definitive effects of future climate change on water supply will be uncertain, and include both spatial variation and temporal variation. In this analysis we have defined the minimum annual water supply for each HUC unit as the lowest value among the 9-year sampling period. This definition represents an extreme case of simultaneous drought across the entire country. In practice, drought in some HUC units will likely be moderated by average or above-average water supply in other, upstream HUC units. Sampling each county separately would account for this factor, though some measure of regional correlation in weather and climate should be expected.

Although we consider only one management option for extracted brine—desalination for use as freshwater—a range of other potential uses for brine exist such as mineral recovery, algae ponds, and de-icing salt production (Breunig et al., 2013). Extracted brine can also potentially be used directly for power plant cooling without desalination (Kobos et al., 2011). In addition, brine extraction could have negative effects on water balances if freshwater is needed for dilution. The significance of these potentials should be explored in future work.

In a more comprehensive analysis, water stress could be considered as a proxy for the cost or value of water. Local water stress could be used as a criterion for weighing the need or likelihood of applying various options. For example, in a water-stressed area, water will be more valuable and it may be more economically feasible to use brine as a water source. The attractiveness of using desalinated brine as a freshwater source would likely depend on the water stress of a region. Areas with greater water stress may be more likely to transport CO_2 and brine across further distances. The potential impacts on water balance of dry cooling methods, which are generally more expensive than water-based cooling methods, could also be elaborated.

This analysis is offered as an initial exploration of the potential local water implications of CCS deployment. Despite the uncertainties involved, we have identified overall trends that appear to be robust. We have modeled the system drivers for county-level water balance, and quantified the effects that CCS and brine desalination may have on local water stress. Variation in water supply due to inter-annual variability and projected climate change are by far the most significant sources of water stress change, though CCS can also strongly affect water supply and demand in specific counties. As the local water supply becomes more limited, the additional freshwater supply made available from desalinated brine becomes more important.

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