ANL/EVS/TM-14/1

Quantitative Assessment of Options for Managing Brines Extracted from Deep Saline Aquifers Used for Carbon Storage

Environmental Science Division

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prepared by
C.B. Harto
Environmental Science Division, Argonne National Laboratory

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NOTATION

The following is a list of acronyms, abbreviations, and units of measure used in this document.

GENERAL ACRONYMS AND ABBREVIATIONS

Argonne	Argonne National Laboratory
BOR	Bureau of Reclamation
CCS	carbon capture and storage
CMUGDI	Carnegie Mellon University Green Design Institute
CO_2	carbon dioxide
CO_2e	carbon dioxide equivalent
DOE	U.S. Department of Energy
EIO-LCA	economic input-output life cycle assessment
EPA	U.S. Environmental Protection Agency
GHG	greenhouse gas
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation
LCA	life cycle assessment
MED	Multi-Effect Distillation
MSF	Multi-Stage Flash
MVC	Mechanical Vapor Compression
NORM	naturally occurring radioactive materials
NREL	National Renewable Energy Laboratory
RO	reverse osmosis
ROW	right-of-way
SCTRWPG	South Central Texas Regional Water Planning Group
TDS	total dissolved solids
UIC	underground injection control

UNITS OF MEASURE

bbl	barrel(s)
BTU	British thermal unit
°C	degree(s) Celsius
ft	foot (feet)
g	gram(s)
gal	gallon(s)
gpm	gallon(s) per minute
in.	inch(es)
kWh	kilowatt hour(s)
lb	pound(s)
m^3	cubic meter(s)
MCF	million cubic feet
mi	mile(s)
mi^2	square mile(s)
mpg	miles per gallon
MW	megawatt(s)
ppm	part(s) per million
psi	pound(s) per square inch
psig	pound(s) per square inch gauge

GLOSSARY

Active Reservoir Management: A term referring to the active management of a carbon storage reservoir through the extraction and/or injection of water into the reservoir.

Area of Review: The area surrounding the storage project that may be impacted by elevated pressure resulting from injection activity. The area of review is important for permitting and influences the extent of ongoing monitoring requirements for long-term storage projects.

Brackish Water: Water that has higher salinity than freshwater but typically lower salinity than seawater. While there is no official range, some organizations consider the upper limit for brackish water to be 10,000 ppm total dissolved solids (TDS).

Brine: A solution of salt and water. It is often used as a generic term for a range of saline waters. In other cases, it is used more specifically to describe saline waters with TDS greater than seawater. In this report, it is used in the more ambiguous sense for any saline water source.

Economic Input-Output Life Cycle Assessment (EIO-LCA): A financially based LCA methodology that utilizes aggregated, sector-level data to estimate impacts. This methodology often suffers from aggregation error associated with the aggregated sector-level approach.

Extracted Water: Water removed from a geological formation receiving carbon dioxide (CO_2) for long-term storage.

Hybrid LCA: An LCA methodology that combines both process and EIO-LCA approaches to generate a more accurate and complete accounting of life cycle impacts.

Injectivity: The ability of an injection well or formation to receive fluid. Formations with greater injectivity allow for higher injection flow rates at a given injection pressure.

Life Cycle Assessment (LCA): LCA is a general approach to estimating the environmental impacts of a product or process from "cradle to grave."

Process LCA: The most common LCA methodology that relies upon direct estimation of impacts for individual processes. This methodology often suffers from cut-off error due to data limitations and processes left outside of the system boundary.

Recovery Ratio: The production rate of treated water divided by the flow rate of feedwater into a treatment system.

Saline Water: Water with salt or TDS content. Although the term can be used for any water with higher salinity than freshwater (i.e., brackish water, seawater), it is often used to describe water with higher salinity than brackish water.

Seawater: Water from the ocean or other saltwater surface water body. The TDS of seawater is typically around 35,000 to 45,000 ppm depending upon location and weather.

Total Dissolved Solids (TDS): The total concentration of all dissolved chemical constituents in a solution. It is often used to describe the concentration of salts and other minerals in a water source.

Treatment Rate: The flow rate of water into a treatment system.

QUANTITATIVE ASSESSMENT OF OPTIONS FOR MANAGING BRINES EXTRACTED FROM DEEP SALINE AQUIFERS USED FOR CARBON STORAGE

ABSTRACT

Active reservoir management, in which brine is extracted from deep saline aquifers utilized for carbon storage, has been proposed as a promising method to manage pressure within the reservoir and reduce risk. However, the burdens associated with the management of the brine extracted from the formation should not exceed the benefits of the extraction itself. A quantitative assessment of a range of potential extracted water management practices has been performed in order to provide valuable data and analysis to help decision makers answer this question. A range of possible management practices were evaluated, including reuse with and without treatment for total dissolved solids, a number of thermal and membrane treatment technologies, and brine disposal.

Each management strategy was evaluated for energy consumption, greenhouse gas emissions, water savings, and cost on a per-volume-of-water-managed basis. The results show that no single management strategy will be ideal in all cases. However, reuse without treatment when feasible, reverse osmosis treatment when the brine chemistry allows, and underground injection when treatment or reuse are not feasible, were the most promising management practices. In general, it appears that in many cases water management can be achieved while emitting less than 1% of the carbon that was originally stored in the formation over the life cycle of the injection and water extraction process. In addition, the costs of water management were estimated to fall within the range of $1 to $3 per ton of carbon dioxide stored, assuming a conservative 1:1 volume ratio of brine injection to water extraction. Overall, transportation mode and distance were found to have a very significant impact on energy consumption, emissions, and cost of water management, and thus should be key considerations in the selection of appropriate extracted water management practices.

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1 INTRODUCTION

1.1 BACKGROUND

The geological sequestration of carbon dioxide (CO_2) has been often cited as an important tool to combat climate change. For sequestration to have a reasonably positive impact on atmospheric carbon levels, the anticipated volume of CO_2 that would need to be injected is very large (many millions of tons per year). Stakeholders have expressed concern about elevated formation pressure following the extended injection of CO_2. If not properly managed and monitored, the increased formation pressure could stimulate new fractures or enlarge existing natural cracks or faults; thus the CO_2 or the *brine* pushed ahead of the plume could escape the formation and migrate vertically.

One possible tool for managing formation pressure is to extract *saline water* already residing in the formation where CO_2 is being stored. The concept is that by removing brine from the receiving formations (referred to as *extracted water* to distinguish it from oil- and gas-produced water), the pressure gradients caused by injection could be reduced, and additional pore space could be freed up to sequester CO_2. This process of extracting water to control pressure within the formation has been referred to as *active reservoir management*. A number of recent studies have begun to quantify the benefits of active reservoir management. The advantages include increased storage capacity, higher *injectivity*, improved reservoir control, lower CO_2 leakage risk, and reduced *area of review* (Kobos et al. 2011; Buscheck et al. 2012). These benefits, however, must be balanced against the costs of managing the extracted brine. This report seeks to improve understanding of these trade-offs.

1.2 PURPOSE OF THE STUDY

This effort supports the U.S. Department of Energy's (DOE's) National Energy Technology Laboratory (NETL) in evaluating management of extracted water. It builds upon the qualitative assessment of extracted water management options previously performed by Argonne National Laboratory (Argonne) and described in *Management of Extracted Water from Carbon Sequestration Projects* (Harto and Veil 2011). This report provides quantitative analysis of the environmental costs and benefits of many of the management options described in that report. The removal of water from formations during carbon capture and storage (CCS) is optional. Thus a key goal of this effort is to provide quantitative analysis that can be used to help determine the conditions under which the removal and management of water may result in positive environmental benefits. To accomplish this goal, a range of extracted water management scenarios were evaluated using a *hybrid life cycle assessment* (LCA) approach to compare their total energy consumption, greenhouse gas (GHG) emissions, and net water savings. Where sufficient data were available, rough approximations of costs for these scenarios are also provided. This study, however, only addresses the water management side of the equation. Thus the results presented here should be used, along with studies of the potential benefits of water extraction, to determine when water extraction is preferable.

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2 METHODOLOGY

2.1 LIFE CYCLE ASSESSMENT APPROACH

A hybrid LCA approach was used to estimate the environmental performance of the evaluated extracted management strategies. Hybrid LCA combines a traditional process-based LCA approach with an *economic input-output LCA* (EIO-LCA) approach to expand the system boundaries and include impacts for process where detailed information may not be available, such as the manufacturing of capital equipment (Williams 2004).

A *process LCA* requires detailed information about all the direct energy and material inputs into each life cycle stage. It can be more accurate for a specific system when the required data can be obtained. However, in many cases, all the data required are not available, and thus this approach has a tendency to leave out potentially important components of the life cycle for which data are not available.

The EIO-LCA methodology only requires information about the cost of different life cycle components and some information about the sectors of the economy to which these costs can be attributed. The model considers economic impacts for entire sectors of the economy, including the impacts resulting from input materials from other economic sectors. The results are normalized per dollar of economic output from the individual sectors to produce impact factors such as energy consumption or GHG emissions, normalized in terms of impact per dollar of output. This approach produces a much more complete accounting of all impacts from the full life cycle of a process. The limitation is that it is not good at distinguishing impacts from similar products from the same economic sector, as they are all assumed to have the same impacts per dollar spent. The specific EIO-LCA model used in this study is the US 2002 producer price model implemented at EIOLCA.net and developed by the Carnegie Mellon University Green Design Institute (CMUGDI 2011). It includes impact factors for 428 specific economic sectors. Since this model is from 2002, the impact factors were adjusted using the producer price index (PPI) to 2010 dollars (Economagic.com 2011).

For this analysis, the process-based approach was used to calculate direct impacts from processes where more information was available, such as transporting water or operating treatment systems. The EIO-LCA methodology was used to include life cycle environmental impacts of manufacturing capital equipment and some chemical inputs. This was done to take the greatest advantage of both the specificity of the process approach and the completeness of the EIO-LCA method. Using this approach to energy consumption, GHG emissions and water savings were calculated on a life cycle basis. All impacts for each stage of the life cycle were calculated using the same LCA methodology (process or EIO-LCA), with the exception of net water savings, which were only calculated based upon the direct water savings determined by the performance of the specific water management practice.

The analysis considered a range of water management strategies: reuse without treatment, treatment with a range of thermal treatment systems, treatment with *reverse osmosis* (RO), and disposal in injection wells. Each water management strategy was evaluated independently, assuming that all extracted water from a project would be managed in the same way. Additional management options, including disposal through evaporation and some novel, produced water treatment systems, were also evaluated and discussed; however, sufficient data were not obtained to quantify the full life cycle impacts. Both water trucks and pipelines were considered as water transportation options. The transportation distance to disposal, treatment, or reuse location was also evaluated to better understand its impact on the full lifecycle.

2.2 SYSTEM BOUNDARIES

The system boundaries of a LCA define what is included and what is excluded from the analysis. The environmental impacts of managing brine were considered from the point that it is extracted at the wellhead, until it is either delivered to a location where it will be reused, treated to the point that it can be reused or discharged, or properly disposed of in a disposal well, including any transportation required up until this point. The operational costs and benefits to a carbon sequestration project realized as a result of extracting water from the formation were deliberately not included in order to focus exclusively on the relative performance of different water management practices. It is expected that these operational impacts will be independent of the water management practices used. A full analysis to determine whether water extraction should or should not occur, must include detailed modeling of the reservoir, with and without water extraction, to evaluate all the benefits of extraction relative to the cost of managing the extracted water.

Each water management scenario includes the transportation burdens to the point of reuse, treatment, or disposal; the operational and capital costs of the treatment or disposal operation; and in the case of treatment systems, the cost of transport and disposal of the concentrate stream (all treatment systems produce both a clean water stream and a concentrated wastewater stream that contains all the salts and minerals in the original inlet stream). For all transportation legs, both the operational cost and capital costs of building the pipeline or water trucks are included. For reuse, the specific purpose for which the water is reused is not included within the system boundary; different options for reuse, however, are discussed in Section 3.2.

For evaluation of water savings, only the direct water savings were included. It was assumed that any extracted water that was either reused without treatment or cleaned by a treatment system and then reused would offset an existing water use resulting in a one-to-one water savings. This approach ignores any water consumption that might take place throughout the life cycle of the management process. While the direct water consumption from the practices evaluated is likely to be small, there is some indirect water consumption associated with the energy consumption for transportation and treatment that was excluded from this analysis.

2.3 FUNCTIONAL UNIT

A key step in any LCA is to define the functional units that are the units of analysis. The functional unit for this analysis is one barrel of water extracted and managed (reused, treated, or disposed of). The barrel (bbl) is the most common unit used in the oil and gas industry when discussing produced water; it is equal to 42 gal or 0.159 m^3. Also, while most desalination processes for water supply often use freshwater output as their functional unit, total extracted water managed (input into the treatment system) was determined to be a more relevant measure for this study because disposal of the waste stream (input) is the primary objective, while production of freshwater (output) is secondary.

Energy consumption was evaluated in units of British thermal units (BTUs) of primary energy consumption. This unit also takes into account the energy consumption required to extract and deliver the energy. For example, for a BTU of electricity consumed in a process, the fuel consumed at the power plant that supplied the electricity is counted to account for the full life cycle primary energy consumption. GHGs are accounted for in a similar manner, except that the unit used is grams of CO_2 equivalent. Water savings are measured as a fraction of the barrel extracted that is put to a functional use. For example, a direct reuse of water without any loss or treatment would result in a value of one barrel of water saved per barrel of water extracted. In the case of direct disposal, the value would be zero barrels of water saved per barrel of water extracted. Other units (e.g., kWh/bbl) are used throughout this report for input or intermediate values; however, all values were converted to the units discussed above for calculation of the full life cycle impact.

2.4 DATA SOURCES

LCA studies are only as good as the data they rely on. This study pulled from multiple types of sources, including, in order of preference, direct data from the water treatment and management industry, peer-reviewed literature, and grey literature. In addition, the study relied on impact factors for common materials pulled from the Argonne Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) LCA model that quantify the amount of GHGs emitted or primary energy consumed per unit of material or energy consumed (GREET 2012).

The ideal situation is to get data directly from companies manufacturing or operating water treatment or disposal facilities. Such data, however, are difficult to obtain and were only found for a few treatment systems. The advantage of these data is that they are direct, real world, and operational, and are likely to be the most accurate. However, this type of data is not available for most systems and scenarios, and some of the data must be treated as proprietary, which makes it harder for studies using the data to be completely open and transparent. For this reason, no proprietary data were used in this study.

Where industry data are not available, peer-reviewed literature is a good alternative. Peer-reviewed papers have a high degree of credibility and are typically fairly rigorous studies. However, in many cases the data provided are theoretical in nature and may not include all impacts and inefficiencies present in real-world applications.

Finally, where higher quality sources were not available, the grey literature was used. This includes conference papers, white papers, industry literature, industry handbooks, and state agency data that can provide useful data points. This type of data has lower credibility than peer-reviewed literature, but in some cases it may be the only option available.

One important point of discussion on data is that most data on water treatment is available for one of two primary applications—*seawater* desalination or produced water treatment. Neither of these applications is a perfect equivalent of CCS extracted water management. Table 1 compares produced water treatment, seawater desalination, and extracted water treatment across a number of key factors. Overall, extracted water management has more in common with produced water treatment than seawater desalination. This is important not only for selecting the most appropriate data to utilize, but also because produced water treatment is often significantly more expensive than seawater desalination, as discussed in Section 3.3. However, there are a few important factors that may reduce both the financial and environmental costs of extracted water management relative to produced water treatment.

The relatively steady and long-term source of input water may allow treatment facilities to be designed along with the project and sited near the extraction wells to reduce transportation costs. Waste disposal may also be factored into design and siting decisions to further reduce costs and impact. However, the composition of many geological brines, including produced water and geothermal fluids, which can have similar compositions to extracted water, have proven to be challenging to work with as they can have high temperatures, high concentrations of scale or precipitate-forming compounds, and contain dissolved naturally occurring radioactive materials (NORM) that can cause operational or waste disposal problems (Clark et al. 2011; Harto and Veil 2011). The specific composition of the extracted fluid can significantly increase the pretreatment steps needed and the costs of operating treatment facilities relative to seawater desalination facilities. All of the factors explored in Table 1 were considered when selecting appropriate data to include in the final LCA calculations for CCS extracted water applications.

TABLE 1 Comparison of Desalination, Produced Water, and CCS Extracted Water

Factor	Seawater Desalination	Produced Water Management	CCS Extracted Water Management
Primary objective	Clean water delivery	Waste elimination	Waste elimination
Water source	Ocean	Multiple wells, possibly multiple fields	Multiple wells from a single or multiple CCS projects
Input water quantity	As demanded	Highly variable	Depends on operational conditions, but likely low variability
Input water quality	Low variability	High variability	Unknown, possibly moderate to high variability
Operational considerations	Near ambient temperature, low concentration of scale, or precipitate-forming ions	Variable temperature, organic contaminants, scale-forming compounds, divalent ions, possible NORM	Variable temperature, scale-forming compounds, divalent ions, possible NORM
Transportation	Located at source, minimal transportation	Typically located in a producing area drawing from multiple wells, transport costs very important	Depends if dedicated to specific project or draws from multiple projects
Concentrate disposal	Minimal concern, returned to source	Disposal in evaporation or injection well, major cost consideration	Disposal in evaporation or injection well, major cost consideration

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3 EXTRACTED WATER MANAGEMENT OPTIONS

3.1 WATER MANAGEMENT HIERARCHY

When evaluating options for management of extracted water, a four-tier hierarchy can be helpful in the decision-making process. The first tier should be minimization of the amount of water that is extracted. If a project is going to use water extraction for reservoir control, modeling should be used to optimize reservoir operations in order to achieve the most benefits from extraction while minimizing the volume of water extracted. The second tier is beneficial reuse that requires minimal treatment (no removal of dissolved solids). The end use location should ideally be located nearby, but this is unlikely to be a common case. The third tier is reuse with significant treatment (removal of dissolved solids). When comparing reuse options it is possible that there could be trade-offs between nearby reuse applications that require expensive treatment and more distant reuse applications that require less treatment but more transportation. The fourth tier is disposal. This is the least desirable option from a water conservation standpoint; however, in many cases, it may be the most economical or even the only option for managing extracted water.

3.2 REUSE WITH MINIMAL TREATMENT

Water is an important resource, and there are a number of potential reuse options for extracted water, many of which were discussed in the previous report on managing extracted water from carbon sequestration projects (Harto and Veil 2011). Of these options, some may allow for reuse without treatment to remove dissolved solids, which can be an expensive and energy-intensive process. These options include, but are not limited to, oil field uses such as enhanced oil recovery or use as drilling or hydraulic fracturing fluid, subsidence control or saltwater intrusion control, and makeup water for geothermal systems.

The specific end use application, however, is not specified or included in the LCA calculations because it is outside the established system boundaries for this study. Reuse is also considered without any treatment, and only transportation to the ultimate reuse location is included. In reality, nearly all reuse applications will require at least some minimal treatment such as the filtration or settling of suspended solids, and chemical treatment to manage the growth of microorganisms or to control precipitation and scaling. These processes, however, are often significantly less expensive and energy intensive than treatment to remove dissolved solids. Many of these processes may also already be in place at the location of reuse to treat water from other raw surface or groundwater sources. These burdens are thus considered outside of the system boundaries for extracted water management, and, instead, are associated with the end use process.

Thus the most important parameter for the costs and environmental impacts of reuse is the distance to the reuse location. Transportation of water over long distances can be expensive and logistically challenging, so transportation of extracted water to distant use locations may not be feasible. The specific parameters used to quantify the environmental burdens associated with extracted water transportation for all management options, including reuse, are specified in Section 3.5.

When considering reuse, it is also important to match the quantity, quality, and timing of the extracted water supply with the demands of the end use. This can be a challenging process, and if any of the above does not sufficiently match, reuse without treatment may not be a viable option. From an operational risk point of view, if a viable reuse application is found, it will still be important to have a backup plan in place in the event that the end user's needs change or they become unable to accept the extracted water for some reason.

3.3 REUSE WITH TREATMENT FOR TOTAL DISSOLVED SOLIDS

While beneficial reuse without treatment is the ideal, many reuse options will require significant treatment of the extracted water. Current regulations state that carbon sequestration can only occur in formations where the brine has a *total dissolved solids* (TDS) level of 10,000 ppm or greater, unless a special waiver is obtained. At these levels, the removal of TDS will be required for discharge to a surface water body or many common reuse applications such as agriculture, cooling water, or public water supplies.

Two primary categories of treatment processes remove TDS—thermal treatment and membrane treatment. Both systems generate a clean water stream that is relatively free of salts and other dissolved solids and a concentrated wastewater stream that contains the majority of the dissolved solids that have been separated out. The fraction of the water that enters the treatment system that becomes clean water versus the fraction that goes to the waste stream can vary significantly depending upon the technology, the inlet TDS concentration, and the operational parameters of the system. The ratio of clean water produced to feedwater input (which in this case would be the total volume of extracted water) into the system is commonly referred to as the conversion ratio or *recovery ratio*. The total input into the plant is referred to as the *treatment rate*. In general, membrane systems are more energy efficient than thermal systems, especially at low TDS concentrations; however, their costs and energy requirements increase as the concentration increases. This factor, combined with physical constraints based upon membrane strength, typically limit these systems to waters with initial TDS concentrations not much higher than seawater (~35,000 ppm). Thermal systems, on the other hand, can treat water with significantly higher TDS concentrations, but they also usually consume more energy. The inlet water quality has less of an impact on the operational costs and energy consumption for thermal systems, although it still has an impact on important operational considerations such as scale formation and recovery ratio.

3.3.1 Thermal Treatment

Thermal treatment methods all use heat and/or mechanical energy to evaporate and then re-condense water. The differences in system designs result from differences in how these thermodynamic processes are carried out and how effectively they recapture the energy released from condensing the vapor generated from the evaporation step. Thermal treatment systems can typically treat water over a wide range of starting water quality. The primary limiting factor for thermal systems is scale formation, which increases with higher inlet TDS concentrations or by operating at high recovery ratios.

All thermal treatment processes are vulnerable to scaling, which occurs when certain species precipitate out of solution and adhere to the surface of process equipment. The most common scale-forming compounds in seawater desalination systems are calcium carbonate, calcium sulfate, and magnesium hydroxide. Scaling has the greatest impact on the operations of a treatment system when it occurs in heat transfer equipment because it results in a reduction in the rate of heat transfer and can reduce the efficiency of the process. Some scale-forming compounds can be controlled through pretreatment; others must be controlled by carefully adjusting the operational parameters to avoid conditions that would result in scale formation. This is typically done by either reducing the operating temperature or reducing the concentration of scale-forming compounds in the concentrate stream by lowering the recovery ratio (BOR 2003).

The inlet water temperature is also an important parameter for some thermal systems. Both Multi-Effect Distillation (MED) and Multi-Stage Flash (MSF) systems actually benefit from low inlet water temperatures as the processes are driven by the temperature difference between the cool inlet water and the heat source fueling process. The larger the temperature difference, the greater the number of stages or effects that can be included in the system, and the higher the system efficiency (BOR 2003). This will be an important consideration when selecting an appropriate brine management strategy for specific carbon storage formations. If brine temperatures exceed viable operating temperatures for some treatment systems, brines will either need to be cooled prior to treatment or different system designs or management strategies will need to be selected.

3.3.1.1 Multi-Stage Flash

In an MSF system, water is superheated under pressure to prevent vaporization. It is then passed through a series of flash tanks at lower and lower pressures to generate steam. In each flash chamber there is a heat exchanger that simultaneously condenses the produced vapor and preheats the feedwater. MSF systems are often paired with thermoelectric power plants to take advantage of efficiencies from combining the two processes.

MSF systems are the most common thermal treatment system used for desalinating seawater. They are especially common in the Middle East where fossil fuels are relatively inexpensive and abundant (Miller 2003). Many of these plants are quite large, with water production rates on the order of 300,000 to 500,000 bbl of output per day (50,000–75,000 m^3/day) per unit (Al-Sahali and Ettouney 2007). They also tend to be reliable, with low maintenance requirements. However, these systems can only be operated at full capacity, limiting operational flexibility and requiring a stable and reliable feedwater source (Darwish and Al-Najem 2000).

Data were obtained from the literature for MSF systems from three different sources. Table 2 is a summary of the relevant system parameters used to calculate the LCA results. The energy costs are presented on a per-barrel-treated basis. They have been converted from a per-barrel-of-clean-water-produced basis, which is commonly used in the desalination literature, by utilizing the reported recovery ratios.

The capital costs were estimated on a per-barrel-treated basis assuming a 90% capacity factor and a 20-year plant lifetime.

3.3.1.2 Multi-Effect Distillation

MED systems operate by utilizing an outside heat source to generate vapor in the first stage or effect. The heat from the vapor generated from the first stage is then used to generate more vapor in the next stage. As the heat is exhausted from the steam from the previous stage, it is condensed and becomes part of the clean water stream. Each subsequent stage is operated at a slightly lower pressure, thus water evaporates at a lower temperature than the previous stage. The vapor from the final stage is condensed in a separate condenser that typically relies on additional feedwater for cooling. The efficiency of the system increases with the number of effects in the sequence, as more and more clean water is produced from the same heat input; however, each additional effect also increases the capital costs.

In general, MED systems tend to have more issues with scaling than MSF systems, and therefore they have been less popular for large-scale desalination projects. They also tend to be smaller than MSF systems; typical units produce between 100,000 and 150,000 bbl of output per day (15,000 and 25,000 m^3/day) (Al-Sahali and Ettouney 2007). They remain of interest, however, because they are generally more energy efficient than MSF systems, and newer systems have been designed to limit scale formation (Miller 2003).

Data were obtained from the literature for MED systems from two different sources. Table 3 is a summary of the relevant system parameters used to calculate the LCA results. All

TABLE 2 Summary of Multi-Stage Flash System Parameters

Data Source	Raluy et al. 2004	Al-Sahali and Ettouney 2007	Darwish and Al-Najem 2000
Recovery ratio	0.43	0.43	0.38
Inlet TDS (ppm)	Seawater	40,000	Seawater
Treatment rate (bbl/day)	733,000	996,000	779,000
Electricity demand (kWh/bbl)	0.26	0.97	0.25
Thermal energy (BTU/bbl)	21,500	0	17,700
Chemical cost ($/bbl)	0.012[a]	0.010	0.012[a]
Capital cost ($/bbl)	0.025[a]	0.009	0.023[a]

[a] Estimated using cost curves from BOR (2003).

TABLE 3 Summary of Multi-Effect Distillation System Parameters

Data Source	Raluy et al. 2004	Al-Sahali and Ettouney 2007
Recovery ratio	0.35	0.34
Inlet TDS (ppm)	Seawater	42,000
Treatment rate (bbl/day)	360,000	370,600
Electricity demand (kWh/bbl)	0.11	0.67
Thermal energy (BTU/bbl)	13,850	0
Chemical cost ($/bbl)	0.009[a]	0.010
Capital cost ($/bbl)	0.019[a]	0.009

[a] Estimated using cost curves from BOR (2003).

parameters are presented on a per-barrel-treated basis. Capital costs were estimated assuming a 90% capacity factor and a 20-year plant lifetime.

3.3.1.3 Mechanical Vapor Compression

Mechanical vapor compression (MCV) systems operate in a single vessel. Feedwater is sprayed over a heat exchanger, which results in the generation of water vapor. The vapor is then fed to a compressor. The process of compression increases the temperature of the vapor, which is then passed through the heat exchanger where it acts as the heat source for the evaporation process. Once the heat from the vapor has been exhausted in the heat exchanger, it condenses and exits the system as clean water (BOR 2003).

MVC systems tend to be smaller than both MED and MSF systems because they have a fixed power requirement regardless of the size of the system, thus there are minimal economies of scale for these systems. In general, MVC units are limited to around 30,000 bbl of output per day (5,000 m^3/day), with most units around 3,000 bbl per day (500 m^3/day) (Al-Sahali and Ettouney 2007).

Fewer data were available in the literature for MVC systems than for either MED or MSF. Only one source contained enough information to include in the LCA results. One additional MVC system is discussed in Section 3.3.1.4 on produced water treatment systems. Table 4 is a summary of the relevant system parameters. All parameters are presented on a per-barrel-treated basis. Capital costs were estimated assuming a 90% capacity factor and a 20-year plant lifetime.

TABLE 4 Summary of Mechanical Vapor Compression System Parameters

Data Source	Al-Sahali and Ettouney 2007
Recovery ratio	0.3
Inlet TDS (ppm)	42,000
Treatment rate (bbl/day)	10,500
Electricity demand (kWh/bbl)	0.38
Chemical cost ($/bbl)	0.01
Capital cost ($/bbl)	0.007

3.3.1.4 Novel Produced Water Treatment Systems

Three different recently developed thermal produced water treatment systems were also evaluated. While the vast majority of produced water is disposed of through reinjection, or used for oilfield uses, such as enhanced oil recovery, there has been an increase in interest in treatment and recycling of produced water in recent years. These systems are significantly smaller than the ocean desalination systems discussed above because of the smaller volumes of water generated from individual oil and gas wells. They also include more pretreatment steps due to the presence of oil and other organics and higher concentrations of suspended solids. Because of data limitations, only the operational stage could be evaluated for these systems, and thus they are not included in most of the summary LCA results. It is also important to note that most of the data presented for these systems are from early field trials, and the systems have not been fully optimized. However, it is still important to consider these systems because, unlike seawater desalination systems, they are designed to treat inland brines with compositions that more closely approximate extracted water.

The first system evaluated was the NOMAD system manufactured by AquaPure Ventures, Inc., of Calgary, Alberta, Canada. It combines an MVC system with pretreatment steps to remove organics, suspended solids, and control scale. The entire system is modular and designed to be semi-portable (moved and set up in a period of weeks) so that it can move along with drilling operations as needed (Hayes and Severin 2012).

The second system evaluated was HED Model 600 E-M, designed by HED Environmental Systems, Inc., of Houston, Texas. This system is highly portable and operates utilizing high vacuum and low temperatures in a single flash tank. The system can be fueled by natural gas of varying quality, preferably low quality, or unsalable or flared gas available in an oil field. However, this fuel source would not be expected to be available for desalination of extracted water. In addition to a clean water and concentrated brine stream, this system also loses water to atmospheric evaporation, which both reduces the volume of waste brine that must be disposed of, but also reduces the amount of useful clean water produced (Frick 2011).

The final system evaluated was the Altela Rain® 600 system designed by Altela, Inc., with offices in Albuquerque, New Mexico, and Denver, Colorado. The system operates by utilizing a humidification/dehumidification process. This process operates by taking advantage of the difference in the quantity of water that can be held by a given mass of air at a given temperature. Low-temperature air enters the bottom of the system and is heated through a heat exchanger in contact with the inlet water and increases in humidity. At the top of the tower, the humid air is mixed with makeup steam which acts as the heat source for the process. It is then passed to the other side of the heat exchanger where it is cooled as it gives up its heat to the incoming air, thereby condensing out a large portion of the water collected in the initial step. Like the HED system, this system also loses water to the atmosphere because the air that exits the system is saturated with water that cannot be fully recovered. The primary advantages of this system are that it is constructed of inexpensive plastic, which both minimizes capital costs and limits the impact of scaling because most scale-forming compounds will not adhere to the surface (Bruff et al. 2011).

While complete LCA results were not completed for these systems, the available and relevant system parameters for the systems are presented in Table 5. In general, the treatment rates are significantly lower and the energy requirements are significantly higher than the seawater desalination systems discussed above. It is unclear exactly why the energy requirements are so much higher. It may be a combination of the lack of economies of scale, pretreatment and balance of system loads, and lack of system optimization, because all of these systems are relatively new to the market.

3.3.2 Membrane Treatment

Membrane processes use selectively permeable membranes and pressure to treat water. The most common membrane treatment process is RO, and it is the focus of this analysis. Microfiltration, ultrafiltration, and nanofiltration are also common membrane treatment processes, but none of them are able to remove most salinity. However, they may be used as pretreatment steps prior to treatment with RO or thermal treatment to limit the potential for scale and fouling.

TABLE 5 Summary of Produced Water Treatment System Parameters

Data Source	Frick 2011	Hayes and Severin 2012	Bruff et al. 2011
System name	HED 600 E-M	NOMAD	AltelaRain 600
Recovery ratio	0.44	0.72	0.63
Atmospheric loss fraction	0.21	0	0.12
Inlet TDS (ppm)	107,000	50,000	25,000–37,000
Treatment rate (bbl/day)	570	6,300	2,400
Electricity demand (kWh/bbl)	4.63	0.00	2.48
Thermal energy (BTU/bbl)	147,000	71,700	200,000

RO membranes are designed to allow water molecules to flow through, but not salts and other dissolved minerals. In order to induce flow through the membrane, a pressure must be exerted on the fluid that exceeds the osmotic pressure of the fluid. The osmotic pressure of a solution goes up as the concentration of dissolved ions goes up. The flow rate through the membrane is also proportional to the magnitude of the pressure exerted above the osmatic pressure. The main limiting factor in RO systems is the pressure that the RO membrane can withstand. While new membranes are being developed all the time, typical RO membranes can only withstand around 1,500 to 1,800 psi of pressure (Bourcier et al. 2011). This usually limits the viability of RO to treatment to waters with TDS levels of 50,000 ppm or less. Also, as the concentration of TDS increases, the recovery ratio goes down. This reduces the amount of clean water produced and increases the amount of concentrate that must be disposed of.

RO systems also have a range of recovery ratios for a given inlet TDS concentration that result in a minimum energy requirement. For common seawater systems, this optimum is in the range of 35 to 45% recovery. However, for inland treatment systems, where the costs of disposing of concentrated brine must be considered, it may be beneficial to operate the system in a less energy-efficient manner in order to maximize recovery and reduce disposal costs. This trade-off has not been fully explored in this study, but it should be considered if designing an RO system for the treatment of extracted water. The efficiency of many modern RO systems has been improved through the inclusion of energy-recovery systems to capture energy from the pressurized concentrate stream and use it to help pressurize the feedwater.

The biggest challenge with RO systems is that the membranes are sensitive to fouling, which can significantly reduce the flow through the membrane (flux), and thus RO systems can require significant pretreatment. Membranes can be sensitive to temperature, pH, oxidizers, organics, algae, bacteria, particulates, and precipitates (Miller 2003). The presence of calcium ions has also been identified as a key driver of RO membrane fouling (Lee et al. 2006). Recent experiments with the use of RO membranes for the treatment of produced water have largely been unsuccessful because of rapid flux declines due to fouling of the membranes in anywhere from a few hours to a few months. A combination of ultrafiltration and nanofiltration as pretreatment have shown the ability to extend the life of RO membranes to more than 6 months, but this is still significantly shorter than the design life and will increase the cost of operating an RO facility (Muraleendaaran et al. 2009). Most RO membranes are also limited to temperatures below 35 to 45°C, which may necessitate cooling of extracted water before treatment (BOR 2003).

It is unclear at this point to what degree treatment of extracted water with RO will run into these same problems. While extracted water is unlikely to have significant concentrations of organics, brine from certain formations may have high concentrations of suspended solids or minerals that can precipitate and foul the membranes, such as calcium and silica. Pilot tests of RO systems with brines from specific formations expected to be used for carbon sequestration are needed to verify the effectiveness of RO in treating these brines. Data were obtained for four seawater RO systems and one brackish groundwater RO system. Table 6 shows the relevant parameters for these systems. As expected, the *brackish water* system operates at a higher recovery rate and uses less energy than the seawater systems, reinforcing the advantages of using RO for treating low TDS waters. For similar system designs, the addition of energy-recovery

TABLE 6 Summary of Reverse Osmosis Treatment System Parameters

Data Source	Busch and Mickols 2004	Busch and Mickols 2004	Raluy et al. 2004	Darwish and Al-Najem 2000	BOR 2003
Energy recovery	No	Yes	No	Yes	No
Recovery ratio	0.45	0.45	0.45	0.35	0.75
Inlet TDS (ppm)	38,000	38,000	Seawater	43,000	10,000
Treatment rate (bbl/day)	133,000	133,000	ND[a]	102,600	190,000
Electricity demand (kWh/bbl)	0.29	0.1S6	0.29	0.36	0.13
Chemical costb ($/bbl)	0.005	0.005	ND	0.006	0.006
Capital costb ($/bbl)	0.019	0.019	ND	0.016	0.010

[a] ND = no data.

[b] Estimated using cost curves from BOR (2003).

equipment significantly reduced energy consumption; however, the highest energy-consuming system also included energy recovery. While in general energy recovery will reduce energy consumption, for the purposes of the LCA analysis, all four seawater systems were averaged rather than attempting to separate out the impact of energy recovery due to the limited number of data points available.

3.4 DISPOSAL

The least favorable management option from a water resource standpoint is disposal, because the extracted water provides no benefits beyond those associated with pressure management within the reservoir. However, it may be the cheapest or even only option available for water management in some locations due to either the composition of the extracted brine or cost of treatment. The two processes that are likely to be available for disposing of extracted water or concentrate streams from treatment systems are injection into a U.S. Environmental Protection Agency (EPA)-permitted underground injection control (UIC) disposal well or evaporation.

3.4.1 Underground Injection

Underground injection in disposal wells is the most commonly used method for disposing of produced water in oil and gas fields. All disposal wells are permitted by the EPA UIC Program. There are currently six classes of UIC wells, but it is unclear which class would apply to wells used for the disposal of extracted water. The most likely candidates would be Class I, which covers industrial and municipal waste disposal; Class II, which covers oil- and gas-related wells; or Class VI, which covers geological sequestration wells. The permitting and well construction requirements vary depending upon the class of wells (EPA 2012). For the purposes of the LCA analysis, extracted water disposal wells were assumed to be similar to Class II brine disposal wells.

Data on the depth, pressure, and injection rate were obtained for Class II disposal wells from the Powder River Basin in Montana and Wyoming, Texas, and Ohio (Ray and Engelhardt 1992; Railroad Commission of Texas 2012; Tomasik 2012). These parameters were used to calculate the energy required to pump water into each well using a standard method for pump calculations (Geankoplis 1993). Pumps were assumed to be electric with an efficiency of 75%. The cost of constructing the well was calculated using a cost curve for oil and gas wells as a function of depth (Tester et al. 2006). Table 7 summarizes the injection well parameters and calculations. The required injection energy and average well cost are presented as weighted averages based upon the injection rate of the wells and normalized per-life-time barrels of water injected, assuming continuous injection over a 20-year well lifetime. There are clearly some regional differences in the properties of the injection wells. For Ohio, the required injection energy is the lowest due to lower injection pressures, but also the costs for the wells are significantly higher because of lower injection rates. Wells in Texas and the Powder River Basin have slightly higher energy requirements but much lower costs. This variability is due to differences in the geology of the formations used for injection. The viability and performance of injection wells are strongly tied to the availability and suitability of the nearby formations for injection.

The variability among wells is further illustrated in Figures 1 through 4. Note that when reading the histograms, the values on the x-axis of the distributions represent the upper limit of the specific bin. Well depths seem to vary between about 2,000 and 8,000 ft; depth is likely limited at the shallow end by permit requirements and risks to groundwater, while deeper wells are likely limited by cost. The injection rate appears to be more variable; there is a large number of low-volume wells (mostly in Ohio), and the rest of the wells are more broadly distributed between 500 and 16,000 bbl/day. The injection energy can vary anywhere from under 0.1 kWh per barrel all the way up to 0.9 kWh per barrel, with the distribution more heavily weighted toward the lower end. The injection energy is a strong function of the injection pressure, which is directly related to the injection formation properties. With a given well in a given formation, an operator has some flexibility in choosing the pressure at which to inject; however, there is a trade-off between a higher injection pressure that will allow a higher flow rate and the associated

TABLE 7 Summary of Injection Well Parameters and Calculations

State	Montana/Wyoming	Ohio	Texas
Number of wells analyzed	8	66	63
Average depth (ft)	6,094	4,447	4,627
Average pressure (psig)	1,108	666	830
Average injection rate (bbl/day)	4,737	439	3,644
Weighted average injection energy (kWh/bbl)	0.51	0.35	0.43
Weighted average well cost ($/bbl)	0.028	0.192	0.025

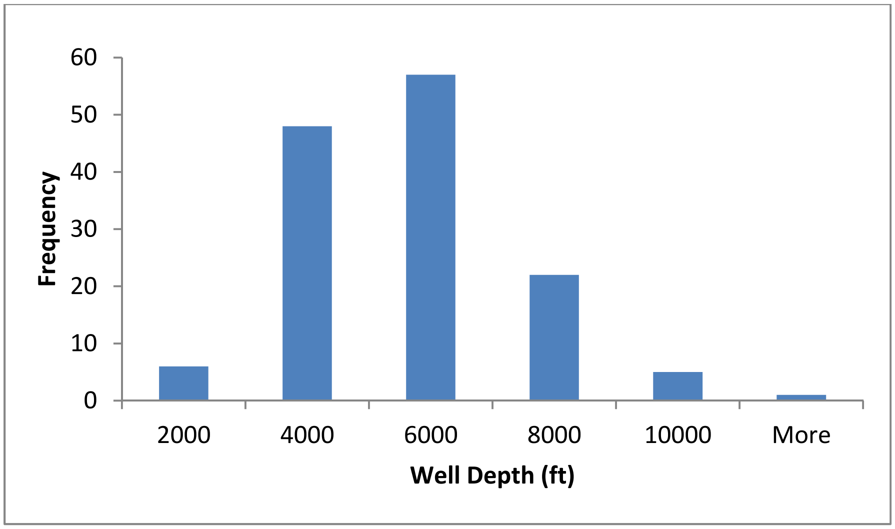

FIGURE 1 Class II Injection Well Depth Distribution

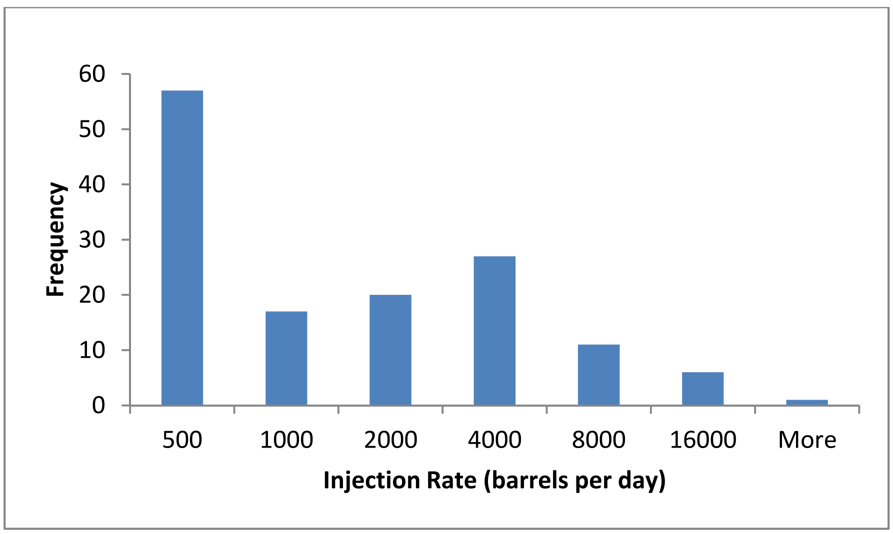

FIGURE 2 Class II Injection Rate Distribution

greater energy consumption per barrel injected. The operator is also usually limited by a maximum injection pressure that has been approved in the UIC permit. The well cost distribution also shows a large amount of variability. While the weighted average well cost for both the Powder River Basin and Texas shown in Table 7 is below \$0.03 per barrel, this weighted average appears to be driven down by a limited number of high-volume, low-cost wells. This indicates a financial risk associated with drilling injection wells. While high-performance wells can be very efficient and inexpensive, many wells that are drilled may have lower performance and therefore significantly higher lifetime costs. An additional potential risk associated with underground injection is the growing concern about the potential for induced seismicity from injection wells (Kim 2013; Ellsworth 2013). In the case of disposal of extracted water, any increase in risk of

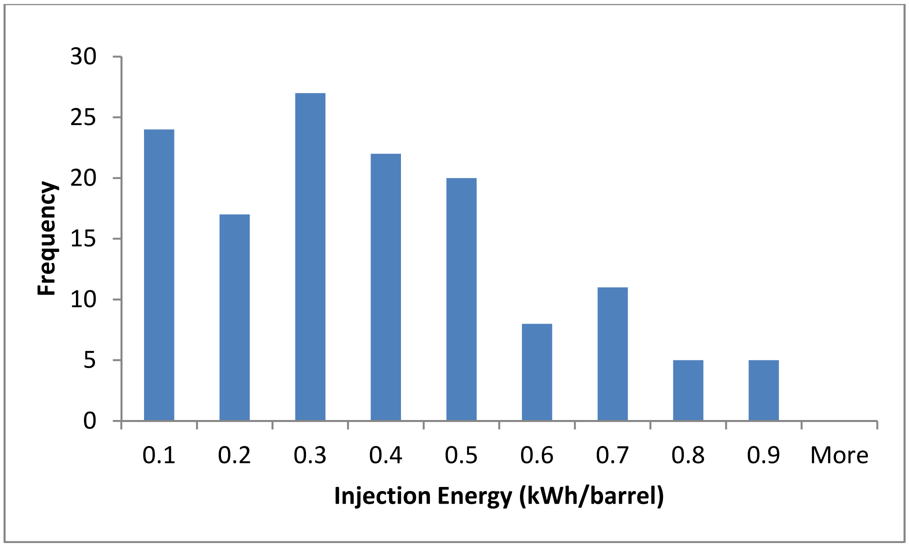

FIGURE 3 Class II Injection Energy Distribution

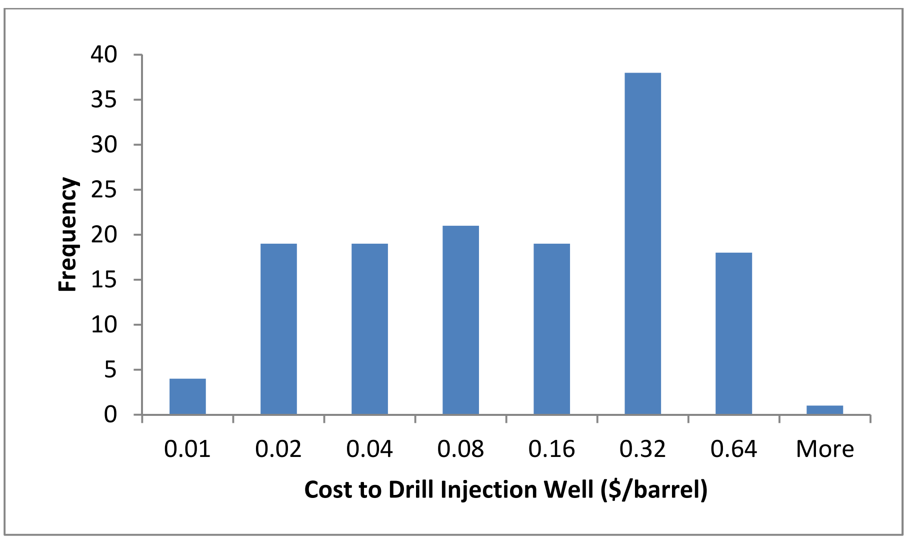

FIGURE 4 Class II Well Cost Distribution

seismicity associated with disposal would need to be weighed against any reduction in seismicity associated with CO_2 injection as a result of pressure management.

3.4.2 Evaporation

Evaporation is the main alternative to injection for produced water disposal. There are two primary methods of evaporation. The first involves solar-driven evaporation from large high-density polyethylene (HDPE)-lined ponds. The other involves engineered systems that use a thermal energy source of some kind to evaporate the water. While both of these disposal methods are discussed, insufficient data were obtained to generate complete LCA results for either of them.

Lined ponds are currently used as a means of disposing of produced water in arid environments. Their main advantages are that they are relatively inexpensive in areas with low precipitation, high surface evaporation rates, and large quantities of low-value land. However, their usefulness is limited geographically. An industry study looked at four different sites for new evaporation facilities in Wyoming, New Mexico, and Utah and estimated the annual evaporation rates at each site to be between 45 and 50 in. per year (Nowak and Altman 2011). An effective evaporation rate of 45 in. per year works out to be approximately 80 bbl per acre per day. At that rate, a disposal facility sufficient to dispose of 100,000 bbl per day would require nearly 2 mi^2 of effective pond area. While the energy and GHG emissions from such a facility would be expected to be relatively low, especially after construction, the large area of land required is likely to cause other environmental problems and permitting challenges. For these reasons, it is unlikely that this technology would be a viable option except for small-scale operations in a limited number of geographical settings.

Thermal evaporation systems are also likely to be of limited applicability for disposing of extracted water. The main challenge for thermal evaporation systems is the large quantity of energy required to evaporate water. The heat of vaporization of pure water at standard conditions is 340,000 BTU/bbl, and none of that energy is recovered in a condensation step as it is in most thermal treatment systems. One company is marketing a thermal evaporation system for oil field wastes, but the average energy consumption from its system is 560,000 BTU/bbl, which significantly exceeds the energy demand for all of the thermal treatment systems evaluated (Stone and Christensen 2011). The selling point is that the system can use low-grade waste heat to operate, which is likely the only way such a system could operate economically. This may be a possibility at a small scale in some oil and gas fields that have excess or low-quality natural gas that would otherwise be flared, but it is unlikely to be a viable option for large-scale disposal of extracted water.

Both of these evaporation systems will also require a method of disposing of the solids that precipitate out as the water is evaporated. Depending on the inlet TDS concentration of the water, this could result in a very significant solid waste stream. Also, depending on the initial water composition, this solid waste stream could contain NORM, which would require careful handling and expensive disposal only at permitted facilities. While NORM may also be an issue for some other treatment systems because of precipitation and scaling, in those systems most of the NORM will likely remain in solution in the concentrate stream and can still be disposed of in a UIC disposal well without additional handling costs. While lined ponds may be a viable option for some smaller scale operations in ideal climates, and thermal evaporation systems may be viable in locations with abundant, low-value waste heat sources, the challenges associated with evaporation systems are likely to limit their broad applicability for managing extracted water from CCS projects.

3.5 TRANSPORTATION

The transportation and logistics of managing extracted water can be as important as the ultimate use, treatment, or disposition. The two most common means of transporting water are trucks and pipelines. Trucks are far more flexible in that they can be hired only when needed and can deliver water any distance and to any location accessible by existing roads. However, they tend to be expensive because they require a paid driver and are limited in the amount of water that they can transport per trip; thus a large number of trucks are required for transporting large volumes of water. These characteristics make trucks a common means of transporting produced water in oil and gas fields where smaller volumes of water are produced from many different wells throughout an area. Pipelines, however, are ideal for transporting steady volumes of water between two fixed locations. They require far less labor than trucks, but they also can be difficult to permit and build because of challenges with obtaining the right-of-way (ROW) to build the pipeline, depending upon who owns the land between the two locations. One way to minimize these challenges for CCS projects would be to build water pipelines in conjunction with CO_2 pipelines along the same ROW.

3.5.1 Water Trucks

Water trucks are limited by the U.S. Department of Transportation regulations on heavy trucks, which limits their total weight to 80,000 lb. Because of this, most water trucks are sized to carry between 80 and 150 bbl of water. For the LCA calculations, trucks were assumed to have a payload of 120 bbl. Fuel consumption for truck trips was calculated round trip, with a fully loaded fuel efficiency of between 5 and 7 mpg while returning empty (Delorme et al. 2009; Davis et al. 2010). The cost of water trucks was obtained from a website for the sale of new and used construction and heavy equipment (rockanddirt.com 2012). The average price of new water trucks normalized for capacity was approximately \$1,900/bbl of capacity. Used trucks were slightly cheaper at \$1,600/bbl. Assuming a 400,000-mi lifetime for a new truck, the total cost of water delivery was calculated to be just under \$0.005/bbl-mi. This value represents the capital cost associated with transporting 1 barrel of water 1 mi, assuming the truck is always full. In the scenarios analyzed, the truck was always assumed to be returning empty, so the effective cost would be \$0.009/bbl-mi (discrepancy due to rounding) for the 50% of the time that it is transporting water. This value was used to calculate the energy and GHG emissions associated with manufacturing the truck.

3.5.2 Pipelines

Water pipelines were sized and modeled using a coal-bed methane-produced water management tool developed by the Colorado School of Mines (Colorado School of Mines 2012). Pipelines were modeled with flow rates from 10,000 to 1,000,000 bbl per day (pipelines were sized to handle flow up to 20% greater than the design flow) and distances of 10 and 100 mi. The pipeline capital costs were calculated using recent data on cost estimation for water systems in South Central Texas (SCTRWPG 2010). Costs were considered for both constructing the pipeline and for pumping stations.

Table 8 shows some key model parameters and outputs for the pipeline scenarios modeled. The results show very strong economies of scale in response to increasing pipeline flow rate. This is because at higher flow rates, larger diameter pipes can be used, which reduces friction losses per unit of flow, thereby reducing energy consumption and pump size. Also, while larger diameter pipes are more expensive, the cost of the pipe increases approximately linearly with diameter, while flow increases by the square of the diameter. The effect of economies of scale is far less substantial as a function of pipeline length. The energy difference for longer versus shorter pipelines is relatively minimal and likely only due to end effects having a more pronounced impact on a per-barrel-per-mile basis for shorter pipelines. End effects are losses associated with fluid inertia at both the beginning and end of a pipeline. From a capital cost perspective, no economies of scale are recognized for the pipe itself; however, longer pipelines generally require larger pumping stations, which allows for some economies of scale to be recognized for the cost of pumps.

Table 9 shows the sensitivity of the required pump energy to changes in elevation. Elevations of plus or minus 1,000 ft were considered. The relative effect of elevation changes is far more pronounced for larger flows and shorter distances. The reasons are that for lower flows, the mass of water being lifted is low and the frictional losses on a per-volume-basis are high; thus the impact of elevation changes is relatively small. Also, for longer distances, the impact of the elevation change is spread over a longer distance so that the impact per-barrel-per-mile is also small. That being said, the absolute impact of an elevation change appears to be consistent across flow rates. When normalized for distance, an elevation change of 1,000 ft results in an absolute change in energy consumption of approximately 0.2 kWh/bbl in the direction of the elevation change. This is logical given that the change in potential energy for lifting (or dropping) a unit of water is constant and independent of the amount of water you are lifting.

TABLE 8 Summary of Pipeline Model Parameters and Calculations

Flow Rate (bbl/day)	Distance (mi)	Pipe Diameter (in.)	Specific Pump Energy (kWh/bbl-mi)	Capital Cost ($/bbl-mi)
10,000	10	4	0.116	0.0149
10,000	100	4	0.110	0.0091
33,000	100	8	0.050	0.0046
100,000	10	12	0.031	0.0025
100,000	100	12	0.029	0.0017
1,000,000	10	36	0.008	0.0006
1,000,000	100	36	0.008	0.0005

TABLE 9 Pipeline Energy Sensitivity to Elevation Changes

Flow rate (bbl/day)	Distance (mi)	Pump Energy +1,000 ft (kWh/bbl-mi)	Pump Energy Flat (kWh/bbl-mi)	Pump Energy −1,000 ft (kWh/bbl-mi)
10,000	10	0.135	0.116	0.096
10,000	100	0.112	0.110	0.108
100,000	10	0.050	0.031	0.010
100,000	100	0.032	0.029	0.028
1,000,000	10	0.029	0.008	0.000
1,000,000	100	0.010	0.008	0.006

4 LCA RESULTS

4.1 LCA SCENARIOS

Complete LCA results were generated for nine different scenarios. These included five treatment scenarios and two scenarios each for underground injection and reuse without treatment. All scenarios assumed transport by pipeline, and transportation burdens were calculated based upon a 100,000 bbl/day flow rate. This flow rate was selected as it is approximately the flow rate expected from the 3.8 million tons per year CO_2 storage operation modeled by Buscheck et al. (2012), assuming a one-to-one volumetric displacement of brine This is also approximately the flow of CO_2 from a 500-MW coal-fired power plant with 90% capture. The transportation distance for most scenarios was a fixed 10 mi. While this may be an overestimation of transportation distance in some cases, some transportation is expected to be associated with all systems as water will need to be transported from multiple extraction wells to a single treatment, reuse, or disposal location. Two additional scenarios for injection and reuse were run with a transportation distance of 100 mi, assuming there were no suitable nearby injection zones or reuse opportunities. Disposal of all concentrate from treatment systems was assumed to be through injection in disposal wells 10 mi away. All scenarios were calculated using the average of parameter values from the literature for the relevant management strategy, as presented in Chapter 3. Table 10 summarizes the important assumptions for each scenario.

4.2 ENERGY CONSUMPTION

Figure 5 shows the energy consumption LCA results. Energy consumption is broken down into major categories, including operations, capital, transportation, and waste disposal. The general trend in the results is not surprising, with thermal treatment methods being the most energy intensive (with the exception of injection and reuse 100 mi away), followed by RO,

TABLE 10 Summary of LCA Scenarios

Scenario	Technology	Water Source	Transport Distance (mi)	Number of Data Points Averaged
MSF	Multi-Stage Flash	Seawater	10	3
MED	Multi-Effect Distillation	Seawater	10	2
MVC	Mechanical Vapor Compression	Seawater	10	1
Ocean RO	Reverse Osmosis	Seawater	10	4
Brackish RO	Reverse Osmosis	Brackish Groundwater	10	1
Injection 100 mi	Underground Injection	Any	100	137
Injection 10 mi	Underground Injection	Any	10	137
Reuse 100 mi	Reuse (no treatment)	Any	100	1
Reuse 10 mi	Reuse (no treatment)	Any	10	1

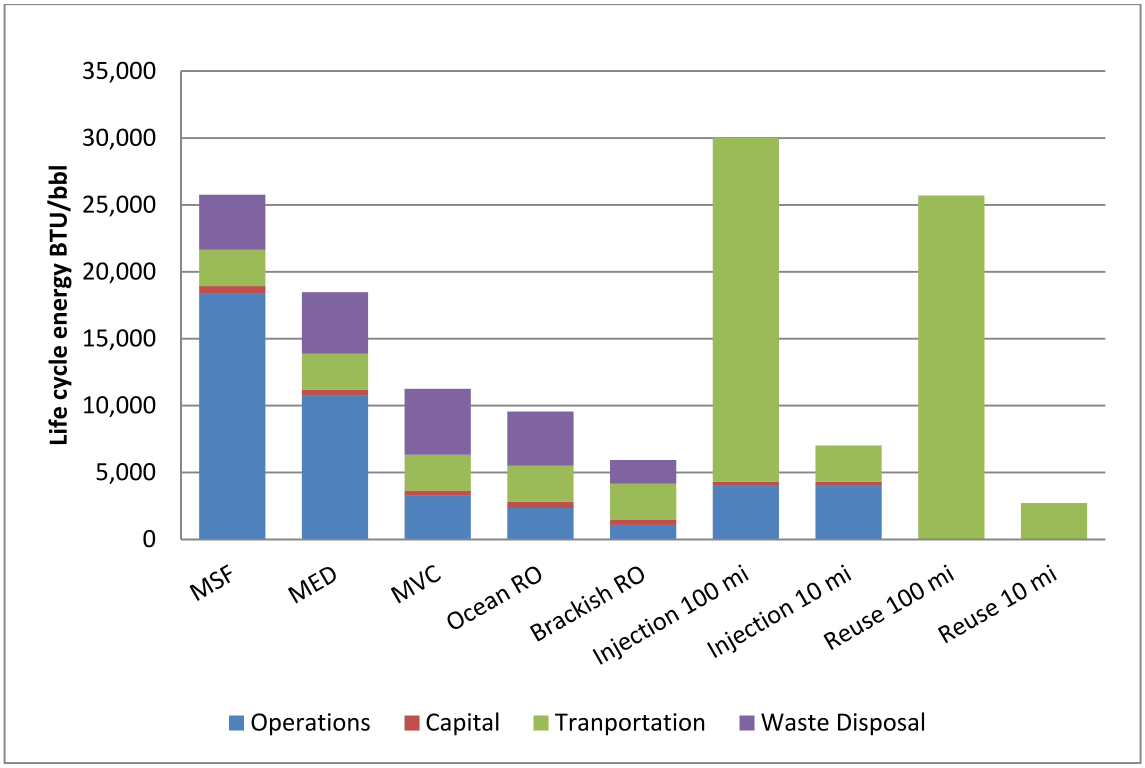

FIGURE 5 Life Cycle Energy Consumption from Extracted Water Management

injection, and reuse. Comparing injection and reuse at 10 mi versus 100 mi, however, illustrates the significant role transportation should play in the decision-making process. If a specialized reuse opportunity is far away, it may not justify the transportation costs versus treating or disposing of the water locally. Waste disposal burdens are nontrivial for most of the treatment scenarios. The high waste disposal costs are driven by the low recovery ratios for most of the seawater desalination technologies evaluated. Part of this is because the systems evaluated were optimized to minimize the cost of producing freshwater from seawater, without regard for waste disposal (as it is typically discharged to the ocean). It is possible that these same systems could be optimized to minimize waste at a cost of additional operational energy requirements. The brackish water RO system was the most efficient treatment system with even lower life cycle energy consumption than disposal, partially due to the high recovery ratio and low waste disposal costs. At a minimum, RO treatment should be strongly considered for low TDS brines.

To explore the impacts of economies of scale, the life cycle operational energy requirements for each individual treatment system were plotted against the treatment rate for the system, as shown in Figure 6. The graph shows strong clustering by technology, but no clear trend across technologies. Of note is that the operational energy consumption for the smaller scale produced water treatment systems was around an order of magnitude larger than most seawater desalination systems. The exact drivers of this significant discrepancy are unclear at this point, but the result is consistent with the difference in quoted costs for produced water treatment versus seawater desalination in the literature.

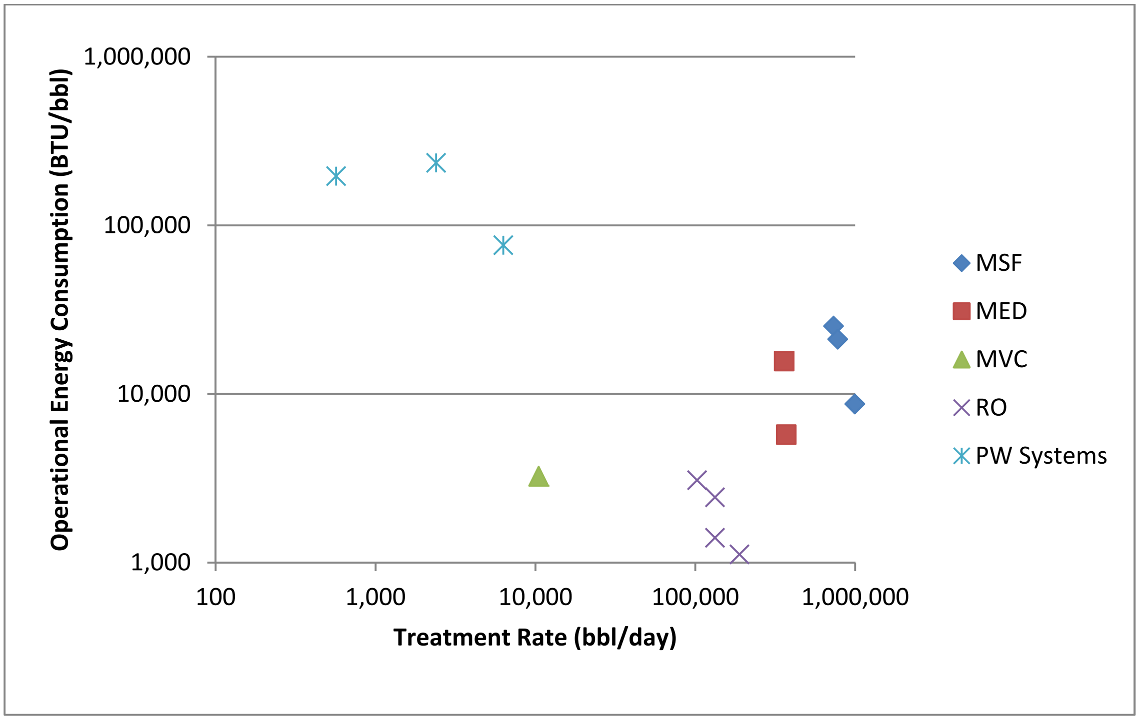

FIGURE 6 Operational Energy Consumption versus Treatment Rate

4.3 GREENHOUSE GAS EMISSIONS

Figures 7 and 8 show the life cycle GHG emissions. In Figure 7, the results are presented by life cycle stage in terms of grams of CO_2 equivalent per barrel of water managed. The general trends here are almost exactly equivalent to the trends in energy consumption discussed in Section 4.2. In Figure 8, the results are presented in terms of the fraction of total CO_2 stored that is re-emitted from extracted water management. It assumes that an equal volume of water is extracted from the formation to the volume of CO_2 stored. The specific gravity of CO_2 was assumed to be 0.72 for this calculation; however, the actual density varies slightly depending on the depth of the storage formation. In general, the lower this fraction, the more viable that water extraction becomes from a GHG perspective. Many of the extracted water management scenarios resulted in emissions equivalent to less than 1% of the stored CO_2. Emissions at these levels are unlikely to disqualify water extraction as a viable practice.

4.4 WATER SAVINGS

The net water impacts of the management strategies are summarized in Figure 9. They are presented in terms of the amount of water delivered or displaced for each barrel of water extracted from the formation. All water treatment systems that treat for TDS, including both thermal and membrane processes, result in two outputs—a clean water stream and a concentrated

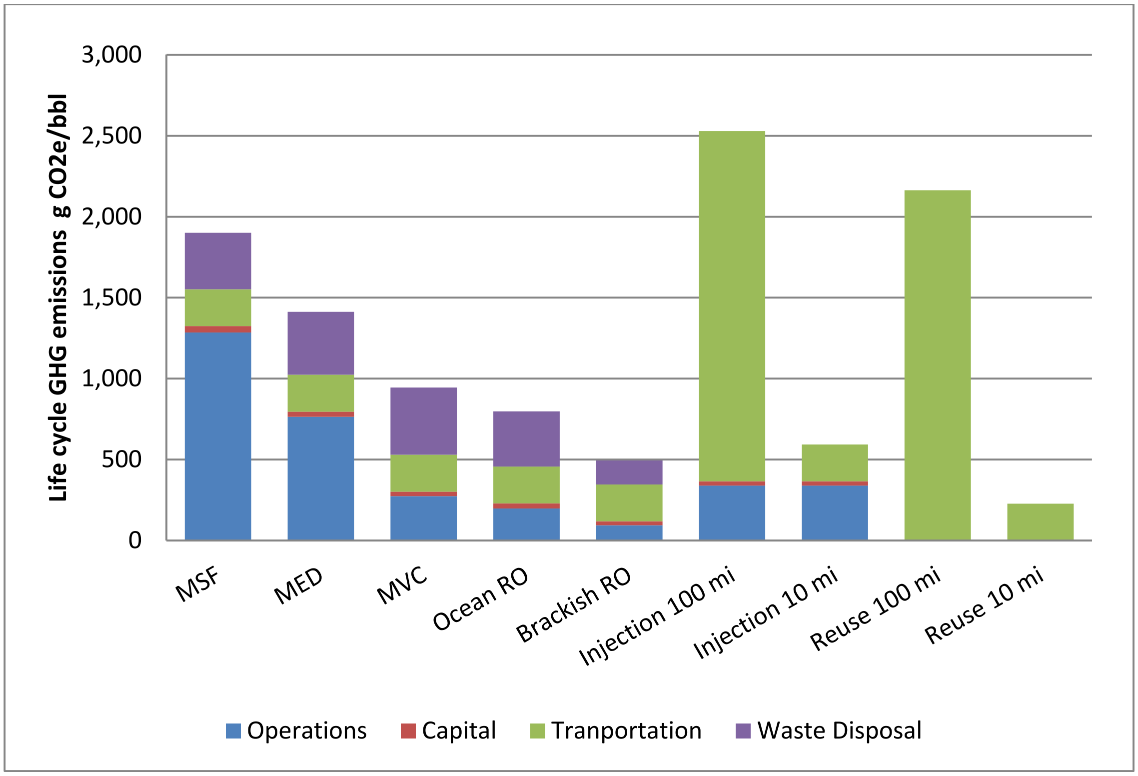

FIGURE 7 Life Cycle GHG Emissions from Extracted Water Management

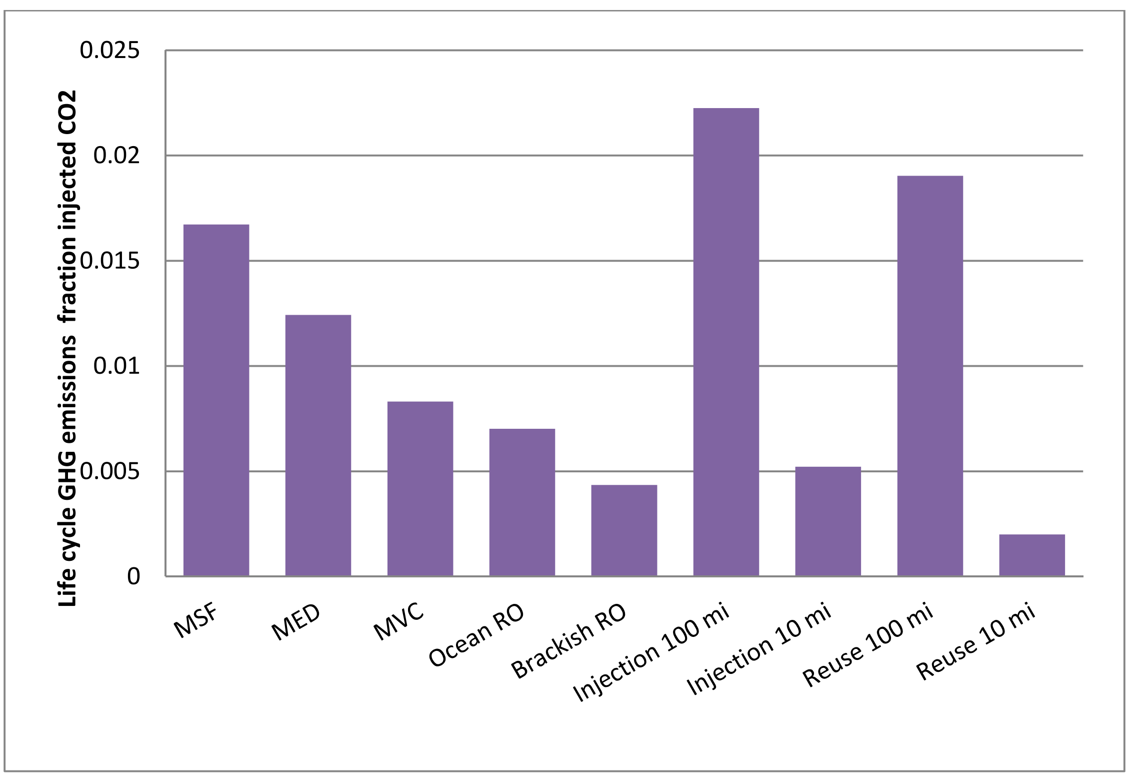

FIGURE 8 Life Cycle GHG Emissions as a Fraction of Carbon Stored

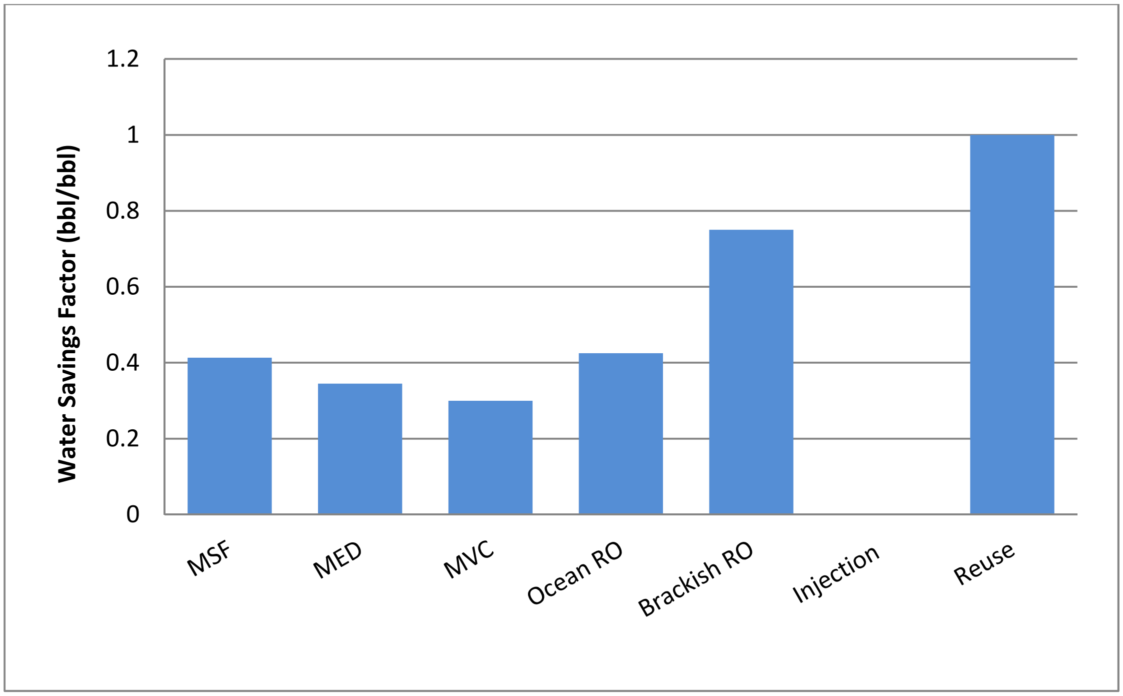

FIGURE 9 Life Cycle Net Water Savings for CCS Extracted Water Management

waste stream. Optimizing these systems for higher clean water output typically comes at a cost of higher capital costs or energy consumption. Conversely, optimizing them for energy consumption or capital costs often results in lower freshwater recovery rates. Most of the water treatment systems included here have been optimized to minimize the cost of freshwater supply. The water savings from treating extracted water with these systems is likely to be greater when optimized to minimize the cost of managing inland brine with concentrate disposal costs included. For the reuse without treatment scenario, it was assumed that 100% of the extracted water is able to be reused and that it can displace consumption of another water source. This is likely to be an optimistic assumption, as some fraction of the water may have to be treated or disposed of, or it may allow an activity that may not have occurred without access to the extracted water.

4.5 IMPORTANCE OF TRANSPORTATION

The results of the analysis are extremely sensitive to transportation mode and distance. As discussed in Section 3.5, pipeline and truck are the two primary modes of transporting water. While trucking exhibits virtually no economies of scale, the impact from pipelines decreases slightly with distance and significantly with volume as discussed in Section 3.5. Figures 10 and 11 illustrate the life cycle energy consumption and GHG emission by transportation mode as a function of the flow rate. The energy and GHG burdens associated with transportation increase nearly linearly with distance, thus the results are presented on a per-barrel-per-mile basis. Overall, trucking seems to be more efficient for low volumes; pipelines, however, become more efficient at flow rates above around 33,000 bbl/day (~1,000 gpm). Like all of the treatment

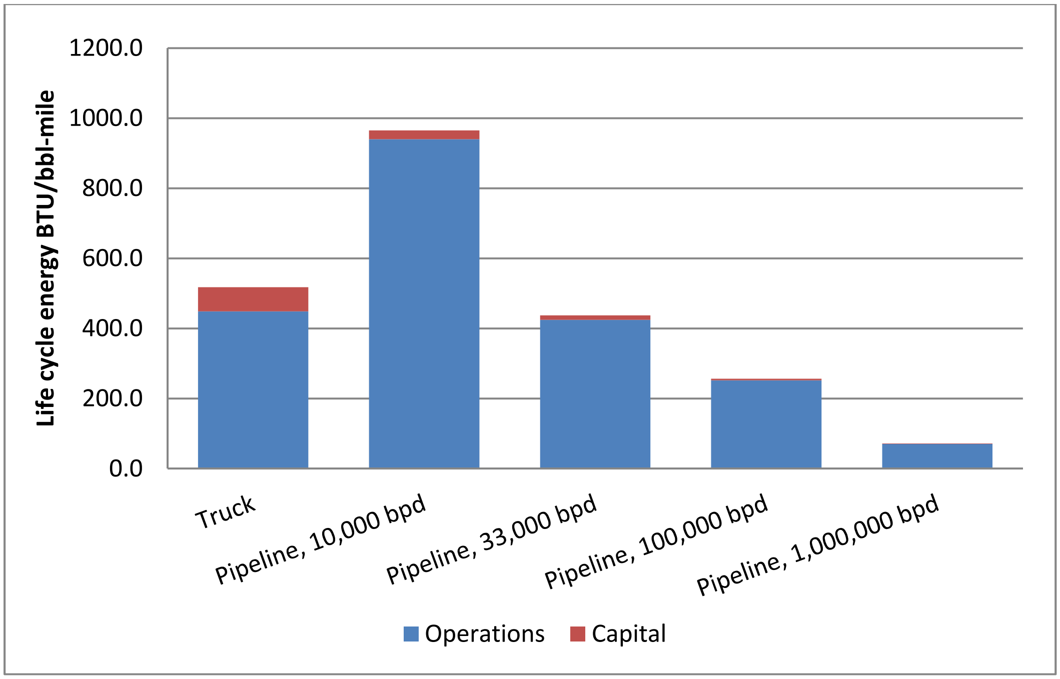

FIGURE 10 Transportation Economies of Scale, Life Cycle Energy Consumption

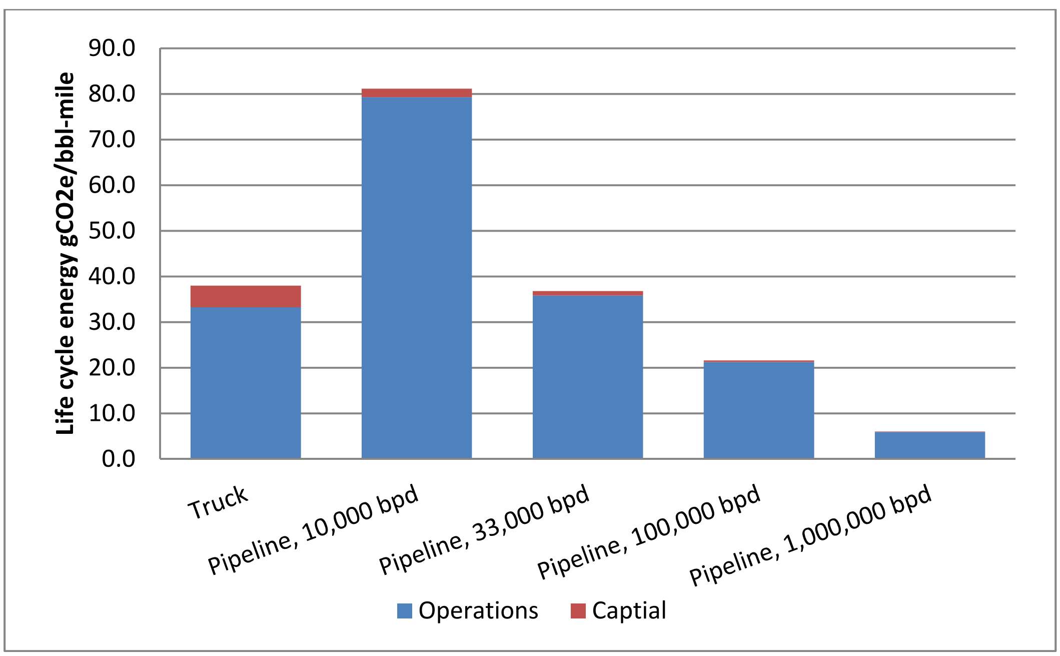

FIGURE 11 Transportation Economies of Scale, Life Cycle GHG Emissions

systems, the life cycle burdens associated with manufacturing the capital equipment used for transportation are significantly dwarfed by the burdens associated with system operations.

4.6 ESTIMATED COSTS

While the estimation of costs was not a primary goal of this effort, a significant amount of cost information was obtained in the process of developing the life cycle inventories. The total cost values presented here are rough approximations and should be viewed as order of magnitude estimations. Capital equipment was assumed to be amortized over 20 years with a discount rate of 8%. Electricity was assumed to cost $0.10 per kWh, and all thermal energy was assumed to be supplied by natural gas at a cost of $6/MCF. These energy costs are a bit higher than current market rates but are thought to be reasonable long-term estimates of future energy costs. The higher energy costs also help to offset the fact that no labor or operations costs are included in the calculations except for consumable chemicals.

Figure 12 shows the direct costs for each management strategy. These costs only include operations and capital costs. These costs were calculated as the average of the data points available, except for injection where the weighted average was taken on a volume-injected basis. Like the LCA results, the costs are presented on a per-barrel-managed basis. Figure 13 shows the full life cycle costs, including transportation and waste disposal, for the same scenarios presented in the LCA results. This range of extracted water management costs implies a total increase in cost of carbon sequestration of approximately $1 to $3 per ton of CO_2 sequestered, assuming a 1:1 volume displacement (excluding the long distance injection or reuse scenarios). This compares favorably with current estimated costs of between $60 and $100 per ton of carbon avoided for existing carbon capture, use, and storage (CCUS) technologies and DOE's intermediate-term goal of reducing costs to $40 per ton of carbon captured (NETL 2012; DOE 2013). Recent literature has suggested that most of the benefits of active reservoir management can be achieved at significantly less than a 1:1 volume extraction ratio, which would further reduce the incremental cost to the project (Berkholzer et al. 2012).

For comparison, the life cycle costs for brackish and ocean RO are equivalent to just over $1/m^3 and $3/m^3 of clean water produced, respectively. This is higher than most estimates quoted in the literature for RO treatment, but those estimates do not include the costs associated with transporting water to the treatment system or disposing of concentrate. These cost estimates are much lower than the cost data available for existing produced water management practices and should probably be considered a lower bound on actual costs for managing extracted water (Harto and Veil 2011). The costs quoted for the produced water management systems discussed in Section 3.3.1.4 ranged from $3 to $6 per barrel (Bruff et al. 2011; Hayes and Severin 2012).

4.7 SUMMARY

The results were combined and are summarized in Table 11. In addition to the energy, GHG emissions, water savings, and costs, the table also includes a list of additional qualitative

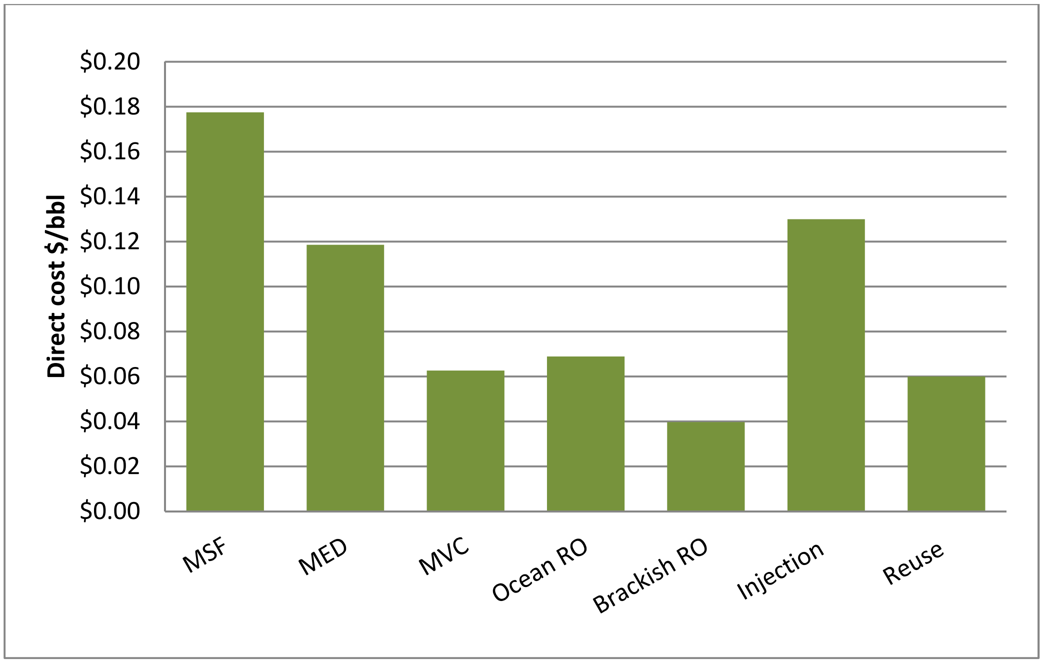

FIGURE 12 Direct Costs of Extracted Water Management per Barrel Managed

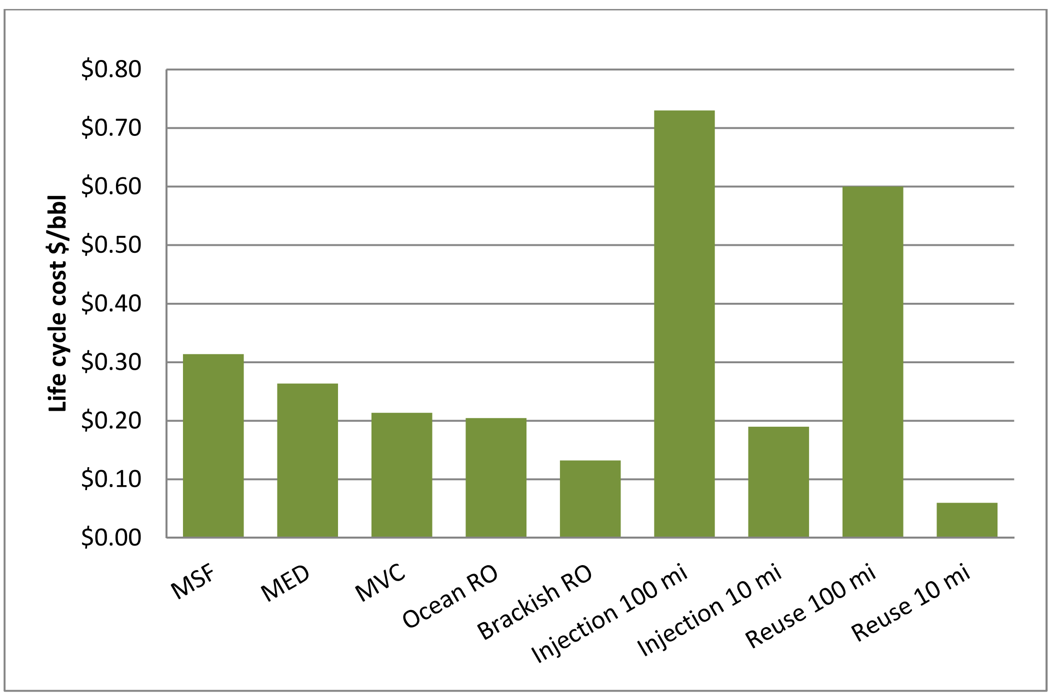

FIGURE 13 Life Cycle Costs of Extracted Water Management per Barrel Managed

factors discussed throughout that may affect the decision to use a specific treatment technology or management practice.

TABLE 11 Comparison of Extracted Water Management Options

Management Practice	Energy (BTU/bbl)	GHG Emissions (g CO_2e/bbl)	Water Savings (bbl saved/bbl extracted)	Estimated Costs ($/bbl)	Additional Considerations
Reuse with Minimal Treatment[a]	2,700	230	1	0.06	Distance to point of use; potential challenges matching the quantity, quality, and timing of extracted water supply to the demands of the end user
Multi-Stage Flash	26,000	1,900	0.41	0.31	Some scaling concerns; extracted water temperature (lower = better); limited operational flexibility, requires constant and reliable fee water source; can treat high TDS brines; most effective for large systems (50,000–75,000 m^3/day)
Multi-Effect Distillation	18,000	1,400	0.35	0.26	Moderate scaling concerns; extracted water temperature (lower = better); can treat high TDS brines
Mechanical Vapor Compression	11,000	940	0.3	0.21	Minimal scaling concerns; typically smaller scale systems (>5,000 m^3/day); can treat high TDS brines
Reverse Osmosis	5,900–9,600	490–800	0.43–0.75	0.13–0.21	Significant scaling and fouling concerns; membranes sensitive to temperature, pH, oxidizers, organics, algae, bacteria, particulates, and precipitates; appropriate pretreatment required to protect membranes; recovery ratio declines with higher TDS brines (typically limited to <50,000 ppm)
Underground Injection[a]	7,000	600	0	0.19	Availability of suitable nearby formations for injection; chemical compatibility of extracted water with selected injection formation; induced seismicity concerns
Solar Evaporation	Not Evaluated	Not Evaluated	0	Not Evaluated	Availability of cheap land; high natural evaporation rates and low precipitation required; availability of solid waste disposal; potential for NORM disposal issues
Thermal Evaporation	560,000[b]	Not Evaluated	0	Not Evaluated	Likely only viable when significant quantities of low-grade waste heat or stranded gas are available; availability of solid waste disposal; potential for NORM disposal issues

[a] Numbers presented are based upon 10-mi transportation distance scenario only.

[b] Value for direct operational energy consumption only; not full life cycle value.

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5 CONCLUSIONS

This study has shown that brines extracted from carbon sequestration formations can potentially be managed with acceptable environmental and financial burdens. Importantly, when the appropriate management practices are selected, the management of extracted brines should not significantly contribute to the carbon footprint of the overall sequestration process. Selection of management practices will be highly site specific and highly dependent upon brine properties. The preferred option for managing brines is to reuse the brines at a nearby location with minimal treatment. However, the availability of suitable reuse applications that fit these requirements is likely to limit this management option. Also, the cost and environmental burdens associated with transporting water over longer distances favors treatment and reuse nearby over transporting water longer distances for reuse.

The preferred treatment method is RO, especially for brines with low TDS. However, its applicability is limited for higher TDS brines, and it is highly vulnerable to scale and sensitive to inlet water properties, potentially necessitating significant pretreatment for many extracted brines. Further study is recommended to evaluate the efficacy of RO in treating extracted brines from different formations and to improve understanding of pretreatment requirements and costs. When RO is not viable, MVC appears to be the best thermal treatment option, with GHG emissions and energy consumption only marginally higher than seawater RO. MVC is more flexible in the TDS concentrations that it can handle and is not as sensitive to inlet brine conditions.

Deep well injection is probably the only option for disposing of brines where treatment is not viable in most areas. The energy, GHG, and financial burdens of disposal, however, do not appear to be significantly lower than those associated with treatment and reuse, thus it should be viewed as a last option. However, it may be necessary for many high TDS brines and concentrates generated by treatment systems. Overall, transportation distance should be a major driver of the decision-making process and should be minimized to the extent possible.

Significant cost and energy consumption discrepancies continue to exist between large-scale systems for seawater desalination and smaller scale treatment systems for produced water management. This study has attempted to narrow that gap by including transportation and concentrate disposal costs, but significant uncertainty remains. In addition, further study is needed to balance the costs and environmental burdens associated with extracted water management with the benefits associated with active reservoir management.

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Environmental Science Division
Argonne National Laboratory
9700 South Cass Avenue, Bldg. 240
Argonne, IL 60439-4847

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